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IN THE UNITED STATES DISTRICT COURT
FOR THE DISTRICT OF MONTANA
MISSOULA DIVISION

ALLIANCE FOR THE WILD ROCKIES, Plaintiff, v. CHRISTOPHER SAVAGE, Kootenai National Forest Supervisor, et al., And LINCOLN COUNTY, et al.,	CV-15-54-M-DLC BRIEF OF AMICI CURIAE Blue Mountains Biodiversity Project; Buckeye Forest Council, Inc., Conservation Congress, Friends of Bell Smith Springs, Friends of the Bitterroot, Friends of the Clearwater, Friends of the Wild Swan, Heartwood, Idaho Sporting Congress, Kootenai Environmental Alliance, Montana Ecosystems Defense Council, Native Ecosystems Council, Sequoia ForestKeeper, Swan View Coalition, Tennessee Heartwood, Western Lands Project, Yellowstone to Uintas Connection, Dr. Brian Horejsi, Barry Rosenberg, Dr. Chad Hanson, Dick Artley, Doug Soehren, Frank Robey, George Wuerthner, Jeff Juel, Larry Campbell, Paul Edwards, Polly Pfister and Rocky Smith.

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I. Amici

Amici are a collection of conservation groups and individuals who work to protect and conserve public lands throughout the United States. Amici recognize that National Forests have intrinsic and natural value, are exceedingly important to conserve habitat and species, watersheds and water quality, and represent one of our best chances at combatting the negative effects of climate change.

One thing that Amici all have in common is, though they may work in specific States or regions, they recognize the national ecological importance of federal public lands, regardless of geographic location, especially when it comes to clean water, aesthetic enjoyment, conserving native ecosystems, species conservation and climate change.

II. The Origin and Purpose of National Forests

"All public lands of the United States are held by it in trust for the people of the United States." *Utah Power & Light v. United States*, 243 U.S. 389, 409, 37 S.Ct. 387, 61 L.Ed. 791 (1916). Yet, a disturbing trend has been developing in this country that emphasizes local control over National Forests and wildlands. Even here, where this court is reviewing the merits of Alliance for the Wild Rockies' legal claims, several of the participants in this litigation are using their participation to infuse these proceedings with arguments focused on how their private interests will be affected by the outcome of this case. However, National Forests were not created to serve local interests. In fact, forest reserves, as they were originally called[1], were created specifically to reserve forest lands from the public domain and protect them, whereas previously anyone could privately benefit from these areas in accordance with their efforts and abilities, and where application of previous acts of Congress, the Timber Culture Act of 1873 and the Homestead Act of 1873, were riddled with fraud for personal gain. *United States v. Parker*, 36 F.Supp. 3d 550, 569 (W.D.NC. 2014). The Forest Reserve Act (originally known as the Act of March 3, 1891) authorized the President of the United States to "set apart and reserve, from time to time, in any state or territory having public land bearing forests, any part of the public lands wholly or in part covered with timber or undergrowth, whether of commercial value or not, as *public* reservations; the same to be by public proclamation, declaring the establishment of such reservations and the limits thereof." *Chicago, M. & St. P. Ry. Co. of Idaho v. United Stat*es, 218 F. 288, 292 (9th Cir. 1914), *aff'd* 244 U.S. 351, 37 S. Ct. 625, 61 L. Ed. 1184 (1917), *citing* 26 Stat. 1103 (*emphasis added*).

While it is true that, since the Forest Reserve Act was passed, there has been pushback from Western States and timber interests to prioritize and increase timber production on these lands [*see e.g.*, The Forest Management Act of June 4, 1897 (30 Stat. 11); The Weeks Act (36 Stat.961)], National Forests are firmly established as

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[1] Administration of forest reserves was transferred from the Secretary of the Interior to the Secretary of Agriculture in 1905 (Act of Feb. 1, 1905, 33 Stat.628, 16 USC §472) and what would become the National Forest System (governing the forest reserves) was established by the Weeks Forestry Act in 1911 [Act of Mar. 1, 1911, ch. 186, §§ 4-14, 36 Stat.961, 962-3 (1911)].

areas which Congress has directed to be managed for multiple use for national and public purposes, not local gain, and shall be, as far as practicable, controlled and administered in accordance with the following purposes: improving and protecting the forests within the boundaries; securing favorable conditions of water flows; furnishing a continuous supply of timber for the use and necessities of citizens (16 U.S.C. §475); and providing for outdoor recreation, range, timber, watershed and wildlife and fish purposes (16 U.S.C. §528). *See also Sierra Club v. Morton*, 405 U.S. 727, 748 and ftnt. 7, 92 S.Ct. 1361, 1372 (1972).

Defendants in this case assert that the East Reservoir Project complies with the law in part because they (US Forest Service) received support for, or gave support to (Kootenai Collaborators), the East Reservoir Project. *See e.g.*, Doc. 40, pp. 1-5. However, the fact that a group of local, like-minded individuals/groups, with superficial differences[2], agree on a multi-year logging project, which will economically benefit a subset of the local community and the Forest Service, is not a proxy for legal compliance. *Id.* In fact, it is not even evidence that what has been proposed is a good

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[2] For example, the Montana Wilderness Association is fully committed to extensively logging outside of Wilderness areas so long as Wilderness areas are not logged, and no wilderness areas are involved in the East Reservoir Project; and the Yaak Valley Forest Council and Lands Project both are invested in collaboration and fully support logging for local economies. In fact, the Kootenai Forest Stakeholders Coalition is fully aligned with the Forest Service's timber objectives, approving of the annual logging of 70-90 million board feet of saw logs from the Kootenai National Forest and even encouraging active timber management on lands which have been designated as *unsuitable* for timber production (https://www.workingforest.com/forest-coalition-adopts-guidelines/).

idea or that it will be ecologically beneficial. All that this agreement indicates is that these individuals and organizations are invested in a particular subset of their community and its human inhabitants. Certainly Defendants disparage Alliance for the Wild Rockies for not getting behind the Project, but Alliance's interests are broader than those of a local grouping of collaborators (as this Amici Brief attests), and these interests are in fact divergent from those of Defendants. Alliance for the Wild Rockies' mission is to "secure the ecological integrity of the Wild Rockies Bioregion"; they view the region through the science of conservation biology and see the thousands of miles of logging roads (and the logging these roads facilitated) as a fundamental problem because they "fragment[] ecosystems and eliminate habitat for sensitive species" (www.allianceforthewildrockies.org). Their solution is to use every tool available to them (including going to Court to ensure compliance with environmental laws passed by Congress) to "[p]rotect large blocks of secure habitat to serve as core areas, and connect these blocks through linkage corridors" for imperiled wildlife. *Id.* The East Reservoir Project authorizes over 8,000 acres of commercial logging and almost 6,000 acres of thinning, increases the total amount of roads within the project area (including permanent and temporary), and amends the Forest Plan to increase the size of areas that can be clearcut, locating many of these "regeneration" harvests in the critical habitat of the ESA-listed Canada Lynx where it will eliminate foraging use for 10 (summer) to 30 (winter) years. Doc. 35, ¶¶15-17, 19-22, 36, 55-57 and 61; Doc. 19, ¶¶34 and 58-60. Instead of being condemned, Alliance for the Wild

Rockies should be recognized for representing a broad national and public interest consistent with the purposes for which National Forests were established, and one that extends far beyond the borders of County and State.

As intended, National Forests transcend the needs of the State and the local economies. With rural development and 100 + years of resource extraction on public and private lands, the importance of National Forests as refuges for native ecosystems and wildlife, sources of clean water, and as areas which can help combat the anticipated negative effects of climate change, has been amplified. Amici from as far away as Tennessee, Maine and Ohio are affected by the logging proposed in the East Reservoir Project and its impact on recovering wildlife species (*see* ESA section below). This logging project would also affect Amici by reducing the project area's ability to store carbon (through the removal of tens of thousands of trees), while increasing carbon emissions into the atmosphere from logging and road building activities which will persist through the life of the project (including chainsaws, fellerbunchers, log trucks, bulldozers and increased mill activity, as well as reduced carbon storage in these forests), both of which will contribute to climate change. AR30278, AR17457, Exhibit A (abstract from Campbell et al. 2011). Put simply, State borders and County lines no longer contain the adverse effects from resource extraction, nor do they contain the fiscal costs of such activities. The East Reservoir Project would cost American taxpayers, not just Lincoln County taxpayers, over $2.5

million dollars to implement. Doc. 19, ¶39. The Project is forecast to generate less than half of that amount in revenue to Lincoln County. Doc. 41, ¶35.

In addition, the supposed national and public benefits of the extensive logging approved by the East Reservoir Project are shrouded in fear and misunderstanding about fire's role in the forest ecosystem, and the unfounded belief that extensive landscape level logging, including removing large swaths of the forest ecosystem through clearcutting, or as Defendants like to call it "regeneration" harvest, will reduce future fire risk and intensity or increase the ecological resilience of an ecosystem adapted to mixed-severity fire.

There has been extensive research in forests about the ecological benefits of mixed-severity (which includes high-severity) fire over the past two decades, so much so that last year science and academic publishing giant Elsevier published a four hundred page book, *The Ecological Importance of Mixed-Severity Fires: Nature's Phoenix*[3] which synthesizes published, peer-reviewed science investigating the value of mixed- and high-severity fires for biodiversity (hereafter "*Nature's Phoenix*"). This book includes research documenting the benefits of high-intensity wildfire patches for

[3] http://store.elsevier.com/The-Ecological-Importance-of-Mixed-Severity-Fires/Dominick-DellaSala/isbn-9780128027493/. Judicial notice of this publication is appropriate Under FRE 201(b). *Se,e e.g., Von Saher v. Norton Simon Museum of Art at Pasadena*, 578 F.3d 1016 (9th Cir. 2009), *amended and superseded, Von Saher v. Norton Simon Museum of Art at Pasadena*, 592 F.3d 954 (9th Cir. 2010).

species at issue in this case,[4] as well as a discussion of mechanical "thinning" logging, approved here, and its inability to reduce the chances of a fire burning in a given area, or alter the intensity of a fire, should one begin under high fire weather conditions, because overwhelmingly weather, not vegetation, drives fire behavior. Exhibit B (*Nature's Phoenix*, Ch. 13, pp. 382-384). There is also strong consensus from over 260 US scientists that clearcutting ("regeneration logging") destroys and removes wildlife habitat in forests, does not create natural habitat heterogeneity, and does not mimic the ecological benefits of fire. Exhibit D (letter from 262 scientists to Congress). Finally, ecological resilience, which Defendants imply they are creating through this project, is not the absence of natural disturbances like wildfire or beetle kill, rather it is the opposite. AR3226, AR32351-64, AR32222, Exhibit B (*Nature's Phoenix*, Chapter 1,

[4] Mixed-severity fires, and in particular patches of high-severity fire, benefit Grizzly bears by: increasing cover of berry producing shrubs (such as huckleberry) that the bears rely upon to get fat before winter; promoting regeneration of whitebark pine—the seeds of which are an important food source for the bears; and by improving foraging habitat for prey species such as moose, elk, and deer. AR25569-71; AR25575-76 and Exhibit B (*Nature's Phoenix*, Ch. 4, pp. 89, 101). Grizzly Bears avoid logged and roaded areas because they do not provide the forest structures of post-fire habitat which benefit this species**.** *Id.* Natural, temporary pulses of sedimentation after mixed-intensity wildfire, in areas that are not post-fire logged, assist in restoring populations of native trout, including the bull trout, by creating gravel/sand beds in streams that serve as spawning grounds, and by increasing native aquatic insects upon which fish feed. AR15019; *see also* Exhibit B (*Nature's Phoenix*, Chapter 5, pp. 120-121). Chronic sedimentation, which occurs after mechanical disturbances of soil by road building and logging, does not provide the same benefits, and in fact degrades stream ecosystems. *Id.* and Exhibit B, p. 138. And, Canada Lynx benefit from patches of high-intensity fire, as well as patches of forest with high tree mortality from native bark beetles, due to increased prey/foraging opportunities, whereas Lynx avoid logged areas, and scientists recognize fire suppression and logging as threats to the Lynx. AR30059-63 and Exhibit C (abstracts from Fox 1978 and Mowat and Slough 2003).

pp. 12-13). Logging to prevent mixed-severity fires, as proposed by the East Reservoir Project, reduces the ecological resiliency of a forest, rather than achieving it. This is because processes like fire create and maintain the full range of natural habitat types and heterogeneity across the forest landscape, which in turn maintain the full range of native plant and animal biodiversity in the forest. Exhibit E (Thompson et al 2009)[5]. What Defendants are promoting here is the human control of the forest ecosystem through mechanical means in order to maintain unnatural stasis by eliminating, suppressing or altering natural disturbances such as wildfire, to facilitate the orderly extraction of commercial resources for human use. This is known as engineering resilience and it is the antithesis of ecological resilience and conservation of native biodiversity. *Id.*

III. The Endangered Species Act:

Similar to Congressional acts which established National Forests for benefit of the Nation, so too was the Endangered Species Act passed for the benefit of all Americans. The Endangered Species Act was enacted after Congress determined that:

> **(1)** various species of fish, wildlife, and plants in the United States have been rendered extinct as a consequence of economic growth and development untempered by adequate concern and conservation;

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5 Thompson, I. *et al.* (2009) *Forest Resilience, Biodiversity, and Climate Change. A Synthesis of the Biodiversity/Resilience/Stability Relationship in Forest Ecosystems.* CBD Technical Series No. 43, Secretariat of the Convention on Biological Diversity, United Nations Environment Program.

> **(2)** other species of fish, wildlife, and plants have been so depleted in numbers that they are in danger of or threatened with extinction;

16 U.S.C.A. §1531(a). It contains a comprehensive scheme with the broad purpose of providing "a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved". *Id.* at §1531(b). In the course of establishing this conservation program Congress found that "these [threatened or endangered] species of fish, wildlife, and plants *are of esthetic, ecological, educational, historical, recreational, and scientific value to the Nation and its people*". *Id.* at 1531(a)(3) (emphasis added).

At issue in this case are three species listed as threatened under the Endangered Species Act, the Grizzly Bear and Canada Lynx which will likely be adversely affected, and the Bull Trout which may be affected, by the East Reservoir Project. These species are not just of value (or a hindrance as the case may be) to local interests, but they are of value to the American public, and represent a national interest protected by the Act. Where once Grizzly bears were over 50,000 strong roaming the United States from Alaska to Mexico and California to Ohio, in the lower 48 U.S. states, small breeding populations are currently confined to Wyoming, Idaho and Montana and Washington, with an approximate population of only 1,500 bears between these four states. *Alliance for Wild Rockies v. Bradford*, 35 F.Supp.3d 1246, 1249 (D. Mont.2014). It has taken almost 40 years for this small recovery to occur, and the fact that there are still Grizzlies roaming the lands of the United States is a great comfort to Amici, some of whom study the species, and many of whom recreate in their habitat and delight in the esthetic and ecological evidence of their presence (including potential sightings). Even those who may never have the privilege of traveling to these states and being in the presence of, or walking on the same trails as, these iconic omnivores care deeply about their conservation and recovery. Just knowing the Grizzly bears are there, being able to read about their behaviors in scientific studies and other publications, watching videos or viewing photos of Grizzly bears in the lower 48 states, and learning of their re-establishment in areas which they historically occupied but had been extirpated from by hunting and trapping, all represent hope for Amici. Hope that we as a country and a people can care for and conserve species other than our own, hope that progress can occur without obliterating the Nation's heritage in fish, wildlife and plants, and hope that places wild enough to support Grizzly Bears will continue to exist in this country. Natural resource extraction projects, such as the East Reservoir Project, that will likely harm the continued recovery of these species and threaten the establishment of healthy viable populations which continue in perpetuity, are of great concern to Amici and their members. These same interests also extend to the threatened Canada Lynx and Bull Trout.

As has been acknowledged by the Forest Service here, Bull Trout may be present on the Forest, are known to be in Lake Koocanusa (from which they migrate into streams for spawning and rearing), and have historically been present within Fivemile Creek, both of which are watersheds within the [Project] analysis area that may be adversely affected by the project activities, and chronic sedimentation from logging and roads negatively impacts fish reproduction and survival. Doc 35, ¶¶79-82 and 88-89.

Federal Defendants admit that logging under the East Reservoir Project will occur on over 5,000 acres of Lynx critical habitat and that the 1,269 of these acres which will be clearcut will impede foraging behavior of the Lynx and its prey (snowshoe hare) for one to three decades depending on the season. Doc. 35, ¶¶55-58 and 61. They also agree that in winter Lynx stay in their home ranges and utilize a narrow subset of habitat (dense old forest which would be removed by clearcutting), that the proposed logging would remove 14 areas of movement corridors, and that Lynx movements would be further impeded by the 3,500+ acres of forest thinning. *Id.* at ¶¶58 and 63-64. The East Reservoir Project would more than double the amount of habitat within the Lynx analysis unit which is unsuitable for winter snowshoe hares and more than triples the percentage of habitat in this Lynx analysis unit which would be rendered unsuitable through clearcut/regeneration logging. *Id.* at ¶¶58-59. Even still, the Forest Service claims that the East Reservoir Project is not likely to adversely affect lynx or lynx critical habitat. Doc. 17, §IV(C)2 at pp. 19-23.

The East Reservoir Project would impact the Tobacco Bears Outside Recovery Zone, an area where the federally listed Grizzly Bear is known to occur. Doc. 35, ¶¶40-42. The Forest Service concedes that the existing condition of the Tobacco Zone is already having adverse effects on grizzly bears. *Id.* at ¶49. The Project's logging would further reduce vegetative cover in the Tobacco Zone and create additional distances for bears to cover, leaving them more vulnerable to poaching (*Id.* at ¶¶53-54). In addition, there would be an increase within this Zone in the total linear miles of roads. Doc. 19, ¶48. Total linear miles of roads and motorized trails would also increase elsewhere in the Project area where Grizzly Bears and their prey Elk may occur. Doc. 35, ¶70-78. Motorized access poses the most imminent threat to grizzly habitat and negatively affects habitat security for elk and grizzly bears. Doc. 35, ¶¶66-69. The Forest Service also claims that the East Reservoir Project is not likely to adversely affect grizzly bears. Doc. 17, §IV(B)(2) at pp. 10-14.

The East Reservoir Project does not safeguard, for the benefit of all citizens, the Nation's heritage in fish and wildlife. 16 U.S.C. §1531(5). Full compliance with the Endangered Species Act is required to protect the public's interest in our Nation's fish and wildlife. This includes accurately representing when a threatened or endangered species "may" be affected by a resource extraction project, and admitting when such a project is likely to adversely affect listed species. *Karuk Tribe of California v USFS*, 681 F.3d 1006,1027 (9th Cir.2012) (*en banc*); ESA §7 Handbook at xv-xvi, 4-1 and B-56. Neither of these required safeguards has been met in the preparation of the East Reservoir Project.

DATED this 24th day of January, 2016

By: */s/ Rachel M. Fazio*
Rachel M. Fazio
Attorney for Amici Curiae

Certificate of Service

I, Rachel M. Fazio, certify that on the 24th day of January 2016, I filed a copy of the preceding document and Exhibits A-E through the Court's CM/ECF electronic filing system, affecting service on all parties to this litigation.

/s/ Rachel M. Fazio
Rachel M. Fazio

Frontiers in Ecology and the Environment

Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions?

John L Campbell, Mark E Harmon, and Stephen R Mitchell

Front Ecol Environ 2011; doi:10.1890/110057

This article is citable (as shown above) and is released from embargo once it is posted to the *Frontiers* e-View site (www.frontiersinecology.org).

Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions?

John L Campbell[1*], Mark E Harmon[1], and Stephen R Mitchell[2]

It has been suggested that thinning trees and other fuel-reduction practices aimed at reducing the probability of high-severity forest fire are consistent with efforts to keep carbon (C) sequestered in terrestrial pools, and that such practices should therefore be rewarded rather than penalized in C-accounting schemes. By evaluating how fuel treatments, wildfire, and their interactions affect forest C stocks across a wide range of spatial and temporal scales, we conclude that this is extremely unlikely. Our review reveals high C losses associated with fuel treatment, only modest differences in the combustive losses associated with high-severity fire and the low-severity fire that fuel treatment is meant to encourage, and a low likelihood that treated forests will be exposed to fire. Although fuel-reduction treatments may be necessary to restore historical functionality to fire-suppressed ecosystems, we found little credible evidence that such efforts have the added benefit of increasing terrestrial C stocks.

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Various levels of tree removal, often paired with prescribed burning, are a management tool commonly used in fire-prone forests to reduce fuel quantity, fuel continuity, and the associated risk of high-severity forest fire. Collectively referred to as "fuel-reduction treatments", such practices are increasingly used across semiarid forests of the western US, where a century of fire suppression has allowed fuels to accumulate to levels deemed unacceptably hazardous. The efficacy of fuel-reduction treatments in temporarily reducing fire hazard in forests is generally accepted (Agee and Skinner 2005; Ager *et al.* 2007; Stephens *et al.* 2009a) and, depending on the prescription, may serve additional management objectives, including the restoration of native species composition, protection from insect and pathogen outbreaks, and provision of wood products and associated employment opportunities.

In a nutshell:

- Carbon (C) losses incurred with fuel removal generally exceed what is protected from combustion should the treated area burn
- Even among fire-prone forests, one must treat about ten locations to influence future fire behavior in a single location
- Over multiple fire cycles, forests that burn less often store more C than forests that burn more often
- Only when treatments change the equilibrium between growth and mortality can they alter long-term C storage

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Recently, several authors have suggested that fuel-reduction treatments are also consistent with efforts to sequester C in forest biomass, thus reducing atmospheric carbon dioxide (CO_2) levels (Frinkral and Evans 2008; Hurteau *et al.* 2008; Hurteau and North 2009; Stephens *et al.* 2009b). It is argued that short-term losses in forest biomass associated with fuel-reduction treatments are more than made up for by the reduction of future wildfire emissions, and thinning practices aimed at reducing the probability of high-severity fire should therefore be given incentives rather than be penalized in C-accounting programs. This is an appealing notion that aligns the practice of forest thinning with four of the most pressing environmental and societal concerns facing forest managers in this region today – namely, fire hazard, economic stimulus, so-called forest health, and climate-change mitigation. However, we believe that current claims that fuel-reduction treatments function to increase forest C sequestration are based on specific and sometimes unrealistic assumptions regarding treatment efficacy, wildfire emissions, and wildfire burn probability.

In this paper, we combine empirical data from various fire-prone, semiarid conifer forests of the western US (where issues of wildfire and fuel management are most relevant) with basic principles of forest growth, mortality, decomposition, and combustion. Our goal is to provide a complete picture of how fuel treatments and wildfires affect aboveground forest C stocks by examining these disturbance events (1) for a single forest patch, (2) across an entire forest landscape, (3) after a single disturbance, and (4) over multiple disturbances. Finally, we consider how wildfire and/or fuel treatments could initiate alternate equilibrium states

THE ECOLOGICAL IMPORTANCE OF MIXED-SEVERITY FIRES
NATURE'S PHOENIX

Edited by Dominick A. DellaSala and Chad T. Hanson

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Chapter 1

Setting the Stage for Mixed- and High-Severity Fire

Chad T. Hanson[1], Rosemary L. Sherriff[2], Richard L. Hutto[3], Dominick A. DellaSala[4], Thomas T. Veblen[5] and William L. Baker[6]

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1.1 EARLIER HYPOTHESES AND CURRENT RESEARCH

In the late 19th century and early 20th century, fire—especially patches of high severity wherein most or all of the dominant vegetation is killed—was generally considered to be a categorically destructive force. Clements (1936) hypothesized that the mature/old state of vegetation would result in a stable "climax" condition and described natural disturbance forces such as fire as a threat to this state, characterizing mature forest that experienced high-severity fire as a "disclimax" state. One early report opined that there is no excuse or justification for allowing fires to continue to occur at all in chaparral and forest ecosystems (Kinney, 1900). After a series of large fires in North America in 1910, land managers established a policy goal of the complete elimination of fire from all North American forests (a "one size fits all" policy) through unsuccessful attempts to achieve 100% fire suppression (Pyne, 1982; Egan, 2010). Through the mid-20th century, and in recent decades, views have shifted to broadly acknowledge the importance of low- and low/moderate-severity fire. In this chapter we focus on drier montane forests of western North America as a case study of how diverse, competing, and rather complex sets of evidence are converging on a new story that embraces not just low-severity fire but also mixed- and high-severity fire in these ecosystems. Thus this chapter exemplifies how mixed- and high-severity fire is being better understood and appreciated as scientific evidence accumulates.

A commonly articulated hypothesis is that dry forests at low elevations in western North America were historically open and park-like, and heavily dominated by low-severity and low/moderate-severity fire (Weaver, 1943; Cooper, Only 16% of the study area recorded a shift from historical low-severity fire to a higher potential for crown fire today. A historical fire regime of more frequent, low-severity fires at elevations below 2260 m is consistent with the view among land managers that these forests be thinned both to restore historical structure and to reduce fuels in this area of widespread exurban development. By contrast, at higher elevations in the upper montane zone (i.e., 2260-3000 m), mixed-severity fires were predominant historically and continue to be so today. Thus thinning treatments at higher elevations of the montane zone are inappropriate if the management goal is ecological restoration. Comparison of the severity of nine large fires that occurred between 2000 and 2012 with the severity of fires before the 20th century revealed no significant increase in fire severity from the historical to the modern period except for a few fires that occurred within the lowest elevations (16%) of the montane study area (Sherriff et al., 2014). This spatially extensive tree ring–based reconstruction is strongly corroborated by land survey records of higher-severity fire patches across the same area (Williams and Baker, 2012b).

Charcoal and Sediment Reconstructions

Paleoecologists have explored fire-induced sediment layers in alluvial fans (e.g., Pierce et al., 2004) and charcoal sediments (e.g., Whitlock et al., 2008; Colombaroli and Gavin, 2010; Jenkins et al., 2011; Marlon et al., 2012) to reconstruct historical fire occurrence. They found numerous periods of large and severe fire activity over the past several centuries and millennia in North American mixed-conifer and ponderosa pine forests (see Chapter 9 for many additional citations). Thus paleoecological methods and evidence further corroborate findings based upon other methods, discussed above, regarding historical mixed- and high-severity fire in these forests.

1.2 ECOSYSTEM RESILIENCE AND MIXED- AND HIGH-SEVERITY FIRE

Along with the surge in scientific investigation into historical fire regimes over the past 10-15 years has come enhanced understanding of the naturalness and ecological importance of mixed- and high-severity fire in many forest and shrub ecosystems. Contrary to the historical assumption that higher-severity fire is inherently unnatural and ecologically damaging, mounting evidence suggests otherwise. Ecologists now conclude that in vegetation types with mixed- and high-severity fire regimes, fire-mediated age-class diversity is essential to the full complement of native biodiversity and fosters ecological resilience and integrity in montane forests of North America (Hutto, 1995, 2008; Swanson et al., 2011; Bond et al., 2012; Williams and Baker, 2012a; DellaSala et al., 2014). Ecological resilience is essentially the opposite of "engineering resilience," which pertains to the suppression of natural disturbance to achieve stasis and control of resources

(Thompson et al., 2009). Ecological resilience is the ability to ultimately return to predisturbance vegetation types after a natural disturbance, including higher-severity fire. This sort of dynamic equilibrium, where a varied spectrum of succession stages is present across the larger landscape, tends to maintain the full complement of native biodiversity on the landscape (Thompson et al., 2009). Forests that are purported to be burning at unprecedented levels of high-severity fire are generally responding well in terms of the forest succession process and native biodiversity (see Chapters 2–5), so the widespread fear of too much severe fire seems to be unfounded in the vast majority of cases (see, e.g., Kotliar et al., 2002; Bond et al., 2009; Donato et al., 2009; Burnett et al., 2010; Malison and Baxter, 2010; Williams and Baker, 2012a, 2013; Buchalski et al., 2013; Baker, 2014; Odion et al., 2014; Sherriff et al., 2014; Hanson and Odion, 2015a). We acknowledge that more research is needed for some forest regions, such as some areas of the southwestern United States experiencing increasing fire severity (Dillon et al., 2011), to determine the effects of climate change on forest resilience.

As discussed above, in mixed-severity fire regimes, higher-severity fire occurs as patches in a mosaic of fire effects (Williams and Baker, 2012a; Baker, 2014). In conifer forests of North America, higher-severity fire patches create a habitat type, known as complex early seral forest (DellaSala et al., 2014), that supports levels of native biodiversity, species richness, and wildlife abundance that are generally comparable to, or even higher than, those in unburned old forest (Raphael et al., 1987; Hutto, 1995; Schieck and Song, 2006; Haney et al., 2008; Donato et al., 2009; Burnett et al., 2010; Malison and Baxter, 2010; Sestrich et al., 2011; Swanson et al., 2011; DellaSala et al., 2014). Many rare, imperiled, and declining wildlife species depend on this habitat (Hutto, 1995, 2008; Kotliar et al., 2002; Conway and Kirkpatrick, 2007; Hanson and North, 2008; Bond et al., 2009; Buchalski et al., 2013; Hanson, 2013, 2014; Rota, 2013; Siegel et al., 2013; DellaSala et al., 2014; Baker, 2015; see also Chapters 2–6). The scientific literature reveals the naturalness and ecological importance of multiple age classes and successional stages following higher-severity fire, as well as the common and typical occurrence of natural forest regeneration after such fire (Shatford et al., 2007; Donato et al., 2009; Crotteau et al., 2013; Cocking et al., 2014; Odion et al., 2014). These and other studies suggest that mixed-severity fire, including higher-severity fire patches, is part of the intrinsic ecology of these forests and has been shaping fire-dependent biodiversity and diverse landscapes for millennia (Figure 1.2).

1.3 MIXED- AND HIGH-SEVERITY FIRES HAVE NOT INCREASED IN FREQUENCY AS ASSUMED

Fire history studies show that for many montane forests, including mixed-conifer and ponderosa pine forests, fire frequencies in most forested regions were substantially less during the 20th century (and the early 21st century) compared with the previous few centuries (e.g., Odion et al., 2014). Nonetheless,

Chapter 4

Mammals and Mixed- and High-severity Fire

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4.1 INTRODUCTION

Mammals are ecologically and economically important members of the landscapes in which they live. Large herbivores like deer (*Odocoileus* spp.) and elk (*Cervus elaphus*), and predators like bears (*Ursus* spp.) and wolves (*Canis lupus*), are highly conspicuous and well-known "flagship" mammal species, whereas rodents, bats, and mustelids are cryptic but no less important in their ecosystems. Many species have developed broad ecological tolerance from exposure to environmental variation and natural disturbances over long time periods (Lawler, 2003). However, widespread hunting and excessive habitat fragmentation of landscapes by modern-day humans are qualitatively and quantitatively different from the natural disturbances to which these mammals were exposed in the past (Spies and Turner, 1999), and they have resulted in contraction of historical ranges and population declines. In North America alone notable population declines include elk, grizzly bears (*Ursus arctos*), gray wolves, Canadian lynx (*Lynx canadensis*), bighorn sheep (*Ovis canadensis*), beaver (*Castor canadensis*), the larger species of forest mustelids, and several herteromyid rodents.

Mixed- and high-severity wildfire is a natural disturbance in many vegetation systems of North America, the Mediterranean, Australia, and Africa (see Chapters 1, 2, and 8). The effects of severe fire on organisms vary spatially and temporally, by habitat type, and by species, but how do these disturbances specifically impact mammals? As with any natural disturbance, some species are adversely affected ("fire-averse" species), others benefit ("fire-loving" or pyrophilous species), and still others have a neutral response to fires.

The dynamics of populations and communities of mammals after severe fire depend on factors such as the degree of ecological change, time since fire, size and spatial configuration of burned and unburned areas, extent of edge, isolation of habitat patches by urbanization and roads, and invasion of nonnative species (Smith, 2000; Shaffer and Laudenslayer, 2006; Arthur et al., 2012; Diffendorfer

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FIGURE 4.3 Representative foraging location based upon global positioning system coordinates for a confirmed female Pacific fisher scat detection site several hundred meters into the interior of the largest high-severity fire patch (>5000 ha) in the McNally Fire of 2002, Sequoia National Forest, California. *(Photo by Chad Hanson (2014).)*

2 years after fire in interior chaparral, Madrean evergreen woodland, and ponderosa pine forest in Arizona (Cunningham et al., 2006).

Striped skunk (*Mephitis mephitis*), ringtail (*Bassariscus astutus*), and raccoon (*Procyon lotor*) were photocaptured only in mixed-conifer forests in southern California burned by high-intensity fire, but each were photographed only once (Borchert, 2012). Bobcat (*Lynx rufus*) were photocaptured in similar numbers in severely burned and unburned forest, but captures in the burned area decreased over time over the 4 years of the study. Finally, mountain lion (*Puma concolor*) were photocaptured more often in severely burned forest, but the overall sample was small (four lion in burned areas, one lion in unburned areas).

Bears

Although grizzly bears are flexible in the habitats they use, in British Columbia, Canada, radio-collared grizzly bears strongly selected open forest burned by wildfires 50-70 years earlier at high elevations because these sites supported prolific huckleberries (McLellan and Hovey, 2001) (see Box 4.3). Wildfire also promotes the regeneration of whitebark pine (*Pinus albicaulis*) seeds, another important food source for bears (Kunkel, 2003). Wildfire is not equivalent to logging, as regenerating timber harvests were rarely used by bears in any season (McLellan and Hovey, 2001).

One study compared the demographics and physiology of black bears (*Ursus americanus*) occupying burns of two ages, 13 and 35 years old, in spruce

Chapter 5

Stream-Riparian Ecosystems and Mixed- and High-Severity Fire

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5.1 DEFINING WILDFIRE SEVERITY AND STREAM-RIPARIAN BIOTIC RESPONSES

Wildfire is an important natural disturbance that has consequences for both structural and functional characteristics of riparian and stream ecosystems (Resh et al., 1988; Gresswell, 1999; Verkaik et al., 2013a). More than 20 years of studies now point to a diverse array of responses by stream-riparian organisms and ecosystems to wildfire. Ecological responses vary along gradients of fire characteristics, including severity, extent, frequency, time since disturbance, and hydrological context (Agee, 1993; Arkle et al., 2010; Romme et al., 2011), among others. Although high-severity fire can result in major changes to stream and riparian areas, including erosion and sedimentation, opening of the riparian canopy, inputs of large wood to the stream channel, and changes in water temperature and chemistry, low-severity fire may have little to no effect (Jackson and Sullivan, 2009; Arkle and Pilliod, 2010; Malison and Baxter, 2010a; Jackson et al., 2012) (Figure 5.1). Stream-riparian biota respond both directly to wildfire as well as indirectly via wildfire-induced changes in physical habitat (Arkle et al., 2010). Land managers often work to keep high-severity fire out of riparian zones using a suite of techniques, including fuel reduction (removal of trees and understory vegetation through mechanical thinning and/or prescribed fire) and suppression (Stone et al., 2010). However, stream and riparian organisms often are highly adapted to disturbances, including floods, drought, and wildfire (Dwire and Kauffman, 2003; Naiman et al., 2005), and dynamic fire regimes that operate over time and space may be important in maintaining the integrity and biodiversity of linked stream-riparian ecosystems (Bisson et al., 2003).

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well as resource management uncertainty and public dialogue, has been centered on the costs and benefits of wildfire (Pyne, 1997, 2004; Hutto, 2008). Moreover, we afford particular attention to fire-food web dynamics because food webs are a valuable window into the structure, function, and productivity of linked stream-riparian ecosystems (Wallace et al., 1997; Power and Dietrich, 2002; Baxter et al., 2005) and can provide spatially and temporally integrated perspectives on the effects of wildfire (e.g., Mihuc and Minshall, 2005). We conclude with a broad discussion of the potential importance of high-severity wildfire for biodiversity, conservation, and management of stream-riparian ecosystems.

Importance of Stream-Riparian Ecosystems

Even though aquatic ecosystems make up only about 2% of terrestrial landscapes, they are disproportionately relied on by humans for numerous natural resources (Postel and Carpenter, 1997). Streams and riparian areas act as conduits, reservoirs, and purification systems for fresh water (Sweeney et al., 2004). Riparian zones sustain unique communities of organisms, contributing >50%, on average, to regional species richness values (Sabo et al., 2005), and a disproportionate number of threatened and endangered species rely on aquatic and riparian habitats (Carrier and Czech, 1996), as do many organisms that provide food, medicine, and fiber to humans. In addition, these areas are valued as scenic and used for recreation.

The influence of wildfire as an agent of natural selection has resulted in a suite of organisms that exhibit apparent adaptations that make them resistant or resilient to wildfire, and riparian and aquatic organisms are no exception. Because riparian zones are transitional areas (or ecotones) between aquatic and terrestrial habitats, a diverse array of animals are associated with riparian corridors, ranging from aquatic (fish, benthic invertebrates) to amphibious (frogs, salamanders) to terrestrial (riparian birds, mammals, and reptiles), each exhibiting responses to wildfire that vary across gradients of fire severity (Box 5.1, Figure 5.2).

Despite their importance, riparian areas have been degraded worldwide, and in some regions the majority of riparian zones have been lost altogether. For

BOX 5.1

Examples of stream-riparian animals that may benefit from high-severity wildfire

(1) Immediate impacts may be negative, but stream invertebrate abundance and biomass frequently increase in the short to midterm following fire (Minshall, 2003; Verkaik et al., 2013a), and the production of emerging adult insects (i.e., aquatic insects that emerge from the water as winged adults) can increase as well (Mellon et al., 2008; Malison and Baxter, 2010a). Such increases may be accompanied by reductions in species diversity and dominance by insects that are habitat and trophic generalists, are drift-dispersers, and have multivoltine (having multiple generations per year) life cycles (e.g., Chironomidae, Baetidae) (Mihuc and Minshall, 1995; Minshall et al., 2001b). Climate and hydrologic context following wildfire may mediate mid- to longer-term impacts, however; for instance, Rugenski and Minshall (2014) reported increases in both invertebrate biomass and diversity in wilderness streams of Idaho more than 5 years following severe wildfire during a period of time characterized by reduced peaks in spring floods.

(2) Despite a long-standing assumption that high-severity wildfire has negative effects on stream fishes, in many cases immediate effects on fishes seem slight or recovery of populations occurs rapidly (Rieman et al., 1997; Sestrich et al., 2011), and there is mounting evidence of numerous indirect, positive effects on fish populations that may follow severe wildfire. For instance, the pulse in invertebrate production that can follow severe wildfire (Malison and Baxter, 2010a; also see Chapter 6) may provide increased food resources to fish. Even when wildfire is followed by scouring debris flows that may, at least temporarily, extirpate fish from a local stream reach (Howell, 2006), the combination of increased downstream transport of sediment and large wood that creates and maintains essential habitat (Bigelow et al., 2007), and increased export of drifting invertebrate prey from such tributaries (Harris et al. In press), may lead to net positive effects on fishes in recipient habitats. The pulse of natural erosion/sedimentation that can occur soon after high-severity fire can be associated with increases in native fish populations by ≥3 years after fire (Sestrich et al., 2011), possibly partly a result of enhanced spawning grounds.

(3) Streams and their adjacent riparian zones provide important foraging habitat for insectivorous bats (Seidman and Zabel, 2001; Russo and Jones, 2003; Fukui et al., 2006), where aquatic insects that emerge from streams as adults can comprise the majority of bat diets (Belwood and Fenton, 1976; Swift et al., 1985). The combination of increased emergence of stream insects and removal of the riparian canopy following high-severity fire may provide bats with better foraging conditions (Malison and Baxter, 2010b; Buchalski et al., 2013) (see Box 5.2 for additional details and Chapter 4 for a similar discussion of bat use of burned areas).

(4) Many birds that principally occupy riparian areas also rely on trees burned by fire (i.e., snags) for nesting cavities. For example, in the western United States, Lewis's woodpeckers (*Melanerpes lewis*), a cavity-nester and an aerial insectivore common in riparian zones, have been called "burn specialists" because they tend to be abundant in both recent (2-4 years after fire) and older (10-25 years after fire) high-severity burns (Linder and Anderson, 1998; Vierling and Saab, 2004). Lewis's woodpeckers and other aerial insectivorous birds can also benefit from increases in emergent insects and other aerial insect prey (e.g., Bagne and Purcell, 2011) following high-severity fires (see Chapter 3).

(Stone et al., 2010), but because prescribed fires typically differ from wildfires in severity, timing, frequency, and extent (McIver et al., 2013), their influence on riparian and aquatic systems remains an open question (Boerner et al., 2008; Arkle and Pilliod, 2010). In addition, methods used during fire suppression efforts can have negative effects on stream-riparian ecosystems. For example, the use of fire retardants around aquatic systems has led to the mortality of aquatic organisms (Gaikowski et al., 1996; Buhl and Hamilton, 2000; Gimenez et al., 2004) and is therefore banned by firefighting agencies, but construction of fire lines within drainages continues. In some cases fire lines can facilitate the introduction of invasive species and be a significant source of chronic sediment delivery to streams following wildfires (reviewed by Beschta et al., 2004 and Karr et al., 2004).

Postfire management has the potential to be more disruptive to stream-riparian ecosystems and have longer-lasting consequences than high-severity wildfire itself (Beschta et al., 2004; Karr et al., 2004); therefore, any postfire management that does not mitigate the effects of suppression activities should be avoided, including planting with nonnative seeds, construction of debris dams, and postfire logging. Debris dams often are insufficient at ameliorating soil erosion and end up in stream channels following storms, where they impede the movement of organisms and disrupt flow. Mechanical disruption of soils, which often occurs as a result of postfire logging, increases chronic erosion and the deposition of fine sediments (McIver and Starr, 2001; McIver and McNeil, 2006), and soil compaction in forests can persist for 50-80 years (Quigley and Arbelbide, 1997), which may exceed the duration of effects from high-severity wildfire. Even dead vegetation provides soil stability; snags are important habitat for riparian organisms, and large wood is a significant and ecologically important structural element of stream-riparian ecosystems (Gregory et al., 2003). Thus postfire logging may reduce the quality of stream-riparian habitat in multiple ways. Whereas postfire management should be used with caution, prefire restoration of stream-riparian ecosystems might reduce potential negative effects of severe wildfire (Beschta et al., 2004); such efforts might include surfacing, stabilizing, and removing legacy roads; discouraging grazing in riparian zones; and restoring fluvial connectivity.

Finally, as we have described in this chapter, the effects of wildfire on stream-riparian ecosystems operate over gradients of severity, space, and time and across levels of ecological organization. For example, although there are likely to be winners and losers at the individual and population levels in the short term and over relatively small spatial scales, community- and ecosystem-level responses seem to be more neutral or positive, are longer lived, and tend to operate at relatively larger spatial scales. Therefore, management of stream-riparian ecosystems in landscapes that experience high-severity fire will benefit from a holistic perspective that takes into account heterogeneous responses over space and time.

Chapter 13

Flight of the Phoenix: Coexisting with Mixed-Severity Fires

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13.1 ECOLOGICAL PERSPECTIVES ON MIXED-SEVERITY FIRE

Throughout this book, we have presented compelling evidence of fire's beneficial ecological role mainly in western North America but with relevant case studies in other regions. Even though most people recognize the importance of maintaining fire on the landscape, few realize the myriad ecosystem benefits associated with large fires of mixed severity. Habitat heterogeneity, which may be maximized by mixed-severity fire that includes large patches of high severity, and the successional mosaic such fire creates, is one of the most dependable predictors of species diversity (Odion and Sarr 2007, Sitters et al., 2014). This ecological tenet has yet to be fully realized in management circles. If such fires are operating within historical bounds, then ecosystems will remain resilient to them; indeed, deficits of these fires relative to the natural range of variability, in places such as montane forests of western North America, are degrading to fire-dependent biodiversity (Odion et al., 2014a; Sherriff et al., 2014). This is particularly the case when reductions in fire extent and/or severity occur in combination with forest management practices, such as post-fire logging, that undermine development of complex early seral forests (Chapter 11).

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evidence that house-to-house ignitions by airborne firebrands were responsible for many of the destroyed homes."

Investments in making homes and communities fire safe are clearly fiscally prudent and responsible homeownership that can save lives and homes by reducing risks to all, especially firefighters. Moreover, proper land use zoning that reduces housing development in firesheds is key to the survival of home structures over the larger area (Syphard et al., 2014).

In sum, these recent studies show that overcoming misperceptions about homeowner losses is urgently needed because those misconceptions are a driving factor in many inappropriate fuel reduction projects in wild areas. We hypothesize that with stepped-up planning directed at proper homeowner safety (as demonstrated in the above studies), public attitudes about large and intense fires may begin to shift from fear-based primal responses to more of a neocortex-like awareness of fire as nature's phoenix. This could be tested using before-and-after polling about large, higher-severity fires with and without proper public safety measures in places.

13.4 TO THIN OR NOT TO THIN?

One of the most significant challenges involved in changing the way land managers think about fire in the forests is how the US Forest Service views forest fires. The agency is deeply invested in continuing the fire management trajectory of the past—a situation compounded by the budgetary issues associated with the agency's direction of much, and often most, of their tax-based support to selling timber from public lands, and the agency's retention of most of the revenue from such timber sales to fund staff salaries and operations. Though in recent years we have learned much about the ecological benefits of higher-severity fire and the risks to fire-dependent wildlife species from further suppressing these fires, which are deficient in most western US conifer forests (Chapters 1–5), the Forest Service continues to aggressively promote landscape-level mechanical thinning (North, 2012; Stine et al., 2014) and postfire logging (Collins and Roller, 2013) ostensibly to reduce fuels and prevent and mitigate future fire. These forest management policies are promoted based on the assumption that decades of fire suppression have created forests "overloaded with fuel, priming them for unusually severe and extensive wildfires" (Stine et al., 2014; see also North, 2012). The basic concept being articulated by the Forest Service is that, because of decades of fire suppression and "fuel accumulations," we cannot simply allow wildland fires to burn because long-unburned forests will "uncharacteristically" burn almost exclusively at higher severities (North, 2012; Stine et al., 2014). Under this premise, recommendations focus on how to manage forests through logging and fire suppression to further reduce and prevent the significant occurrence of mixed-severity fire (North et al., 2009; North, 2012; Stine et al., 2014). Yet these sources do not include a discussion of the current deficit of these fires in most forests of western North America (Odion et al. 2014a; see also Chapters 1, 2, and 9) or meaningful content on the ecological importance of mixed-severity fire for many rare and imperiled wildlife species (Chapters 2–5). Nor do they explore the validity of the basic premise that long-unburned forests will burn much more severely.

Studies that empirically investigated the "time-since-fire" issue in the Sierra region of northern California and the Klamath Mountains of Oregon and California tended to find that, contrary to popular assumptions, the most long-unburned forests experience mostly low- and moderate-severity fire and do not have significantly higher levels of higher-severity fire than more recently burned forests (Odion et al., 2004, 2010; Odion and Hanson, 2006, 2008; Miller et al., 2012; van Wagtendonk et al., 2012). One modeling study predicted a modest increase in fire severity with increasing time since fire, but the strength of inference was limited by a lack of data for all but long-unburned stands, especially in the largest forest types, such as mixed-conifer forest. Even the most long-unburned forests were predicted to have ~70-80% low/moderate-severity effects (Steel et al., 2015), well within the range of natural variability (see Chapter 1). In fact, long-unburned forests sometimes have the lowest levels of higher-severity fire; understory vegetation and the lower limbs of conifers self-thin as canopy cover increases and available sunlight in the understory decreases with increasing time since fire (Odion et al., 2010). Therefore the argument that we cannot allow more wildland fires to burn without suppression in natural areas is not valid for many dry montane forests in western North America (Odion et al., 2010).

Problems with Fuel Models and Fire Liabilities

Government programs that aim to make forests safe places for people to live are based on theory rather than actual evidence about historical forests. As discussed above, the common argument has been that fuels have unnaturally accumulated from fire exclusion and land uses, and if fuels are restored to low levels, fires will burn primarily at low intensity rather than as high-intensity crown fires (e.g., Agee and Skinner, 2005). Thus forests can be restored while also making them safe places to live—a win-win solution that is appealing to the public. Little evidence about actual historical fuel amounts in forests to support this argument was available, however; instead, evidence is mostly based on the idea that frequent fires would have kept fuels at low levels. When records from land surveys before fire exclusion were examined (Baker, 2012, 2014; Baker and Williams, 2015; Hanson and Odion, in press), understory fuels (shrubs, small trees) that would naturally have promoted intense fires were found to have been common and often abundant in many areas, and small trees were dominant, not rare. This direct evidence suggests that fuel treatments would typically have to artificially remove natural shrubs and small trees and adversely alter habitat for native species in a quest to make forests safer places for people to live.

Fuel reduction also has been overpromised to be effective, using questionable logic and unvalidated models. First, fire intensity in most forest types is much more strongly affected by wind than by fuel. High fire-line intensity, the primary fire characteristic that promotes crown fires, is the product of the energy released by burning fuel and the rate of spread of fire (Alexander, 1982). Energy release by fuel varies over perhaps a 10-fold range, however, whereas rate of spread can vary over more than a 100-fold range; thus a high rate of spread caused by strong winds can easily overcome the limited reductions in fuel that are feasible (Baker, 2009). This was confirmed by a recent analysis of the 2013 Rim Fire in California, which concludes: "Our results suggest that even in forests with a restored fire regime, wildfires can produce large-scale, high-severity fire effects under the type of weather conditions that often prevail when wildfire escapes initial suppression efforts. . . . During the period when the Rim fire had heightened plume activity... no low severity was observed [in thinned areas], regardless of fuel load, forest type, or topographic position" (Lydersen et al., 2014, p. 333). Second, common fire models used to show that forests would be fire-safe after fuel reductions have an underprediction bias and are not validated. These flawed models include NEXUS, FlamMap, FARSITE, FFE-FVS, FMAPlus, and BehavePlus (Cruz and Alexander, 2010; Alexander and Cruz, 2013; Cruz et al., 2014). The underprediction bias means that these models often predict that fuel reductions would reduce or eliminate the potential for crown fires in forests, when in fact fuel reductions do not achieve this effect. Fixing these models would be difficult and has not yet occurred (Alexander and Cruz, 2013). Also, these models have not been sufficiently tested and validated using a suite of actual fires, in which case they would likely be shown to fail (Cruz and Alexander, 2010). Alternative validated models are available and could be further developed, but they are not being used (Cruz and Alexander, 2010). Further, studies of tree mortality in thinned areas following fire do not typically take into account the mortality caused by the logging itself before the fire, leading to further biased results.

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

Finally, another land management liability that is frequently overlooked when assessing fire-related economic losses is the role of silviculture. For instance, before the 2013 Rim Fire, a significant portion of the Stanislaus National Forest in central California's Sierra Nevada Mountains consisted of even-aged monoculture tree plantations (following past clearcuts) distributed across large landscapes (Figure 13.3). Land managers often claim that clearcutting over large landscapes

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Forest fires and the snowshoe hare-Canada lynx cycle

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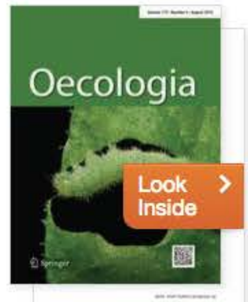

Summary

This paper shows that there is a reasonable coincidence between the Canada lynx cycle and the occurrence of forest and brush fires. Fires set in motion plant succession, potentially leading to an increase in snowshoe hares (Grange, 1965). Snowfall is also correlated with the lynx cycle and tends to account for the variation not accounted for by fires. I conclude that Grange's (1949, 1965) hypothesis that fire and plant succession drive the snowshoe hare cycle deserves serious consideration, as do Butler's (1962) and Watt's (1968, 1973) suggestions that precipitation may drive or decisively modify furbearer 'cycles'. On this evidence, the snowshoe hare-Canada lynx cycle seems likely to be a forced oscillation rather than a predator-prey, parasitehost, or herbivore-vegetation limit cycle, as proposed by several authors. Fire clearly appear to be a periodic phenomenon, not-withstanding the difficulty of showing periodicity in simple weather time series. Fire is itself a meteorological phenomenon, although complexly related to simple weather variables.

Supplementary Material (0)

References (121)

About this Article

Title	Topics	Authors
Forest fires and the snowshoe hare-Canada lynx cycle	» Ecology » Plant Sciences	John F. Fox (1)
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Habitat preference of Canada lynx through a cycle in snowshoe hare abundance

Garth Mowat and Brian Slough

Abstract: We assessed habitat preference of a lynx (*Lynx canadensis*) population through 8 years of a snowshoe hare (*Lepus americanus*) cycle. Seventy-four percent of our southern Yukon study area was approximately 30-year-old regenerating forest resulting from a large wildfire. The study area was not trapped and lynx density was very high compared with other populations in North America. Contrary to our prediction, there was no discernable shift in habitat preference through the hare cycle; however, our habitat types were coarsely mapped and our radiolocations relatively inaccurate. Lynx may have altered their habitat preferences at finer scales (for patches <2 ha). Lynx showed strong preference for regenerating habitats over mature white spruce (*Picea glauca*) and alpine–subalpine. Lodgepole pine (*Pinus contorta*) dominated regenerating stands were preferred over spruce–willow (*Salix* spp.) stands of equal age. Riparian willow stands were also preferred over mature spruce forest and alpine. Lynx used riparian willow stands more in winter, but we detected no other shifts in habitat preference between snow-free and winter periods. We did not detect any difference in habitat preference between sexes. Independent juveniles made greater use of mature forest and perhaps riparian willow than adults, but no other difference in preference between the two age groups was noted. Lynx preference for regenerating habitat over mature forest suggests that burns will benefit lynx, especially if the regenerating community is pine dominated. Logging will only likely provide similar benefits if a dense pine understory results, which is unlikely in intensively managed stands. The suppression of forest fires in recent decades may have contributed to the decline of lynx numbers in the south of their range.

Résumé : Nous avons déterminé le choix d'habitat chez une population de lynx (*Lynx canadensis*) au cours des 8 années d'un cycle d'abondance du lièvre d'Amérique (*Lepus americanus*). Soixante-quatorze pour cent de la région d'étude dans le sud du Yukon est couverte d'une forêt en régénération à la suite d'un important feu de forêt il y a une trentaine d'années. Il n'y a pas eu de trappage dans la région et la densité des lynx y est particulièrement élevée, par comparaison aux autres populations de l'Amérique du Nord. Contrairement à nos prédictions, il ne s'est pas produit de changement apparent dans les choix d'habitat au cours du cycle d'abondance des lièvres; cependant, notre cartographie des types d'habitats était grossière et nos déterminations par radio des emplacements des animaux relativement imprécises. Il se peut que les lynx aient modifié leur choix d'habitat à des échelles plus fines (parcelles de <2 ha). Les lynx préfèrent de beaucoup les habitats en régénération aux pessières matures (*Picea glauca*) et aux zones subalpines et alpines. Ils choisissent les parcelles en régénération dominées par le pin vrillé (*Pinus contorta*) de préférence aux parcelles de même âge dominées par l'épinette et les saules (*Salix* spp.). Ils préfèrent aussi les parcelles riveraines de saules à la pessière mature et à la zone alpine. Les lynx utilisent plus les zones riveraines de saules en hiver et c'est le seul changement d'habitat observé entre la période hivernale et la période sans neige. Il n'y a aucune différence de choix d'habitat entre les sexes. Les jeunes indépendants utilisent davantage les forêts matures et peut-être aussi les saulaies que les adultes et c'est la seule différence observée entre les choix d'habitat chez ces deux groupes d'âge. La préférence des lynx pour les habitats en régénération plutôt que pour la forêt mature laisse croire que les feux de forêt sont bénéfiques aux lynx, particulièrement lorsque la communauté en régénération est dominée par les pins. La coupe forestière ne fournit les mêmes avantages que si un sous-bois dense de pins se développe, ce qui est improbable dans les forêts fortement aménagées. La suppression des feux de forêt au cours des dernières décennies peut avoir contribué au déclin des densités de lynx dans la partie australe de leur répartition géographique.

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Open Letter to U.S. Senators and President Obama from Scientists Concerned about Post-fire Logging and Clearcutting on National Forests

As professional scientists with backgrounds in ecological sciences and natural resources management, we are greatly concerned that legislation which passed the House in July 2015, H.R. 2647, would suspend federal environmental protections to expedite logging of both post-fire wildlife habitat and unburned old forests on national forest lands. This legislation would also effectively eliminate most analysis of adverse environmental impacts, and prevent enforcement of environmental laws by the courts.

A similar measure, S. 1691, currently proposed in the U.S. Senate, would override federal environmental laws to dramatically increase post-fire logging, increase logging and clearcutting of mature forests, eliminate analysis of environmental impacts for most logging projects, and effectively preclude enforcement of environmental laws. The bills propose these measures under the guise of "ecosystem restoration," ostensibly to protect national forests from fire.

Not only do these legislative proposals misrepresent scientific evidence on the importance of post-fire wildlife habitat and mature forests to the nation, they also ignore the current state of scientific knowledge about how such practices would degrade the ecological integrity of forest ecosystems on federal lands. We urge you to vote against this legislation, and urge President Obama to veto these bills if they are passed in some form by Congress.

National Forests were established for the public good and include most of the nation's remaining examples of intact forests. Our national forests are a wellspring of clean water for millions of Americans, a legacy for wildlife, sequester vast quantities of carbon important in climate change mitigation, and provide recreation and economic opportunities to rural communities if responsibly managed. Though it may seem at first glance that a post-fire landscape is a catastrophe, numerous scientific studies tell us that even in the patches where forest fires burn most intensely, the resulting wildlife habitats are among the most ecologically diverse on western forestlands and are essential to support the full richness of forest biodiversity.[1]

Post-fire conditions also serve as a refuge for rare and imperiled wildlife species that depend upon the unique habitat features created by intense fire. These include an abundance of standing dead trees, or "snags," which provide nesting and foraging habitat for woodpeckers and many other plant and wildlife species responsible for the rejuvenation of a forest after fire.

The post-fire environment is rich in patches of native flowering shrubs that replenish soil nitrogen and attract a diverse bounty of beneficial insects that aid in pollination after fire. Small mammals find excellent habitat in the shrubs and downed logs, providing food for foraging spotted owls. Deer and elk browse on post-fire shrubs and natural conifer regeneration. Bears eat and disperse berries and conifer seeds often found in substantial quantities after intense fire, and morel mushrooms, prized by many Americans, spring from ashes in the most severely burned forest patches.

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[1] See http://store.elsevier.com/The-Ecological-Importance-of-Mixed-Severity-Fires/Dominick-DellaSala/isbn-9780128027493/.

This post-fire renewal, known as "complex early seral forest," or "snag forest," is quite simply some of the best wildlife habitat in forests, and is an essential stage of natural processes that eventually become old-growth forests over time. This unique habitat is not mimicked by clearcutting, as the legislation incorrectly suggests. Moreover, it is the least protected of all forest habitat types, and is often as rare, or rarer, than old-growth forest, due to extensive fire suppression and damaging forest management practices such as those encouraged by this legislation. Much of the current scientific information on the ecological importance of post-fire habitat can be found in several excellent videos, including ways for the public to co-exist with fires burning safely in the backcountry.[1,2]

After a fire, the new forest is particularly vulnerable to logging disturbances that can set back the forest renewal process for decades. Post-fire logging has been shown to eliminate habitat for many bird species that depend on snags, compact soils, remove biological legacies (snags and downed logs) that are essential in supporting new forest growth, and spread invasive species that outcompete native vegetation and, in some cases, increase the flammability of the new forest.

While it is often claimed that such logging is needed to restore conifer growth and lower fuel hazards after a fire, many studies have shown that logging tractors often kill most conifer seedlings and other important re-establishing vegetation and actually increases flammable logging slash left on site. Increased chronic sedimentation to streams due to the extensive road network and runoff from logging on steep slopes degrades aquatic organisms and water quality.[3]

We urge you to consider what the science is telling us: that post-fire habitats created by fire, including patches of severe fire, are ecological treasures rather than ecological catastrophes, and that post-fire logging does far more harm than good to public forests. We urge Senators to vote against any legislation that weakens or overrides environmental laws to increase post-fire logging or clearcutting of mature forest as degrading to the nation's forest legacy. And, we urge President Obama to veto any such legislation that reaches his desk as inconsistent with science-based forest and climate change planning.

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[2]http://www.fs.usda.gov/detail/r5/news-events/audiovisual/?cid=stelprdb5431394;
https://vimeo.com/75533376; http://vimeo.com/groups/future/videos/8627070;
http://www.youtube.com/watch?v=iTl-naywNyY&list=PL7F70F134E853F520&index=15;
http://www.youtube.com/watch?v=1BmTq8vGAVo&feature=youtu.be; http://vimeo.com/3428311

[3]Hutto, R. L. 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conservation Biology 20:984-993. Beschta, R.L. et al. 2004. Postfire management on forested public lands of the western USA. Conservation Biology 18:957-967. Lindenmayer, D.B. et al. 2004. Salvage-harvesting policies after natural disturbance. Science 303:1303. Karr, J. et al. 2004. The effects of postfire salvage logging on aquatic ecosystems in the American West. Bioscience 54:1029-1033. DellaSala, D.A., et al. 2006. Post-fire logging debate ignores many issues. Science 314-51-52. Donato, D.C. et al. 2006. Post-wildfire logging hinders regeneration and increases fire risk. Science 311 No. 5759:352.

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FOREST RESILIENCE, BIODIVERSITY, AND CLIMATE CHANGE

A Synthesis of the Biodiversity/Resilience/Stability Relationship in Forest Ecosystems

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FOREST RESILIENCE, BIODIVERSITY, AND CLIMATE CHANGE

A Synthesis of the Biodiversity/Resilience/Stability Relationship in Forest Ecosystems

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that respond autonomously to internal and external drivers. For example, as available water becomes limiting, the height and density of the tree canopies is reduced because of basic ecophysiological relationships governing environmental controls on plant growth (Berry and Roderick 2002). If climate change results in a significant reduction in water availability, then the forest system will naturally change species composition (or state – see definition below). For example, the vegetation will reach a threshold beyond which the vegetation structure is not sufficiently tall and dense to comprise a forest, along with the concomitant changes in the dominant taxonomic composition of the plant community (Stephenson 1990). Under severe drying conditions, forests may be replaced by savannahs or grasslands (or even desert), while under increased temperature, open taiga can be replaced by closed boreal forests (assuming that there is sufficient moisture to support plant growth during the newly extended growing season) (e.g., Price and Scott 2006, Kellomaki et al. 2008).

Forests can also influence regional climates, depending on their extent and this is particularly true of the Amazon forest (Betts et al. 2008, Phillips et al. 2009). Hence, numerous feedbacks exist between climate and forests as the climate changes (Bonan et al. 2003, Callaghan et al. 2004, Euskirchen et al. 2009). These feedbacks are mediated through changes to albedo (Euskirchen et al. 2009), altered carbon cycle dynamics (Heath et al. 2005, Phillips et al. 2009), energy fluxes and moisture exchange (Wildson and Agnew 1992, Bonan et al. 2003), and herbivory resulting in increased fires (Ayres and Lomardero 2000). Hence, maintaining forest resilience can be an important mechanism to mitigate and adapt to climate change.

1.2 Definitions of and related to resilience

We discuss several closely related terms throughout this paper and define them here, including resilience, resistance, state, and stability. We define **resilience** as the capacity of an ecosystem (i.e., forest type, in this paper) to return to the original state following a perturbation, maintaining its essential characteristic taxonomic composition, structures, ecosystem functions, and process rates (Holling 1973). Similarly, Walker and Salt (2006) defined resilience as the capacity of a system to absorb disturbance and still retain its basic function and structure, and therefore its identity (i.e., recognizable as the same by humans).

A forest ecosystem can respond in different ways to disturbances and perturbations. Depending on the capacity of forests to cope with the degree of change, the characteristic taxonomic composition, vegetation structure, and rates of ecosystem processes may or may not be altered; that is, the resilience of the forest ecosystem may or may not be overcome. Forest characteristics can be used individually or in combination to define a forest ecosystem **state**. Most commonly, a forest state is considered in terms of the dominant assemblage of tree species forming an ecosystem at a location, along with the functional roles those species play, and the characteristic vegetation structures (height, layers, stems density, etc.) at maturity. So, a given mature forest type has a particular suite of characteristics that identify its state. (Note that we use the terms 'system' and 'ecosystem' synonymously throughout.)

A difference has been made in the scientific literature between "engineering resilience" and "ecological resilience" (Holling 1973, Peterson et al. 1998, Gunderson 2000, Walker et al. 2004). Engineering resilience is related to the capacity of a system to return to its more-or-less exact pre-disturbance state, and the assumption is that there is only one steady state. The latter concept has also been more recently referred to as equilibrium dynamics. Ecological resilience is defined as the ability of a system to absorb impacts before a threshold is reached where the system changes into a different state altogether. For example, in the case of increasing climatic drought, a resilient forest ecosystem according to the "engineering" definition is one that would recover from drought stress, with little or no change in species composition. If the ecological definition is used, then it is acknowledged that more than one stable system state is possible, with resilience being the measure of a forest ecosystem's capacity to withstand a prolonged drought before being converted into a different vegetation ecosystem (e.g., non-forest); though it might go through several other different but stable forest states with new species compositions, before the conversion to grassland. Many of those successive forest states might be able to provide most or all of the goods and services provided by the initial state, and all would be recognizable as a forest type. This is also referred to as non-equilibrium dynamics.

Forests are engineering resilient in the sense that they may recover, after a period of time, from a catastrophic disturbance to their original, pre-disturbance state maintaining, more-or-less, the original species composition. The main ecosystem states of interest are defined by the dominant floristic (tree) composition and stand structure. However, it is also useful to consider the question of ecological resilience with respect to the capacity of a forest to continue to provide certain (most or all) ecosystem goods and services, even if the forest composition and structure are permanently altered by disturbances.

Resilience is an emergent property of ecosystems that is conferred at multiple scales by genes, species, functional groups of species (see definition below), and processes within the system (Gunderson 2000,

Forest resilience as illustrated by the recovery of mixedwood forest in eastern Canada as a result of red pine plantation on a logged site, with natural infilling by deciduous species over a period of about 50-80 years.

Drever et al. 2006). Maintaining or restoring forest resilience is often cited as a necessary societal adaptation to climate change (e.g., Millar et al. 2007, Chapin et al. 2007). Drever et al. (2006) noted the importance of clarifying the questions: resilience of what and resilience to what? Here, the "of what" are particular characteristics of forest ecosystems (e.g., carbon sequestration, water use/yield), and the "to what" are environmental and human-caused disturbances, especially climate change. For example, an individual species' physiological tolerances may be exceeded by natural environmental change or human-caused events. Consequently, the species composition of a forest may change while other ecosystem characteristics persist.

Forests are generally **resistant** to change, that is, they change little within bounds as a result of non-catastrophic disturbances, such as chronic endemic insect herbivory or minor blowdown and canopy gaps created by the death of individual or small groups of trees. Forests may also be resistant to certain environmental changes, such as weather patterns over time, owing to redundancy at various levels among functional species (as discussed further below, redundancy refers to the overlap and duplication in ecological functions performed by the diversity of genomes and species in an ecosystem). Ecosystems may be highly resilient but have low resistance to a given perturbation. For example, grasslands are not resistant, but are highly resilient, to fire. However, most well-developed forests, especially primary old forests, are both resilient and resistant to changes (e.g., Holling 1973, Drever et al. 2006).

Resistance is related to the concept of **stability** in the sense that, in response to minor perturbations, a forest ecosystem returns to within a range of variation around a specified ecosystem state. Stability reflects the capacity of an ecosystem to remain more or less in the same state within bounds, that is, the capacity to maintain a dynamic equilibrium over time while resisting change to a different state. A stable ecosystem **persists** when it has the capacity to absorb disturbances and remain largely unchanged over long periods of time.

Species stability refers to consistent species composition over time. Drever et al. (2006) suggested that forest types that naturally progress through successional compositional changes are not necessarily changing state. On the other hand, a forest that was once dominated by a certain suite of species and that has changed as a result of new environmental conditions or human interference has changed ecosystem diversity, where higher variance is observed (Hooper et al. 1995, Ives et al. 1999, Lehman and Tilman 2000, Hughes et al. 2002).

Loreau et al. (2002) noted the importance of regional species richness that enables migration into systems as a means to enhance ecosystem adaptability to change over time. Immigration could enhance both genotypic and phenotypic responses to environmental change enabling resilience in the system through compensation. Overall, the evidence is consistent with the concept that diversity enhances the stability of ecosystem processes (Hooper et al. 2005) and the flow of goods and services.

Ecosystems may exist in more than one stable state (Holling 1973), a fact supported by some experimental evidence largely from closely controlled experiments (Schroder et al. 2005). Drever et al. (2006) provided several examples of alternate stable states among the forest biomes. It seems intuitive that forest ecosystems have multiple stable states that depend on the kinds of disturbances that forests regularly undergo (Marks and Bormann 1972, Mayer and Rietkerk 2004, Schroder et al. 2005) and that many or all of these alternative states may deliver similar goods and services. For example, regeneration trajectories following wildfire differ in many forest types depending on previous disturbances, intensity of the fire, time since last fire, whether or not a fire occurs in a year with abundant tree seed, level of endemic insect infestation, age of the trees, and many other factors (Payette 1992, Little et al. 1994, Hobbs 2003, Baeza et al. 2007). While the engineering resilience may be low, in that the identical or similar species mix may not result following recovery from the disturbance, the ecological resilience is high because a forest ecosystem is restored. Lack of convergence to pre-disturbance floristic composition does not necessarily imply a lack of resilience with respect to other forest system characteristics. Rather it implies that successional patterns differ depending on circumstances but that the system is ecologically resilient, even though the dominant canopy species composition has changed along with certain ecological processes.

The capacity of an ecosystem to stay within stable bounds is related to slow processes that can move the system to another state, sometimes a state that is undesirable, from a human perspective (Scheffer and Carpenter 2003). Folke et al. (2004) suggested that biodiversity is one of those slow-changing variables that have consequences for ecosystem state, acting primarily through species with strong functional roles. The capacity of systems to maintain stability in the face of environmental change is also related to the capacity of individuals within species to meet challenges and to the possibility that other species may increase their functionality under changed regimes (biodiversity as insurance). A major factor impeding the recovery and stability of forest ecosystems is degradation and loss of functional species and reduced redundancy caused by land use practices, including unsustainable harvesting. Degradation results in the ecosystem moving to an undesirable state that may have its own high resilience but be undesirable in terms of the reduced goods and services that it provides.

3.3.1 Diversity and invasion of ecosystems

Another measure of stability, and ultimately of resilience in the case of forest pests, is the capacity of an ecosystem to resist invasion by non-local species (i.e., community invasibility). Various factors, both extrinsic and intrinsic to an ecosystem, such as availability of niches, system degradation, and fragmentation, may affect the capacity of alien species to invade. Another factor which may promote invasion is the lack of enemies of the invading species in the new range (Williamson 1996). Most experimental evidence of a diversity-stability relationship in ecosystems again comes from highly controlled experiments using grasses, and many studies are the same as those assessing the diversity-production relationship (e.g., Tilman 1996, Levine 2000, Symstad 2000, Kennedy et al. 2002). Loreau et al. (2002) reviewed numerous studies of the relationship between resistance, diversity, and invasibility, and found that most supported a negative relationship, with the majority again in grasslands. Many of these studies have been criticized based on uncontrolled effects (e.g., Huston 1997, and see Loreau et al. 2002, Vila et al. 2005, Fridley et al. 2007 for summaries of critiques). Liao et al. (2008) conducted a meta-analysis of the effects of plant invasion into various ecosystems, including many forest systems. They found profound effects of invasion on the carbon and nitrogen-related processes in all systems, usually positive in terms of carbon sequestration rates with both positive and negative effects for nitrogen. They did not provide information about the levels of past disturbance in the systems, but for these results to have occurred, the invading species apparently occupied vacant niches, possibly made available from past disturbances. Thus evidence relating resistance to invasion success is based on the capacity of species in more diverse systems to better use and/or partition resources, forests will be considerably different than at present.

5.4 Summary among forest biomes

All forest types will undergo some change as a result of altered climate conditions; some of these changes are already occurring but widespread change is expected over the next 50-100 years (e.g., Alcamo et al. 2007, Fischlin et al. 2009). From the case-studies, it is clear that some forests are considerably more vulnerable (less resilient) than others as a result of altered disturbance regimes that are predicted under climate change. This is especially the case for forests where previously rarely-seen disturbances will become more common, such as fire in rainforests. In some cases, even ecological resilience will be overcome and forests are expected to change states to non-forest or savannah (IPCC 2007), as has happened in many areas previously, such as the northern Sahara area of Africa (Kröpelin et al. 2008). In many cases, forests will change states, however, at least among most boreal and some temperate forests, ecological resilience is expected. In many tropical forests, however, many rainforests may become dry tropical forests with reduced carbon storage capacity (case-studies, Fischlin et al. 2009). The diversity in these tropical regions suggests that some form of forest will continue to exist even with severe disturbance, but that many of the functions will change owing to the lack of resilience and new states, in general, will produce considerably less goods and services while supporting less biodiversity than at present.

6. Conclusions and ecological principles

The biodiversity in a forest is linked to and underpins the ecosystem's productivity, resilience, and stability over time and space. Biodiversity increases the long-term resilience and resistance of forest ecosystem states, increases their primary production, and enhances ecosystem stability at all scales. While not all species play important functional roles in ecosystems, many do, and we may not know or understand the role of a given species. Further, under changed environmental conditions, species with previously minimal functional responsibilities may become highly functional. The persistence of these functional groups within ecosystems is essential for ecosystem functioning and resilience. Capacity for resilience and ecosystem stability is required to maintain essential ecosystem goods and services over space and time. Loss of resilience may be caused by the loss of functional groups, environmental change such as climate change, or alteration of natural disturbance regimes (Folke et al. 2004). Loss of resilience results in a regime shift, often to a state of the ecosystem that is undesirable and irreversible. Resilience needs to be viewed as the capacity of natural systems to self-repair based on their biodiversity, hence the loss of biodiversity could mean a reduction of that capacity. This review, together with those of Loreau et al. (2001), Hooper et al. (2005), and Drever et al. (2006), suggested strong support for the following concepts specific to forest ecosystems and their resilience:

1. Resilience is an emergent ecosystem property conferred at multiple scales by the biodiversity in the forest system. More specifically, forest resilience is related to genetic diversity, functional species diversity, and ecosystem diversity (beta diversity) across a forest landscape and over time (table 2).

2. Most natural forests are highly resilient ecosystems, adapted to various kinds of perturbations and disturbance regimes; but if disturbance exceeds the capacity of the forest to recover (forest degradation owing to human use, for example, which reduces functional components), the system will recover to a different state that may or may not also be highly resilient, but which is unlikely to provide the former level of goods and services.

3. Complex forest ecosystems are generally more productive and provide more goods and services than those with low species richness. Productive forests dominated by mature trees are generally highly stable ecosystems.

4. There is niche differentiation among some tree species in a forest, as well as competition, leading to complexity and variability within and among forest ecosystems and their processes. Some of this variability is related to idiosyncratic local site conditions.

5. Redundancy of functional species is common in complex forest ecosystems and is directly related to ecosystem resilience. Redundancy provides insurance against changing environmental conditions, and species with limited functions under one set of conditions may become driver species under an altered set of conditions.

6. Diverse forest systems are more stable (within defined bounds) than less diverse systems and this is partly related to a robust regional species pool and the beta diversity among ecosystems.

7. Nevertheless, even high diversity is no guarantee for ecosystem resilience once climate conditions move beyond those experienced by most of the component species.

8. Although a forest may change states in response to disturbances, the flow of goods and services may not necessarily be highly altered, suggesting that the ecosystem is ecologically resilient, even though the forest community structure may have changed. Ecological resilience is unlikely, however, in a system that has low redundancy, such as degraded forests.

9. There is a negative relationship between species diversity, landscape diversity, and the capacity of a forest system to be invaded, especially by pests and diseases.

10. Not all forest ecosystems are equally resilient to disturbances, including climate change. Effects of climate change will vary in forests depending on biome, tree species composition, natural disturbance regime, and moisture, temperature and edaphic responses to climate change.

11. Resilience is necessary to maintain desirable ecosystem states under variable environmental conditions.

6.1 Ecological principles to foster forest ecosystem resilience and stability under climate change

Forests have a capacity to resist environmental change owing to their multiple species and complex multiple processes. However, a reduction in biodiversity in forest systems has clear implications for the functioning of the system and the amounts of goods and services that these systems are able to produce. While it is relatively simple to plant trees and produce a short-term wood crop, the lack of diversity at all levels (i.e., gene, species of flora and fauna, and landscape) in these systems reduces resilience, degrades the provision of goods and services that the system can provide, and renders it vulnerable to catastrophic disturbance.

Specifically, with respect to mitigating CO2 emissions from deforestation and degradation, maintaining long-term stable forest ecosystems will be critical, as opposed to for example, rapidly growing simple low diversity forests that have limited longevity, resistance, resilience or adaptive capacity. Further, the application of ecological sustainability principles in the recovery of degraded forests to redevelop their resilience and their former goods and services will provide part of a long-term approach to mitigating and adapting to climate change (e.g., Lamb et al. 2005, Innes et al. 2009). Hence, maintaining resilience in forests, in time and space, is important to maintain their function as an important "buffer" in the global carbon cycle by maximizing their potential to sequester and store carbon; along with the ongoing capacity to provide the other goods and services that humans require. To this end, human use of forests will need to change in order to ensure their conservation, sustainable use, and restoration.

In managed forests, it is imperative that biodiversity and ecosystem resilience be maintained. The principles of sustainable forest management are to maintain ecosystem processes by matching management practices to natural processes (or expected processes, modified under climate change) at multiple scales (e.g., Attiwill 1994, Perera et al. 2004). Restoration of degraded forest landscapes can take advantage of the linkage between biodiversity and ecosystem resilience, by planting to enhance species richness and through the addition of functional species (e.g., N-fixing species) where known (see: Lamb et al. 2005, Brockerhoff et al. 2008, for management recommendations). Various options for policies and measures are available to promote forest conservation and biodiversity, particularly at landscape and regional scales, in addition to conventional protected areas, including payments for land stewardship and ecosystem services (USDA 2007), connectivity conservation programmes (Crooks and Sanjayan 2006), and schemes built around recognition of Indigenous and traditional lands (Australian Government 2007).

The capacity to conserve, sustainably use and restore forests rests on our understanding and interpretation of pattern and process at several scales, the recognition of thresholds, and the ability to translate knowledge into appropriate management actions in an adaptive manner (Frelich and Reich 1998, Gauthier et al. 2008). Caring for forests in ways that maintain their diversity and resilience is being made even more complex owing to climate change (e.g., Chapin et al. 2007, Kellomaki et al. 2008). We suggest the following as ecological principles that can be employed to maintain and enhance long-term forest resilience, especially under climate change (e.g., Thompson et al. 2002, Fischer et al. 2006, Millar et al. 2007, Innes et al. 2009):

1. Maintain genetic diversity in forests through practices that do not select only certain trees for

harvesting based on site, growth rate, or form (see e.g., Schaberg et al. 2008).

2. Maintain stand and landscape structural complexity using natural forests as models and benchmarks.

3. Maintain connectivity across forest landscapes by reducing fragmentation, recovering lost habitats (forest types), and expanding protected area networks (see 8. below).

4. Maintain functional diversity (and redundancy) and eliminate conversion of diverse natural forests to monotypic or reduced species plantations.

5. Reduce non-natural competition by controlling invasive species and reduce reliance on non-native tree crop species for plantation, afforestation, or reforestation projects.

6. Reduce the possibility of negative outcomes by apportioning some areas of assisted regeneration with trees from regional provenances and from climates of the same region that approximate expected conditions in the future.

7. Maintain biodiversity at all scales (stand, landscape, bioregional) and of all elements (genetic, species, community) and by taking specific actions including protecting isolated or disjunct populations of organisms, populations at margins of their distributions, source habitats and refugia networks. These populations are the most likely to represent pre-adapted gene pools for responding to climate change (Cwynar and MacDonald 1987) and could form core populations as conditions change.

8. Ensure that there are national and regional networks of scientifically designed, comprehensive, adequate, and representative protected areas (Margules and Pressey 2000). Build these networks into national and regional planning for large-scale landscape connectivity.