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The 1994 Eastside Screens Large-Tree Harvest Limit: Review of Science Relevant to Forest Planning 25 Years Later

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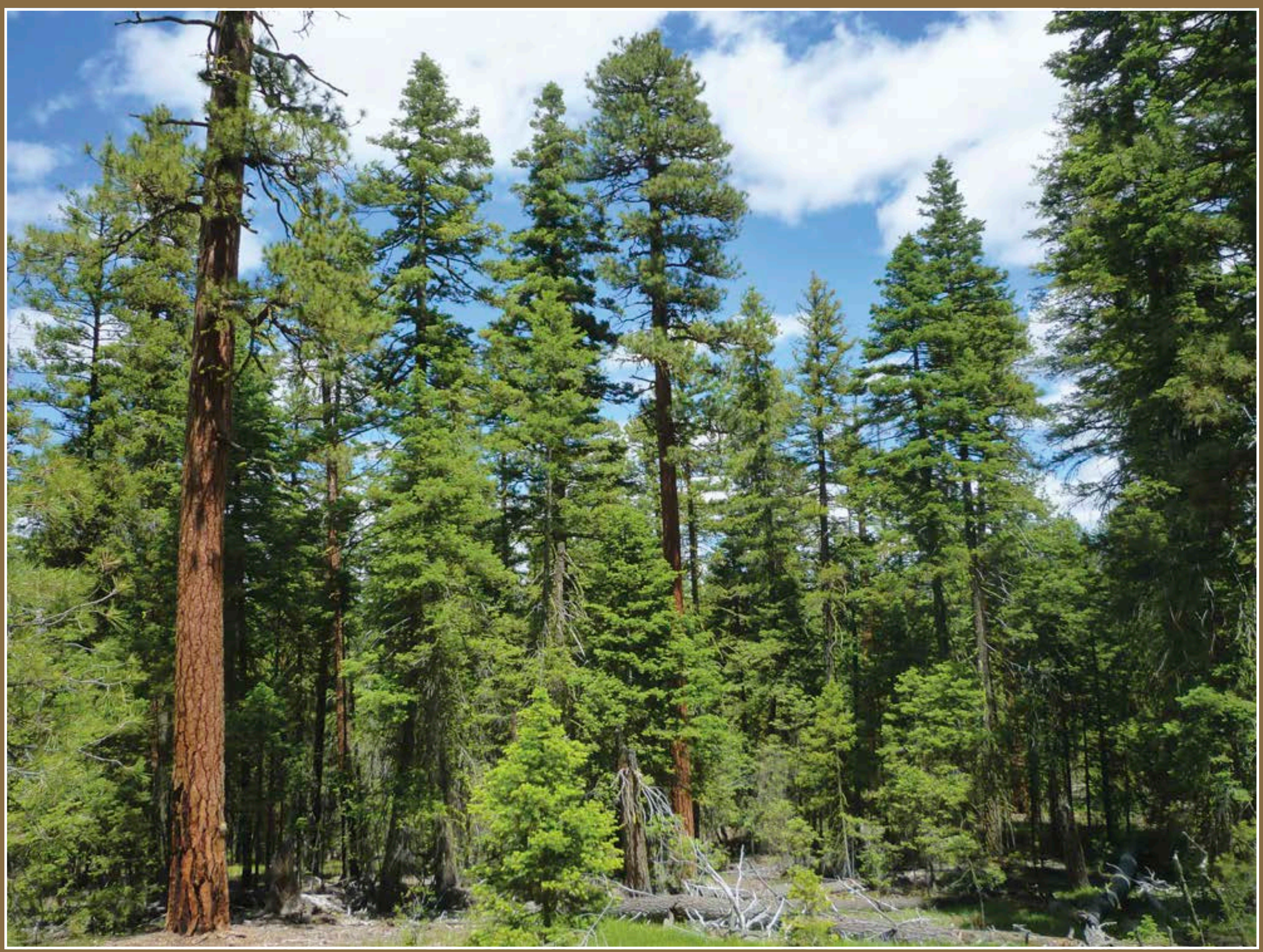

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Cover photos: Dugout Creek Research Natural Area, Oregon.

Abstract

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In 1994, a large-tree harvest standard known as the "21-inch rule" was applied to land and resource management plans of national forests in eastern Oregon and Washington (hereafter, the "east side") to halt the loss of large, old, live, and dead trees and old forest patches. These trees and forest patches have distinct ecological, economic, and social values, as reflected in widespread fish and wildlife use, public support for protecting them, and commercial interest in harvesting them, thus they have been the topic of much discussion and debate. At the request of regional Forest Service managers, we review the scientific knowledge accrued since implementation of the 21-inch rule and discuss the rule's role and relevance to forest planning today.

Critical to our review are new findings from the social sciences and their integration with new biophysical and ecological science to form a more holistic understanding of forest ecosystems and the values they provide. We examine how human values associated with old trees and old forests are nuanced and evolving and discuss important social and economic changes relevant to large, old trees and old forests that have occurred across the Pacific Northwest in the past three decades.

Major advances also have been realized in landscape and fire ecology, climate and carbon science, and wildlife, fishery, and silviculture sciences related to the role and importance of large and old trees in east-side forests. Key findings show that trees of early-seral species that are older than 150 years contribute important ecological values not present in younger large trees. Other findings come from climate change research, landscape assessments, and fire history studies, which have contributed knowledge about the historical and likely future variability in fire frequency and severity in various forest types, landscape dynamics, and how landscape resilience works.

Many forests are now homogenized, with conditions no longer resembling those that existed prior to Euro-American settlement. Disturbance regimes have become more severe in many places, causing widespread ripple effects. The area burned by wildfire will continue to increase under climate change, and disturbance regimes will change further, leading to even broader changes in forest structure and species composition. Moderate or severe fires or fuel treatments, coupled with maintenance burning, may be needed to remove local seed sources and competition from undesirable shade-tolerant trees and help some patches of forest better adapt to fire and climate change. Proactive management can help facilitate some transitions, leading to better outcomes for people, forests, and native species.

Keywords: Climate change adaptation, landscape restoration, wildfire vulnerability, remnant large trees, old forests, old-growth associated species.

Contents

Introduction

Purpose and Need

This review provides agency planners and decisionmakers with a synthesis of current knowledge as it relates to evaluating the 21-inch rule in eastern Oregon and Washington (hereafter, the "east side"). It represents a rapid review of available information by U.S. Forest Service (USFS) scientists drawing on research relevant to the large-tree harvest standard; prior science syntheses involving USFS researchers, and scientists and managers from academic, governmental, university, and nongovernmental organizations (e.g., Halofsky and Peterson 2017; Spies et al. 2018a, 2018c, 2019; Stine et al. 2014); and new peer-reviewed publications, and refereed government and institutional reports by agencies not included in prior syntheses.

Box 1

Three questions framed our review:

- What was the scientific foundation leading to the 21-inch rule and the decision to amend east-side forest plans?
- What science has emerged since then that is relevant to the rule?
- What other important ecological or social considerations provide further context for evaluation of the rule?

The 21-inch rule was implemented in 1994 to translate then-current science into new management direction. This review does not critique past decisions but explores available knowledge acquired since then and its usefulness in relation to the 21-inch rule. Some of the knowledge summarized in this report comes from advances in the social sciences; these findings have improved our understanding of human values, citizen engagement, and policy and management acceptance by communities and society at large. Other knowledge comes from the biophysical and ecological realms.

Flow of the review—

We divided our review into the following nine sections. In turn, we:

1. Introduce and provide background on the 21-inch rule.
2. Review the ecological science and management context of the 1990s that contributed to development of a rule limiting harvest of large trees.
3. Discuss the social, economic, and political drivers that have emerged from

social science research over the past 30 years, which provide important framing for east-side forest planning in the current decade.

4. Summarize recent climate science and how a changing climate likely shapes the future vulnerability and resilience of forest landscapes and large trees.
5. Examine findings from silviculture research that inform our understanding of stand development, succession, and disturbance processes, and their relevance to large and old trees of various species.
6. Discuss the contributions of large and old trees to biodiversity and aquatic, riparian, and terrestrial habitats.
7. Consider the importance of large and old trees to carbon sequestration, distributed hydrology, and other ecosystem services.
8. Synthesize how our overall scientific understanding of east-side Oregon and Washington forests has changed over the past three decades.
9. Discuss context provided by the 2012 Planning Rule[1] that is relevant and timely to forest (and project level) planning and to long-term implications of the 21-inch rule.

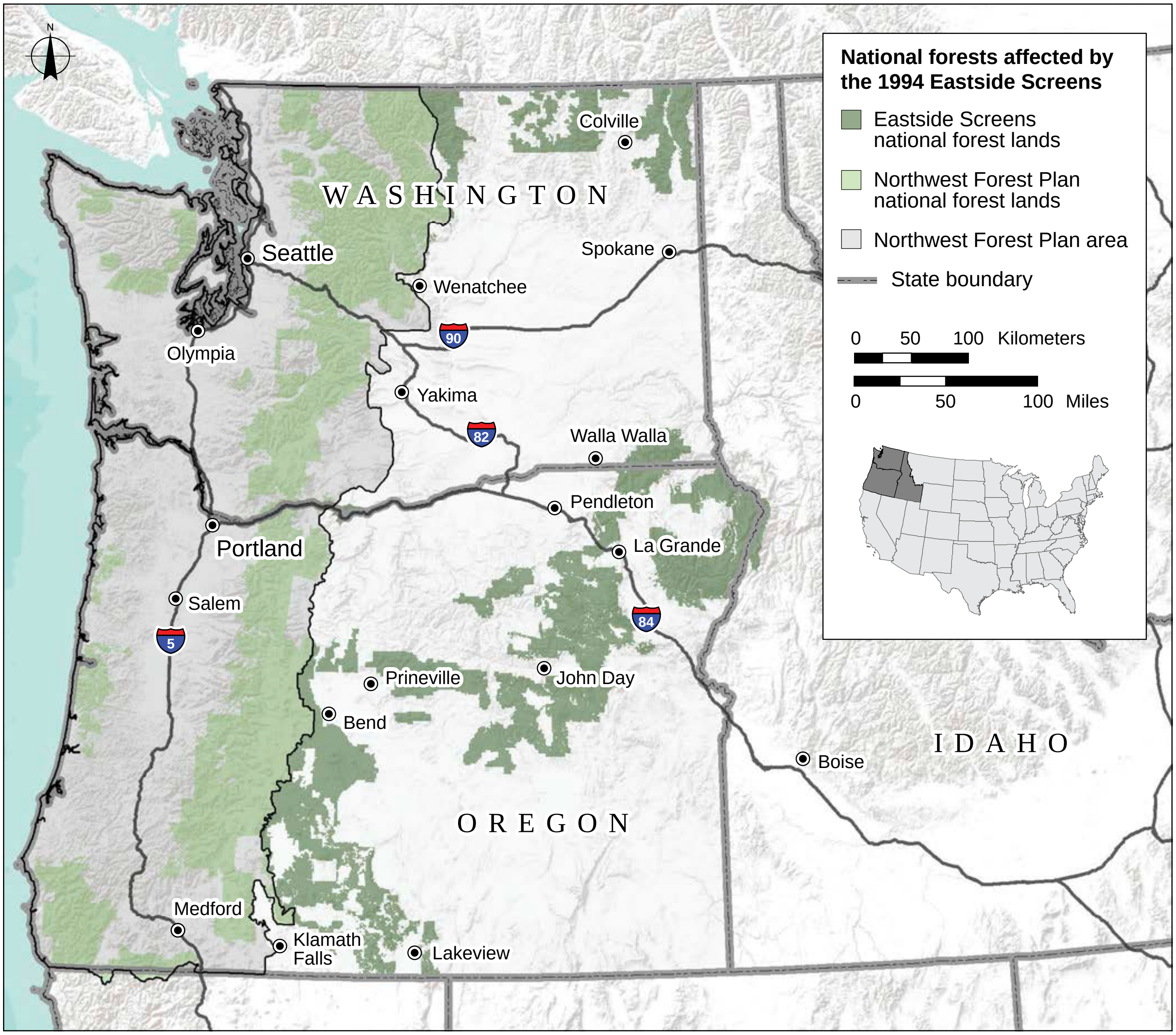
Background

Land and resource management plans (hereafter, forest plans) for national forests in eastern Oregon and Washington[2] (hereafter, east-side forests) (fig. 1) retain a 25-year-old standard that prohibits the harvest of live trees 21 inches in diameter at breast height (dbh) and larger.[3] The standard amended forest plans in 1995 and has been in place ever since. Because east-side national forests are beginning a process of amending their forest plans, we were asked to synthesize the science of the past

[1] The 2012 Planning Rule refers to a national-level rule that provides overarching guidance for amendment of all national forest land and resource management plans (USDA FS 2012a, 2016).

[2] East-side national forests influenced by the 21-inch rule were the Colville National Forest and portions of the Deschutes, Fremont, Malheur, Ochoco, Okanogan, Umatilla, Wallowa-Whitman, and Winema National Forests (Powell 2013). The Wenatchee National Forest and a western portion of the Okanogan National Forest were exempted as northern spotted owl forests covered by the Northwest Forest Plan (fig. 1). The Colville is now exempted because its forest plan is newly revised.

[3] The 21-inch rule defined (1) All sale activities (including intermediate and regeneration harvest in both even- and uneven-age systems, and [postfire] salvage) will maintain snags and green replacement trees ≥21 inches diameter at breast height (dbh) (or whatever is the representative dbh of the [dominant] overstory layer if it is less than 21 inches), at 100 percent potential population levels of primary cavity excavators. This should be determined using the best available science on species requirements as applied through current snag models or other documented procedures (USDA FS 1995: app. N). See also other references to the 21-inch rule interwoven elsewhere in the Eastside Screens (USDA FS 1994, 1995).



Robert Hoyer

Figure 1—National forests in eastern Oregon and Washington affected by the 1994 21-inch rule.

25 years as it relates to this standard, place it within larger ecological, economic, and sociological contexts, and discuss lessons learned.

A large-tree harvest limit (commonly referred to as the "21-inch rule") was an agency decision to stanch the loss of large, live, older trees and old forest patches. The decision arose from new understanding and valuing of the ecological functions of large old trees and old forests in fire-prone and other forests, and from recognition of their significantly reduced abundance and altered spatial distribution. Developed as a temporary (12 to 18 month) measure (Powell 2013), it remains a current standard of affected forest plans.

Restoration-based management seeks to restore and adopt the structure, composition, and processes that are characteristic of healthy, resilient forests and watersheds, and to provide a wide range of ecosystem services for generations to come.

The rule was originally adopted by managers as an expedient way to remove cutting units from contested timber sales. Tree size was used as a surrogate for age as a practical matter because measuring tree ages is a slower process than measuring tree diameters, and evaluation guidelines to expedite the process were unavailable. It was known at the time that some old trees are small, and some large trees are young, but the agency used tree size instead of age to protect what foresters thought was the bulk of the older trees (Powell 2013; USDA 1994, 1995). This would result in protecting the largest and many of the oldest trees, regardless of species, but it would exclude small older trees that did not make the cutoff.

In more recent decades, forest restoration has become a broad focus of management on federal lands nationally, and in eastern Washington and Oregon (USDA FS 2011, 2012a, 2012b). Restoration-based management seeks to restore and adopt the structure, composition, and processes that are characteristic of healthy, resilient forests and watersheds, and to provide a wide range of ecosystem services for generations to come. It emphasizes restoring ecological and human system resilience to wildfire, insect, and pathogen disturbances; extreme effects of climate change; and shifting natural resource and socioeconomic conditions. The timing of forest plan amendment and management focus on forest restoration on federal lands make evaluation of the 21-inch rule opportune.

The 21-Inch Rule—Ecological Science and Management Context in the 1990s

Released in 1994, the Northwest Forest Plan (NWFP), which guides the management of 19 national forests in Washington, Oregon, and northern California, was intended to conserve old forests, associated wildlife species, and Pacific salmon, while providing for a predictable and sustainable supply of forest products to support rural communities and economies (USDA and USDI 1994). It has served that habitat function for more than 25 years, but has not met its social system resilience or timber supply goals (Spies et al. 2019). The NWFP protects late-successional and old forests[4] and associated species on nearly 10 million ha (24.7 million ac), 60 percent of which are found in western Oregon, Washington, and

[4] Terminology for late-successional, old-growth, and old forests can be confusing. Late-successional forest (that has grown long enough for shade-tolerant trees to become dominant), is not the same as old growth or old forest, where trees are old (e.g., >150 years old), but not necessarily very large or shade tolerant. In this report, late-successional forest refers to forests that are often multilayered and dominated by older shade-tolerant tree species. Old growth or old forests will refer to forests with older trees, especially shade-intolerant and fire-tolerant species. See Spies et al. 2018b for a much expanded discussion of these terms.

northern California forests; the remaining 40 percent are in east-slope Cascade Range forests of Oregon and Washington. Most east-side national forests are not in the NWFP area because they are outside the range of the northern spotted owl (*Strix occidentalis caurina*), which defined the plan's boundaries (Thomas et al. 2006).

Concurrent with the release of the NWFP was that of the Eastside Forest Ecosystem Health Assessment (EFEHA) (Everett et al. 1994, Huff et al. 1995, Lehmkuhl et al. 1994). At the time, U.S. Speaker of the House Thomas Foley (Washington) and U.S. Senator Mark Hatfield (Oregon), responding to mounting concerns of constituents, requested that U.S. Department of Agriculture Secretary Edward Madigan convene an interagency science panel (the EFEHA scientists) to evaluate ecosystem sustainability and appropriateness of management practices on east-side national forests, and make recommendations for restoring stressed or damaged ecosystems. The panel was also asked to evaluate the adequacy of scientific information for managing east-side forest ecosystems sustainably. Findings of the panel were to address seven key questions defined by Representative Foley and Senator Hatfield (Everett et al. 1994: 4 and 7–46).

Key findings of the EFEHA included significant loss of live and dead, large, early-seral trees and old forests, and fragmented landscapes driven by small harvest units (fig. 2). The assessment likewise found that forest successional patterns and species composition were no longer characteristic of nearly all forest types, and that conditions were ripe for often large and severe insect, disease, and wildfire disturbances because of large increases in contagion of forested area, or density, or shade-tolerant forest cover. Because they were not part of its charge, the EFEHA did not address social or economic concerns but did acknowledge their importance to ecological and human ecosystem sustainability and identified the need for more information about social values and expectations for management of east-side forests.

Coincident with the release of the EFEHA, the Natural Resources Defense Council (NRDC) petitioned the courts to suspend large-tree harvests on east-side forests outside the NWFP area. Subsequently, John Lowe, then regional forester of the Forest Service Pacific Northwest Region, asked the EFEHA team to devise an interim screening rule for large-tree removals that could be applied to vegetation management and timber sale projects. Within a month, the EFEHA science panel devised the requested rule and recommended sunsetting it within 12 to 18 months (Powell 2013), to be replaced by more formal landscape evaluations that responded to EFEHA key findings. Their recommendation aligned with a need recognized in the EFEHA for flexibility and options in future forest management, given changing public expectations about forest management, changing public perceptions of disturbance-related risks, and the dynamic nature of forests and the factors that shape them.

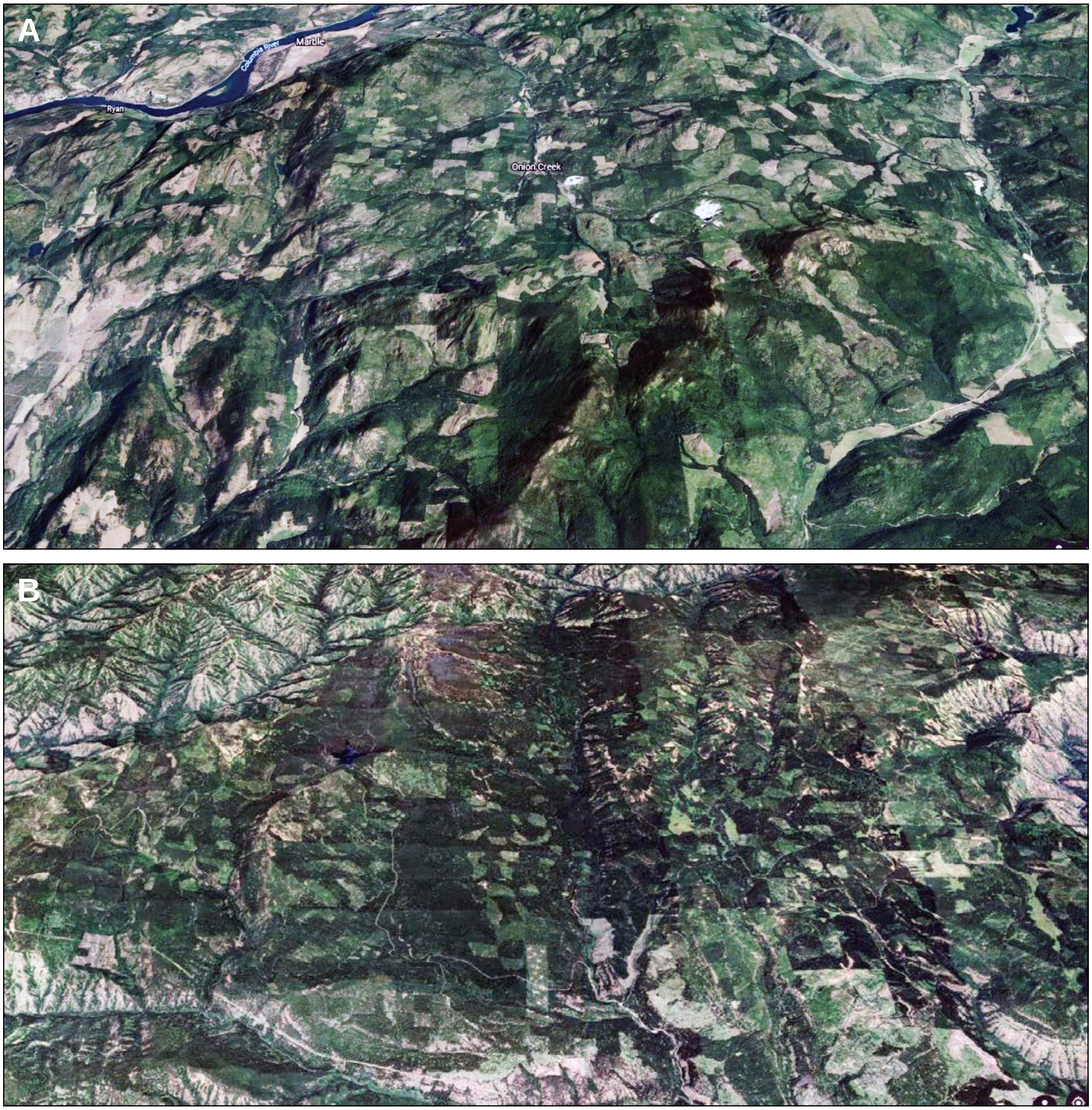


Figure 2—Small harvest units created by 20th-century tree harvests fragment national forest landscapes on (A) Umatilla National Forest, Oregon, and (B) Colville National Forest, Washington. Source: Google Earth™.

The EFEHA team transmitted the large-tree rule to Regional Forester Lowe in April 1994, recommending that the harvest of large and old, live trees be avoided, but they did not specify a diameter limit. In the EFEHA, medium- to large-size trees were 19 to 25 inches dbh, and mature and larger trees were >25 inches dbh. The gist of their recommendation was to conserve remaining large trees and old forests and encourage management actions that promoted their increased abundance and distribution. Thereafter, cutting units involving harvest of large and old trees were removed from the enjoined timber sales, and the sales were cleared for award to purchasers. A lower end size limit of 21 inches was negotiated with the plaintiffs included in the NRDC petition.

The rule was implemented in May 1994 as part of a larger set of "Eastside Screens" (see Powell 2013 for dates and details). Lowe signed the decision notice, and that decision amended forest plans on all east-side national forests except the Wenatchee National Forest and a portion of the Okanogan National Forest because they fell under the jurisdiction of the NWFP. The decision incorporated this interim direction and several other screens as new standards and guidelines (Powell 2013, USDA 1995).

Social, Economic, and Political Contexts

Social Values Associated With Large, Old Trees

Oregon and Washington residents value large, old trees for a variety of reasons, and the majority favors old-growth protection.

In the Pacific Northwest (as elsewhere in the United States), large and old, live trees were historically an important source of timber for the forest products industry, and early federal forest management emphasized their economic value. Timber production from federal lands contributed to economic well-being in many forest communities in the region and engendered important social and cultural meaning among workers in the forest products industry (Charnley et al. 2018). For instance, among loggers, working in the woods fostered a sense of identity, pride, independence, membership in an occupational community, strong social ties, and a rural lifestyle (fig. 3) (Carroll et al. 2000, 2005).

Courtesy of the Forest History Society

Figure 3—Logging built a sense of community among forest workers. (A) Professional timber faller stands next to a recently felled large Douglas-fir. (B) Loggers are positioning Silvey's hydraulic jacks prior to directional felling. (C) Two men are falling a large, old ponderosa pine with a crosscut saw.

Box 2

Old Growth and Public Values

In Oregon and Washington, public values pertaining to the management of old-growth forests in national forests have evolved. Whereas a 1986 survey of Oregon residents showed 70-percent support for old-growth logging, by 1997, 75 percent of Oregon and Washington residents surveyed believed that old growth should be protected from logging, with only slightly more support for this position in urban versus rural counties (Davis et al. 2001). Other studies conducted during the 1990s found that most area residents favored old-growth protection (Charnley and Donoghue 2006). Similarly, research carried out in Oregon and Washington during the 2000s found that most people had negative responses to harvest treatments within old-growth forests (Olsen et al. 2012; Ribe 2009, 2013). However, Ribe (2006) found some disagreement around the acceptability of harvest treatments in old growth, and Ribe and Matteson (2002) found that reactions to harvest treatments differed based on respondents' protection- or production-oriented attitudes, with nonaligned respondents generally favoring old-growth protection.

Since the 1990s, the noneconomic values of large and old trees have received more management attention.

Since the 1990s, the noneconomic values of large and old trees have received more management attention. Old-growth trees have intrinsic and spiritual value for some people (Blicharska and Mikusiński 2013, Lee 2009, Moore 2007), and are perceived by some as being primeval, ancient, irreplaceable, fragile, and endangered (Kimmins 2003, Peterken 1996). Because large, old trees can appear to be "permanent" and "unchanging" over a human lifespan, and endure across many human generations, they can serve as longstanding landmarks that link generations (Blicharska and Mikusiński 2014) and represent a perceived "balance of nature" (Kimmins 2003). The decline of old-growth forests has also caused them to be valued for their perceived scarcity and vulnerability (Pesklevits et al. 2011). More generally, members of the public often believe there is a strong relationship between old growth and biodiversity (Kimmins 2003).

Social Values Regarding Forest Management

Members of the public value national forests for their environmental benefits; studies from eastern Oregon and Washington find that the economic, recreational, and aesthetic values of forests are also a high management priority.

During the 1990s, there was surprisingly little change in the views of Oregon and Washington residents as to how the region's forests should be managed. Throughout this period, people supported forest management to provide a broad array of uses, and both economic and environmental benefits. There was consistently a pro-environment leaning, however, with the majority favoring environmental over economic management objectives when asked to make a choice between them (Charnley and Donoghue 2006).

Few studies have been published on values, attitudes, and beliefs regarding forests and their management in the region during the 2000s, and fewer yet have focused on east-side forests. However, a landscape values mapping study (Brown and Reed 2009) conducted on the Deschutes, Ochoco, and Mount Hood National Forests showed that of 12 predefined landscape values, the top five were developed recreation, primitive recreation, aesthetics, wilderness, and biological diversity. An east-side survey (which included Wallowa County in northeast Oregon and two counties in Idaho) found that, from a set of 13 landscape values, recreational, aesthetic, and economic values of forests were most important to respondents (Nielsen-Pincus 2011). Another survey found residents of Baker, Union, and Wallowa Counties in northeast Oregon to be more likely than nationwide samples to prioritize jobs and economic forest uses over some conservation concerns, such as forest fragmentation, overharvest, and wildfire (Hamilton et al. 2012). Of these respondents, 79 percent believed it was more important to use natural resources to create jobs than to conserve them for the future, and 85 percent said that loss of forestry jobs and income was a serious threat to their community (Hamilton et al. 2014). Only 34 percent of respondents said that overharvesting or heavy cutting of timber was a serious threat. Taken together, this research reveals the importance of recreational and economic uses of east-side national forests, but also their aesthetic values.

Social values regarding forest management often differ in accordance with demographic, social, and economic characteristics of populations. For example, some Oregon studies have found more support for conservation-oriented forest management among urban residents (Kline and Armstrong 2001, Schindler et al. 2002); Democrats (Hamilton et al. 2014, Kline and Armstrong 2001); and newcomers (versus native Oregonians or long-time residents) (Hamilton et al. 2014, Kline and Armstrong 2001). Thus, if managers focus on the dominant values associated with certain sectors of the population, they may overlook values held by other important and underrepresented stakeholders.

Box 3

Public Perceptions About Fuel Reduction Treatments

In general, there is a high level of support for fuels reduction on public lands with high wildfire risk, using both thinning and prescribed fire treatments, though residents in the wildland-urban interface generally prefer thinning treatments (McCaffrey et al. 2013). However, if a restoration project is perceived to be driven more by commercial interests than by ecological considerations, it is more likely to be contested by some members of the public, especially if restoration takes place in old-growth stands (Stidham and Simon-Brown 2011). Social acceptability of forest restoration activities is greatest when members of the public (1) perceive high wildfire risk and poor forest health, (2) are familiar with the proposed treatment types, (3) perceive treatments as being cost-effective and successful at achieving desired outcomes, and (4) trust the implementing agencies (Bright et al. 2007, Brunson and Shindler 2004, McCaffrey et al. 2013, Winter et al. 2002).

Social Values and Forest Restoration

> Active forest restoration is generally supported by Oregon and Washington residents, particularly in areas with high fire risk; however, if active management occurs in old-growth stands or is perceived as being mainly driven by commercial interests, it will more likely be contested.

The USFS has made investments in recent years to increase the pace, scale, and quality of forest restoration (USDA FS 2012b). The Eastside Restoration Strategy (USDA FS 2019) promotes active forest management to increase the resilience of forests to impacts such as insect outbreaks, spreading disease infection centers, and destructive wildfires on east-side forests (fig. 4). Research from Oregon and Washington shows that many citizens acknowledge the need for restorative treatments on federal lands in response to wildfire risk (Brunson and Shindler 2004, Shindler and Toman 2003), and that active forest management to reduce wildfire risk is generally supported, regardless of personal economic or environmental value orientation (Abrams et al. 2005, Charnley and Donoghue 2006). However, perceptions of wildfire risk differ. In northeastern Oregon, Hartter et al. (2015) found that perceptions of decreasing forest health and increasing wildfire risk were associated with having lived locally for more than 10 years, and with better understanding of forest management. Wildfire risk perception in eastern Oregon has also been correlated with past wildfire experience (Fischer et al. 2014, Olsen et al. 2017), among other variables.

All photos by John Marshall

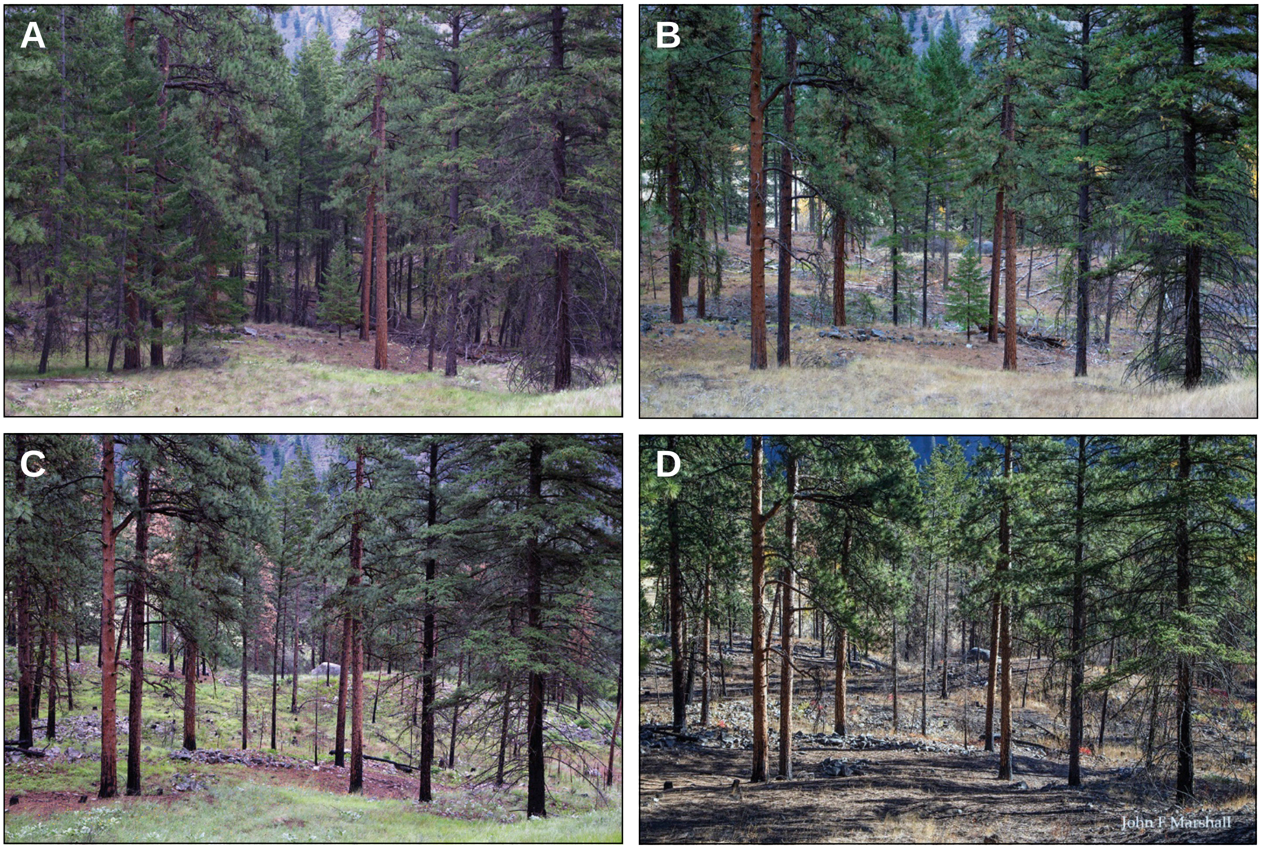


Figure 4—Active forest restoration treatment, Sinlahekin Wildlife Refuge, Washington Department of Fish and Wildlife. (A) Multilayered, dense, dry mixed-conifer forest after 100 years of fire exclusion. (B) Residual forest after a variable-density thinning treatment. (C) Treated condition after pile and broadcast burning. (D) Treated forest 1 year later, after all thinning and burning work was completed.

Large Trees and Tribal Values

> Many American Indian tribal members value old trees for ecological and cultural reasons; the USFS has an ongoing responsibility to consult with and engage tribes when making management decisions that affect tribal trust rights and resources within their ancestral territories that are now federally managed.

Many federally recognized American Indian tribes possess rights to national forest lands and resources that are codified in federal law and policy (Dockry et al. 2018). The USFS has established agreements with numerous tribes—both with and without formal treaty rights—that allow traditional resource harvesting within their ancestral lands (Long et al. 2018). Thus, the agency is required to protect tribal rights, assets, lands, and resources on lands that it manages. The USFS must also conduct government-to-government consultations with federally recognized tribes when making management decisions that can affect tribal trust rights and

Forests are important to tribes because of the environmental services they provide.

resources (Long et al. 2018); there are legal, rational, and equity reasons for doing so. Forests are important to tribes because of the environmental services they provide (e.g., filtering air and water). They are important for their role in sustaining habitats for fish and wildlife; for the foods, medicines, fuels, and materials they produce; and for their importance to tribal members' sense of place, all of which may help sustain the lifeways, cultures, and spiritual practices of tribal members (Gordon et al. 2013). For example, California black oak (*Quercus kelloggii*), found across California and into southwest Oregon, is a cultural keystone species for many local tribal members for whom it may play a fundamental role in their diet, materials, medicines, and spiritual practices (Long et al. 2016). Large-diameter oaks typically produce more acorns, one of the tree's most valued products, than small-diameter oaks (Long et al. 2016). At times, tribes have been concerned about the potential for conifers to displace large hardwoods like oaks and have sometimes encountered resistance to removing those conifers (Long et al. 2017).

Another cultural keystone species for tribes in the Pacific Northwest is huckleberry, especially thinleaf huckleberry (*Vaccinium membranaceum*) (Long et al. 2018, Steen-Adams et al. 2019). Forest tree size and distribution can have an impact on populations of this species, which are most prevalent in open-canopy forest patches of the western Oregon Cascade Range (fig. 5**)**, (Kerns et al. 2004). East of the Cascade crest, the Warm Springs, Wasco, and Northern Paiute peoples (comprising the Confederated Tribes of Warm Springs) historically used fire in the moist mixed-conifer zone to maintain and expand forest openings created by previous ignitions to promote thinleaf huckleberry shrubs and access to harvest sites (Steen-Adams et al. 2019). Cultural burns ceased by the 1940s, causing forest canopy closure and encroachment of other trees and shrubs. These changes contributed to a decline in huckleberry production in traditional harvest areas and declines in social and cultural traditions associated with huckleberry harvests (Steen-Adams et al. 2019). This underscores the importance of considering how changes in forest management can affect culturally important species.

Box 4

Tribes May Value Large, Old Trees for Their Important Ecological Roles

As Hatcher et al. (2017: 3) noted, protecting old trees and increasing their ability to survive is a top priority for forest restoration in the Klamath Tribal Forest Plan. The justification is that old trees formed "…the ecological backbone of the ecosystem" and "the most fire-resistant tree component and source of persistent snags and down wood."

Figure 5—American Indian woman picking huckleberries.

Some tribes also value large, old trees for their ecological roles. Natural resource managers for the Klamath Tribes (Klamath, Modoc, and Yahooskin) of south-central Oregon, together with other forestry professionals, developed the Tribal Forest Plan for managing a portion of the Fremont-Winema National Forest that was previously part of their reservation (Johnson et al. 2008).[5] This plan prioritizes restoration by decreasing stand density in complex mixed-conifer stands containing substantial old growth that is at risk of damage from severe wildfires (Hatcher et al. 2017). The plan also acknowledges the need to retain large old trees because of their importance to stakeholder groups and the character they provide to the forest. The Tribal Forest Plan focuses on protecting live, old trees (defined as >150 years old), rather than retaining live trees >21 inches dbh because young,

[5] More than half of the Winema portion of the Fremont-Winema National Forest was once part of the Klamath Indian Reservation. The tribes lost federal recognition in 1954 and lost their reservation lands as a result.

fast-growing trees over the 21-inch limit (particularly white fir [*Abies concolor*] and Douglas-fir [*Pseudotsuga mensiezii*]) actually pose a threat to older early-seral trees by acting as ladder fuels and competing for soil moisture and nutrients, warranting their removal in some cases (Hatcher et al. 2017). Beyond their important ecological roles, some large trees of particular species have significance as "legacy" trees, e.g., sugar pines (*Pinus lambertiana*) on ridges (Long et al. 2020, USDA FS 2018a). Care should be taken to protect them and to understand the contexts in which they occur. Encouraging age-based rather than size-based thresholds could be a useful way to promote conservation of some of these tribal values.

Large, old trees that have been culturally modified and bear evidence of historical or prehistorical human uses, such as scars from wood, bark, or sap harvest, are also highly valued. Some anthropogenic scars found on North American trees date back as far as the 1400s (Arno et al. 2008, Mobley and Eldridge 1992) (fig. 6). Deur (2009) described Klamath and Modoc tribal use of sap and inner bark (or cambium) from pine (especially lodgepole pine [*Pinus contorta*], ponderosa pine [*P. ponderosa*], and junipers [*Juniperus* spp.]) in south-central Oregon and northeastern California. The Klamath Tribes' historical practice of harvesting cambium for food and medicinal use was also documented by earlier anthropologists (Coville 1897, Spier

Robert Keane

Figure 6—Fire ecologist Steve Arno standing near a culturally scarred ponderosa pine in the Bob Marshall Wilderness Complex, Big Prairie, South Fork of the Flathead River, Montana. American Indians throughout the West regularly peeled the nutritious inner bark of live pines. Removing the thick and corky outer bark over a small chest-high section made inner bark peeling easier and increased the likelihood of tree survival. Bark peeling created distinctive scars on trees, a long-term indicator of this cultural modification. A high density of pine trees with bark-peeling scars indicates the location of a historical encampment (Josefsson et al. 2012).

1930). Today, culturally modified trees provide American Indian communities with a link to traditional cultural practices and beliefs, and a tie to the past (Deur 2009). They also provide information about traditional forest management practices, and beliefs about preservation and conservation (Turner et al. 2009). For instance, the partial harvest of tree products may have reflected a reverence for trees, as only parts of the tree were removed, keeping the tree alive (Deur 2009, Turner et al. 2009, Zahn et al. 2018). Government programs in the United States and Canada are crucial to the preservation of culturally modified trees (Mobley and Eldridge 1992). In that context, Franklin et al. (2013: 27) recommended conserving and restoring culturally modified trees as a management goal on eastern Oregon forests.

Social-ecological systems in the Pacific Northwest have been shaped by indigenous people for millennia, and there is great potential for integrating traditional ecological knowledge into forest management and decisionmaking (Charnley et al. 2007, Long et al. 2018, Steen-Adams et al. 2019). One effective way to do this is to directly engage traditional knowledge keepers as active participants in forest planning, management, and implementation (Charnley et al. 2007) (fig. 7). Collaboration in resource management can build trust between the USFS and American Indian tribes (Dockry et al. 2018). Other ways to build trust include upholding formal relationships and agreements, developing informal and personal relationships, practicing respect, listening, and demonstrating engaged leadership (Dockry et al. 2018).

Social-ecological systems in the Pacific Northwest have been shaped by indigenous people for millennia, and there is great potential for integrating traditional ecological knowledge into forest management and decisionmaking.

Susan Charnley

Figure 7—Forest managers and stakeholders engaging traditional knowledge keepers and tribal partners as active participants in forest planning, management, and implementation. Speaking is Karuk tribal member and Forest Service research ecologist Frank Lake, Pacific Southwest Research Station.

Public Trust in Agency Decisionmaking and the Role of Collaborative Approaches

Collaborative governance—

> Although they take many forms and do not represent all stakeholders, forest collaborative groups can help identify shared values and develop a shared vision for the social, economic, and ecological outcomes of forest restoration efforts.

Collaboration around forest management takes many forms. In the Pacific Northwest, and in the context of forest restoration, collaboration frequently refers to engaging with formal collaborative groups in the course of planning and monitoring forest restoration efforts (Cerveny et al. 2018).[6] Collaborative groups that include agency participants and other diverse stakeholders are prevalent throughout east-side national forests. Collaborative groups generally focus on identifying shared values and developing a shared vision for the social, economic, and ecological outcomes from forest restoration efforts. In some cases, collaborative groups (e.g., Blue Mountains Forest Partners) arrive at formally documented "zones of agreement" or "restoration principles" about what forest management practices are acceptable to the group and under what conditions (Davis et al. 2018b). Other groups, such as the Lakeview Stewardship Group on the Fremont-Winema National Forest, operate more informally and rely on generally accepted group norms about acceptable practices; no one size fits all.

The hope of collaboration is reduction in administrative objections and litigation, building trust in the agency, and increasing the pace of forest restoration efforts (Goldstein and Butler 2010). There is some evidence that collaboration led to a lower likelihood of project litigation in eastern Oregon (Summers 2014); however, further research is needed to affirm this finding over longer timeframes and in other locations. Similarly, there is evidence that collaboration, and the federal forest restoration funded by Oregon and Washington, have increased the extent of timber harvest and precommercial thinning (Davis et al. 2019a, Salerno et al. 2017). However, it is challenging to isolate the influence of collaboration on the pace of restorative treatments because USFS data and a reasonable baseline are generally lacking (White et al. 2015).

[6] Collaboration can also be used by the Forest Service to refer to other processes, such as selecting projects for the use of receipts retained from stewardship authority projects (often with a formal forest collaborative), or engaging with state partners and other stakeholders on Good Neighbor Authority projects (which have thus far included engagement with formal forest collaboratives in Washington state).

Although there has been much focus on collaborative groups to understand stakeholder and partner interests, collaborative participants do not represent a full constituency (Davis et al. 2018a). Furthermore, although collaborative groups are often perceived as local or place-based, they may not represent the broad perspectives of those communities that are local to a national forest (Davis et al. 2020). How collaborative involvement ultimately reflects and influences public support for restoration projects outside the collaborative remains unstudied. However, USFS staff and collaborative participants in eastern Oregon and elsewhere indicate that they believe that collaborative involvement improves project quality, in part because diverse stakeholder perspectives have been incorporated (Davis et al. 2019a).

Collaborative groups in the region engage with science in a variety of ways as they participate in project planning and outcome monitoring (Santo et al. 2018). Within the USFS Pacific Northwest Region, direct science engagement can take a variety of forms, but frequently includes inviting scientists to make presentations or synthesize research, participating in codeveloped research studies, or contracting with a scientist as an advisor (Davis et al. 2019a). Within the collaborative process, the primary effect of science engagement appears to be in improving shared understanding among participants and promoting collaborative agreement about management actions (Davis et al. 2018b, Santo et al. 2018). Collaborative participants in Oregon have cited science engagement as a key factor in improved collaborative dialogue and project quality (Davis et al. 2019a).

To date, most collaboration in the region has focused on providing input into planning projects and assessing how project outcomes compare to goals (Cerveny et al. 2018). Collaboratives have generally had limited input into project implementation activities, although participants in the Blue Mountains region have expressed the desire to be involved in implementation (Santo et al. 2018). Cheng et al. (2019) highlighted the tension that can result when agency implementers are unaware of collaborative agreements around a project, or do not pursue input from a collaborative when new issues arise during implementation. For example, in the case of a new issue such as harvesting large trees, when there is a lack of agreement and social license, collaborative input on implementation and monitoring are likely of high importance for building trust and a shared vision around the issue.

Increasing timber harvest volume is not a primary motivation of collaborative participants (Davis et al. 2017a). More common priorities are improving forest and watershed conditions, building trust, and getting projects implemented. The desire to collaborate on projects that will likely lead to implementation and contribute to trust is reflected in the avoidance of some issues where there is a greater difference in values among diverse participants (e.g., grazing or roads), and a pattern of

Box 5

Top-Down Versus Bottom-Up Direction

If partners and stakeholders perceive that management actions are primarily responsive to output targets set from above rather than meeting shared goals and the collaborative process, there can be tension and reduced social acceptance.

As Abrams (2019) described, a complication with the collaborative process is the tension and dissonance between the requirements placed on local decisionmakers to meet targeted outputs (e.g., acres thinned, volume sold) identified by Congressional appropriations and Forest Service leadership, and the accountability of local decisionmakers to processes and agreements forged through collaboration. Christensen and Butler (2019) described how bureaucratic pressures can cause tension in collaboration when supervisors of local decisionmakers inject direction or policy inconsistent with collaborative agreements and discussions with local agency staff. This tension can be exacerbated when collaborative participants receive mixed signals from the agency about who has decisionmaking authority or what the decision space is (Christensen and Butler 2019). For example, the extent to which harvesting large trees is perceived to be in response to meeting new timber production demands from above, rather than accountability to collaborative and stakeholder desires and goals, may reduce the acceptability of such actions.

focusing on forest settings and conditions where reaching agreement and shared values is more likely (e.g., dry mixed-conifer forests instead of moist mixed-conifer forests). Although some collaboratives have come to limited agreement around harvesting white fir over 21 inches dbh, and potential harvesting of trees over 21 inches is a discussed topic in some collaboratives (Davis et al. 2018a), collaboratives generally do not pursue agreement to harvest large ponderosa pine.

From the efforts of the Blue Mountains Restoration Strategy Team, we learned that top-down, large-scale planning efforts, broad direction on policies, and attempts to scale up collaborative agreements too quickly may not align with existing zones of agreement or collaborative interests. Doing so may reduce the willingness to accept these efforts and bring tension into the collaborative group (Huber-Stearns and Santo 2018). To the extent that forest plan amendment results in any change to the 21-inch rule, and that change is viewed as direction handed down from the Forest Service's regional or Washington offices, the Blue Mountains Restoration Strategy Team experience on the east side of the Pacific Northwest Region is informative. Ultimately, the top-down Blue Mountains Restoration

Strategy Team effort, viewed as dictated by the regional office, struggled to connect well with locally driven collaboration (Huber-Stearns and Santo 2018).

Trust—

For a new management or policy proposition such as altering the 21-inch rule, involving collaboratives early, taking the needed time for dialogue, and soliciting their input on implementation and monitoring will be important for building trust and a shared vision. Many stakeholders still mistrust federal management because of unsustainable harvest rates and methods in prior decades. Trust, or lack thereof, receives a good deal of attention in the forest collaboration literature. For example, Davis et al. (2018a) identified several core components of trust building among collaborative groups: safe environments for dialogue, informal interaction that builds opportunity for identifying shared values, representative group composition, discussion of details, and assurance that the USFS understood and implemented their collaboratively derived recommendations. Cerveny et al. (2018) added that collaboration can build trust when based on clear objectives, consistent communication, transparent processes, reasonable timelines, honored commitments, and opportunities for genuine engagement by diverse stakeholders.

Many stakeholders still mistrust federal management because of unsustainable harvest rates and methods in prior decades.

The literature is clear that collaborative support for new approaches, a broadened pace or scale of activities, or moving into new types of landscapes (e.g., moist mixed-conifer) can be slow going, and that trying to “skip” steps or take shortcuts in dialogue and trust building typically are ineffective (e.g., Davis et al. 2018a, Santo et al. 2018). Collaborative groups in the Blue Mountains region noted that engaging with new issues is both a challenge and an opportunity for collaborative

Box 6

Noncommercial Aspects of Restoration Projects

A complicating factor for acceptance of top-down direction that increases timber production is a belief among collaborative participants that the commercial components of restoration activities are being implemented, while noncommercial components (e.g., prescribed burning, watershed improvement activities, road maintenance and decommissioning) languish (Santo et al. 2018). This perception creates tension within some collaboratives (Santo et al. 2018). Outside reviews of restoration activities in eastern Oregon national forests have shown in recent years that although the area of timber harvest and precommercial thinning has expanded, prescribed burning and other noncommercial watershed restoration activities have not widely increased (Davis et al. 2019a, Salerno et al. 2017).

group function, and such efforts likely require a more deliberative pace (Santo et al. 2018). In their recommendations after reviewing the effectiveness of the Blue Mountains Strategy Team, Huber-Stearns and Santo (2018: 12) stated that "committing time upfront to solicit input and engage stakeholders in setting shared goals and objectives for the (new) project could have improved stakeholder and partner buy-in and engagement."

The influence of large trees on timber sale value—

Including larger trees in restoration prescriptions can increase the acreages where fuel treatments are financially feasible. Prestemon et al. (2012) showed that allowing the harvest of live trees over 21 inches increased the acreage in the West where fuel treatments were economically viable, even without considering avoided damage values. Throughout the West, including live trees over 21 inches in fuel treatment harvests increased the viable treatment area by 2.6 times. In Oregon and Washington, the economic viability of treatment on all lands, public and private, increased by about 2.5 times. In a prior study that used a different modeling technique and focused solely on frequent-fire and wildland-urban-interface (WUI) federal forests, uneven-aged treatments that included removing larger trees in the overstory resulted in a substantial increase in the acreage of federal forests where fuel treatment was economically feasible (Ince et al. 2008). In addition, these authors found that harvesting regimes that included larger trees (e.g., variable-density thinning) resulted in slightly lower harvesting and transport costs per unit volume than thin-from-below prescriptions.

Skog et al. (2006), using an earlier version of the Ince et al. (2008) model, drew similar conclusions to the other studies; however, Skog et al. (2006) went a step further by also considering harvest operations on different slope classes. In Oregon and Washington, they found that when considering treatments on cable yarding ground (slopes >40 percent), it was necessary to include still more large trees in fuel treatment prescriptions to improve economic viability because of increased yarding costs. The foregoing results might be considered within broader financial or commercial contexts as well. For example, Rainville et al. (2008) identified densely stocked stands in the Blue Mountains of Oregon where thinning might provide a reliable source of wood while accomplishing restorative treatments. They examined the quantity, distribution, and economic value of the fiber that might be derived using methods that favored improved forest density and species composition on 5.5 million ac of national forest land. Their findings confirmed local land manager accounts about the ability of the regional land base to support timber harvest targets and restorative treatments: legal restrictions, past harvest levels, and current practices have reduced the acreage available for harvest and restoration via mechanical means. Moreover, they found that on lands where active management is allowed, thinning the most densely stocked stands is not economically viable. Findings from the Rainville et al. (2008) analysis are helpful to establishing common understanding of Blue Mountains forest and economic conditions for managers who are trying to restore national forest lands.

Another study in Oregon and Washington (Nielsen-Pincus et al. 2013) examined the effects of availability and proximity of wood processing infrastructure (sawmills and biomass utilization facilities) to national forest ranger districts. They found that districts located near sawmill and biomass facilities treated more total area, and more area within the WUI, than those farther away; the threshold distance for this effect was about 40 minutes travel time to a wood processing facility. These closer districts also used stewardship contracting to accomplish fuels treatments and incorporated biomass use more into their treatments. Their results demonstrate the importance of nearby infrastructure to supporting restorative work on national forests.

Shifts in the Forest Products Industry

Mill closure and retooling—

The east-side federal timber supply is important to supporting local mills, which are essential for making restorative treatments financially feasible. It is important for managers and stakeholders to consider how large harvested trees can be processed locally to support local mills and be consistent with collaborative group goals.

The pattern of mill closures on the east side is consistent with patterns across the West and the United States as a whole. A literature review shows that mill closures result from a mix of factors, with timber supply changes being one contributor (Charnley et al. 2018). Other factors include reduced demand for wood products in the housing sector, technological improvements that increase efficiency, competition from other mills, and industry restructuring. The importance of the federal timber supply to mill success is much greater, however, in areas such as the east side, where federal forests comprise most of the productive forest lands.

Complete loss of milling infrastructure would present a significant challenge to implementation of fuels reduction in frequent-fire forests. Prestemon et al. (2012) showed that if no timber products could be sold from forest restoration actions, there was no place on the east side where the expected net economic benefit from fuel treatment would be positive, even when accounting for avoided wildfire damage. These results imply that, in the absence of the ability to sell timber, (1) fuel treatments on the east side would have to be paid for, and (2) it makes little economic sense to do fuel treatments when the only economic benefit is avoiding damage to property or natural resources from wildfire. But this is purely an economic view.

In general, mills have trended toward processing smaller logs over the past several decades (Gale et al. 2012, McIver et al. 2015) (fig. 8), but industry-wide specifics are unavailable to the authors. In some cases, east-side mills have added new infrastructure specifically focused on small-log processing (e.g., White 2018). Managers considering harvest of larger trees need to be cognizant of mill piece size restrictions, and investments to recapitalize milling infrastructure to more efficiently handle smaller material. Harvesting large trees from east-side forests only to have them shipped outside the local area for processing is inconsistent with community and local stakeholder motivations to positively affect local economies via restorative activities (Brown 2019, Davis et al. 2018b, White et al. 2015).

North 40 Productions, LLC

Figure 8—Small-diameter logs are increasingly common in some east-side mill yards. Shown is the Vaagen Brothers Mill in Colville, Washington.

The influence of federal timber supply—

Federal timber harvest does not occur in a vacuum; increasing the volume from federal forests can affect timber harvests from nearby private lands.

Federal timber harvest decisions can influence timber markets, including harvest choices made by private landowners and decisionmaking by wood processors.

Federal timber harvest decisions can influence timber markets, including harvest choices made by private landowners and decisionmaking by wood processors. Large increases in the total volume of timber harvested from federal forests can potentially reduce the stumpage values that private landowners can expect and result in declines in private timber harvest, especially in the short term (Adams and Latta 2005: Ince et al. 2008). This relationship would be most pronounced in places with a mix of public and private forests (versus a landscape with mostly private forests), but the pattern can play out across the West when one landowner dramatically changes harvest patterns over a short timeframe (e.g., Charnley et al. 2018).

Box 7

Stewardship Contracting

Sales of large trees under stewardship authorities may increase funding for other restoration activities, but some stakeholders oppose stewardship contracting, and there is inconsistent direction when using stewardship authority.

As noted, a potential benefit from harvesting large trees is the added funding for other restoration activities (Barbour et al. 2008). Using harvest revenue in this manner requires that timber sales be administered under stewardship authorities rather than as traditional timber sales, and also requires collaborative engagement for spending retained receipts (e.g., Mattor et al. 2019). Although widely used across the Pacific Northwest, the use of stewardship authority is not ubiquitous, and some stakeholder groups oppose its use. For example, some county government leaders have noted their dislike of stewardship authority (e.g., Daniels et al. 2018, Moseley and Charnley 2014) because timber harvested under that authority is not subject to revenue sharing with the county. Although promoted as a key means of increasing the extent of restoration treatments (USDA FS 2019) and recognized in the context of shared stewardship (USDA FS 2018b), decisions about whether to use stewardship authority to implement projects do not always reflect collaborative values or goals (Brown 2019).

Recent Climate Science: Future Vulnerabilities and Resilience of Forest Landscapes

Climate Change in the Pacific Northwest

Western U.S. climate projections suggest year-round warming and declines in summer precipitation.

Since development of the 21-inch rule, climate change has emerged as an important contextual issue in forest management. To prepare for its projected effects, all east-side national forests in the region have developed science-based climate change vulnerability assessments and adaptation options (Gaines et al. 2012; Halofsky et al. 2019b; Raymond et al. 2014). Key climate trends, vulnerabilities, and management options relevant to forest management and large trees are highlighted below.

The climate has warmed on the east side during the past several decades; warming will continue throughout the 21st century. Mean annual temperatures in Oregon and Washington have increased by 1.54 °F since the beginning of the 20th century (Vose et al. 2017), and will likely increase by 3.7 to 4.7 °F by the mid-21st

century (2036–2065) and by 5 to 8.5 °F at the end of the century (2071–2100), depending on which future greenhouse gas emission scenario transpires (Representative Concentration Pathway [RCP] 4.5 or 8.5) (Vose et al. 2017) (fig. 9). Warming will likely occur during all seasons, but most models project the largest increases to occur in summer (Mote et al. 2014).

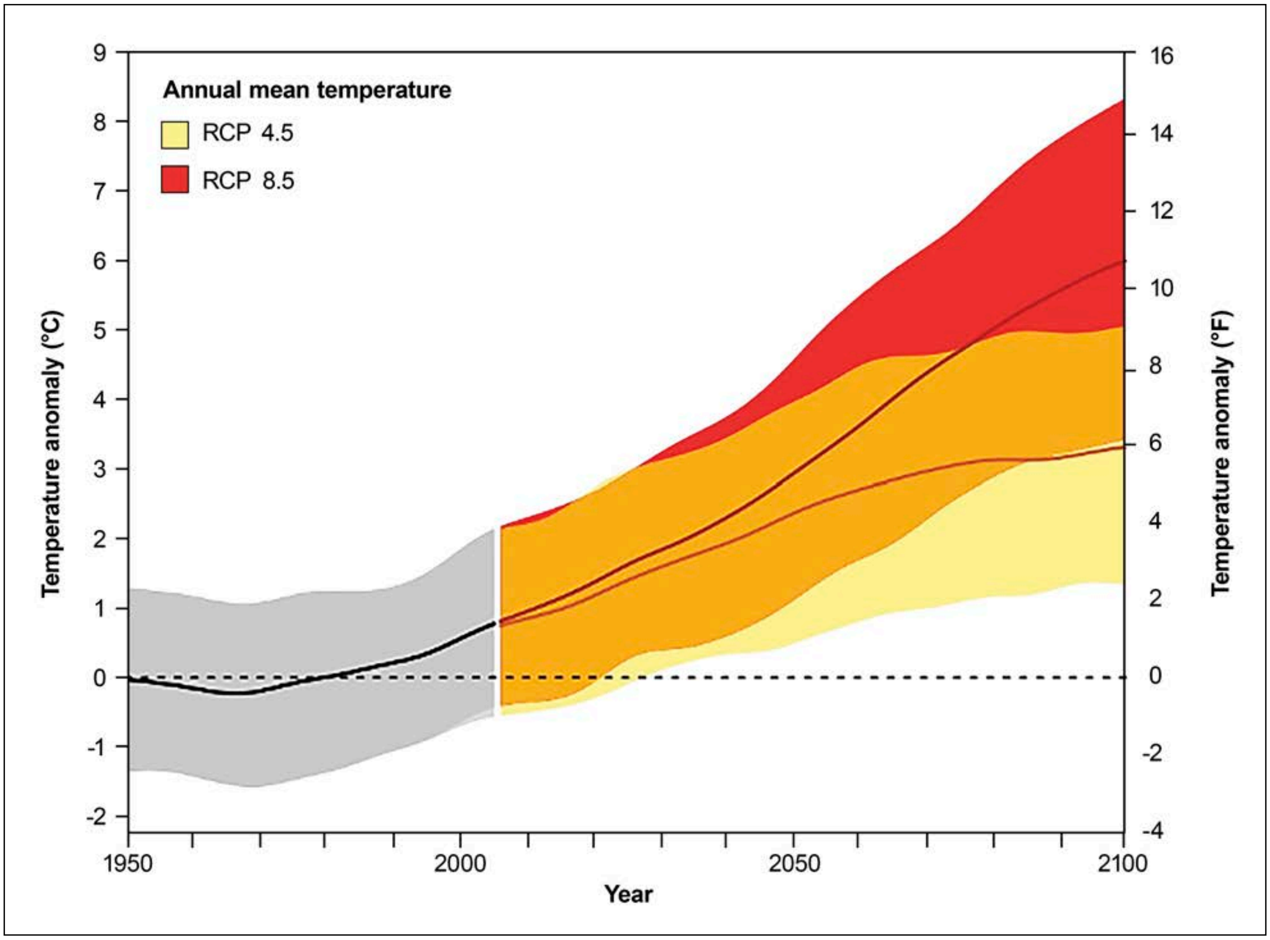


Figure 9—Projected annual temperatures for the Pacific Northwest to the year 2100. Annual temperatures are shown relative to the historical annual average (dashed line). Projections shown here represent the output of simulations from 35 global change models. The shaded areas provide a smoothed approximation of the range of likely temperature anomalies each year. In the Representative Concentration Pathway (RCP) 4.5 greenhouse gas (GHG) emissions scenario, the increase in GHGs is less than in the RCP 8.5 scenario, leading to a slower temperature increase and an eventual leveling off. The RCP 4.5 scenario is a climate change stabilization scenario, in which the radiative forcing level stabilizes at 4.5 W/m^2 before 2100 by employment of a range of technologies and strategies for reducing GHGs. In the high-emissions scenario, RCP 8.5, steady growth in GHGs leads to a steady upward trend in temperatures. The RCP 8.5 scenario is a climate change continuation or "business as usual" scenario, in which radiative forcing continues to increase and reaches 8.5 W/m^2 by 2100 owing to the lack of adequate employment of technologies and strategies for reducing GHG emissions. Data from Rupp et al. (2017).

Precipitation patterns in the region have not changed significantly in recent decades, according to Mote et al. (2014). However, Holden et al. (2018) showed that declines in summer precipitation and wetting rain days from 1979 to 2016 across one-third to nearly half of Western U.S. forests has been a primary driver of increased wildfire area burned. Future precipitation projections are uncertain, but many for the Pacific Northwest suggest little to no increase in annual precipitation (1 to 2 percent), longer dry periods between rain events, and precipitation that does not keep pace with warming (Mote and Salathé 2009, Mote et al. 2014). Precipitation and temperature influences during the fire season tend to strongly increase area burned. For precipitation, influence occurs either directly via changes in wetting or through feedbacks to vapor pressure, primarily creating deficits. If these trends persist, declines in summer precipitation interacting with increasing summer aridity will lead to more burned area across the West.

Climate Change and Wildfire

> A warming climate will have profound effects on wildfire regimes in the Pacific Northwest; fire seasons will lengthen and burned area will increase.

Climate and wildfire are closely linked in the Pacific Northwest, and if the recent past is prologue, fire and other disturbances will likely drive ecosystem change as the climate warms. Paleoecological records (charcoal and pollen deposited in varved [annually deposited] lake, bog, and fen sediments over hundreds to thousands of years) indicate that climate has been a primary control on area burned in the region for millennia, with strong interactions between fire and vegetation (Walsh et al. 2015, Whitlock et al. 2003). Although most of the sites where these records exist are on the west side of the Cascade Range, several east-side studies show the same strong linkages between climate, fire, and vegetation (Kerns et al. 2017).

A warming climate also has profound effects on fire frequency, extent, and severity in the region (Case et al. 2019, Kerns et al. 2017, Raymond et al. 2014). Increased temperatures and decreased summer precipitation will continue to lengthen fire and growing seasons, increase evaporative demand, decrease soil and fuel moisture, increase fuel availability to burn, and increase likelihood of large fires and area burned (Holden et al. 2018, Littell et al. 2010).

Increased temperatures and decreased summer precipitation will continue to lengthen fire and growing seasons.

Box 8

Number of Fires and Area Burned Increase Under Warmer and Drier Spring and Summer Conditions

Fire scar records of the past several centuries indicate that years with increased numbers of fires and area burned in the Pacific Northwest were generally associated with warmer and drier spring and summer conditions (Heyerdahl et al. 2008) (fig. 10). Contemporary fire records from the 20^{th} century similarly indicate that years with relatively warm and dry conditions, particularly in summer, have generally corresponded with larger fires and greater burned area (Abatzoglou and Kolden 2013, Littell et al. 2009). Decreasing fuel moisture and increasing duration of warm, dry weather creates large areas of dry fuels that more readily ignite and carry fire over a longer period (Littell et al. 2009).

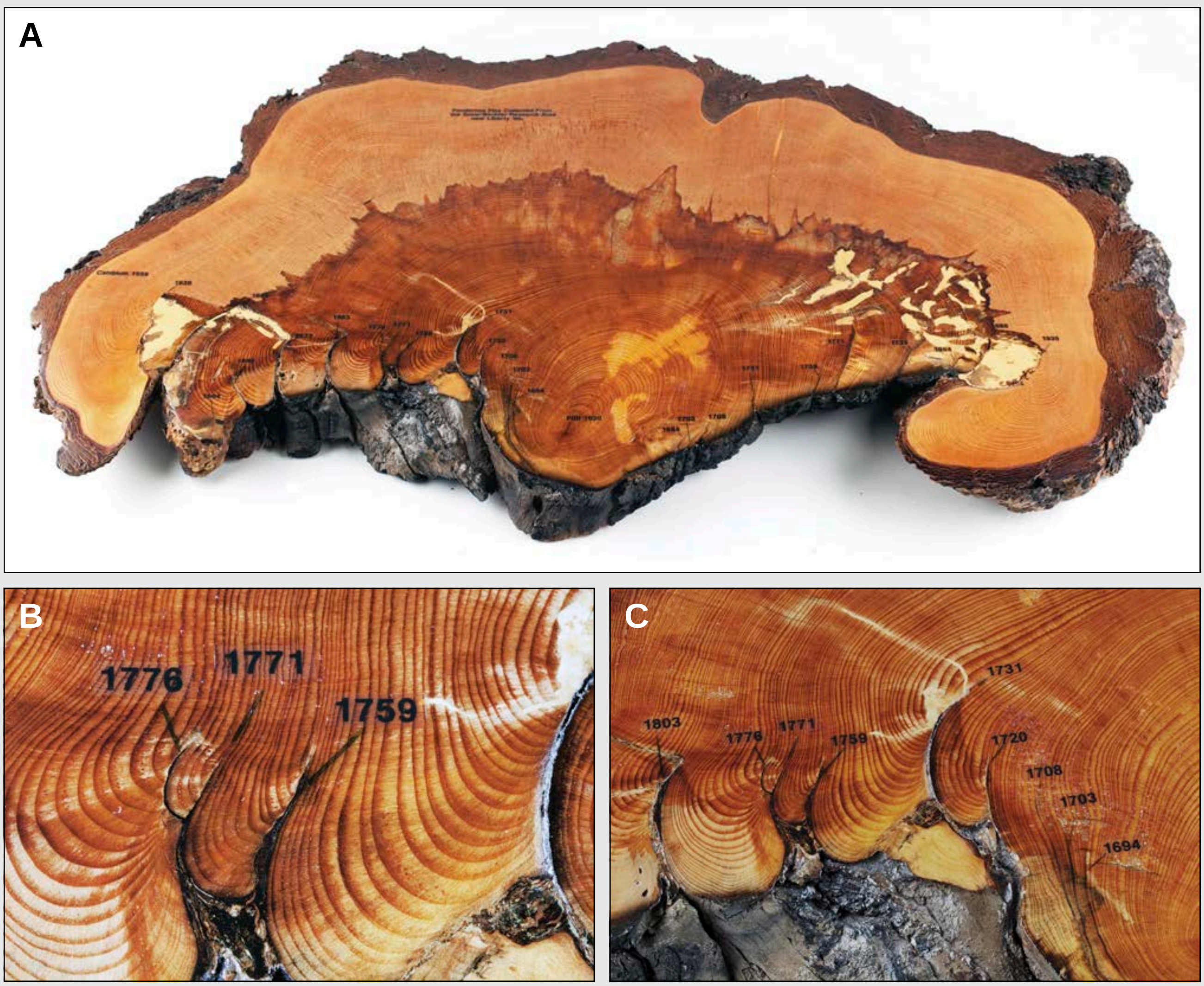


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Figure 10—(A) Fire scars sampled from a recently dead ponderosa pine on the Wenatchee National Forest in the Liberty, Washington, area. (B) Three fire scars in rapid succession between 1759 and 1776. (C) Close-up of fire scarring during the period 1694 to 1803. Note that fire scarring typically occurred in late summer during the period of latewood xylem development (darker portion of each annual ring).

Modeling Wildfire Under Climate Change

For the east side, by the end of the century, models project a three- to fourfold increase in burned area, increasing fire size, and increasing severity in forests that are dense, layered, and abundant with woody fuels.

Statistical and simulation models are our best method for understanding possible effects of climate change. In the eastern Cascade Range, Okanogan Highland, and Blue Mountains, statistical models developed using existing data suggest that mean area burned will increase nearly fourfold in the 2040s, compared to the 1980–2006 period (Littell et al. 2010). Models also suggest that the occurrence of large fires will increase; the proportion of forested area that is highly suitable for fires >100 ac will increase from the current extent of 17 to a range of 63–72 percent in the Blue Mountains, and from 11 to 40–45 percent in the eastern Cascades by the end of the century (Davis et al. 2017b). Process models similarly project increases in area burned and fire size on the east side (e.g., Case et al. 2019, Cassell et al. 2019, Halofsky et al. 2013, Kerns et al. 2017, Kim et al. 2018). Wildfire will become more frequent in most vegetation types, especially dry and moist forests (Case et al. 2019).

Future fire severity will depend on vegetation composition and structure, the abundance of woody surface fuels, the density and layering of forests and distribution of forested area, and growing-season fire weather and drought. In the near term, high forest density and layering as a result of fire exclusion and past management will likely increase fire severity in dry and moist mixed-conifer forests (Cassell et al. 2019). Over the longer term, fire severity will depend on fuel availability (Dillon et al. 2011; Fried et al. 2008; Miller et al. 2009, 2012; van Mantgem et al. 2013), future climate, and the combined effects of forest management and wildfires on future forest landscape patterns, stand structure, and species composition.

Changing Disturbance Regimes Under Climate Change

Fire will interact with other stressors, including drought and insect outbreaks, to affect forests.

Water deficit will increase in a warming climate, contributing directly to lethal stresses in forests by intensifying negative water balances (McKenzie and Littell 2017). Water deficit also indirectly increases the frequency, extent, and severity of other disturbances, especially fire (McKenzie et al. 2004, van Mantgem et al. 2013) and insect outbreaks (Logan and Powell 2009). These combinations of biotic and abiotic stressors will drive shifts in forest ecosystems in the coming climate (McKenzie et al. 2009).

Across forested landscapes, fire directly influences the spatial mosaic of forest successional patches (Agee 1993). More extreme fire weather conditions with climate change are already leading to larger and more frequent fires, resulting in larger burn patches, more open forest conditions, and simpler landscape structure (Cassell et al. 2019). More frequent severe fire will likely decrease overall forest ages, the fractional coverage of old-growth forests, and landscape connectivity of old-growth forest patches (McKenzie et al. 2004). However, more frequent low- and mixed-severity fires can reduce fuels in dry and moist forests, leading to lower intensity fires, more open canopy conditions, and more often clumped and gapped tree distributions (Chmura et al. 2011, Churchill et al. 2013, Larson and Churchill 2012).

In general, increased fire frequency will favor species with life history traits that allow trees or tree populations to persist during and after fire (Agee 1993). These include (1) species that can resist fires (e.g., thick-barked species such as Douglas-fir, western larch [*Larix occidentalis*], and ponderosa pine), (2) species with foliage geometry that dissipates rather than traps heat energy during wildfires, (3) species with high seed-dispersal ability that can establish after fires (e.g., Douglas-fir), (4) species with serotinous cones that allow seed dispersal from the canopy after fires (e.g., lodgepole pine in some areas), and (5) hardwoods that resprout from stumps or roots or readily regenerate from wind- or animal-dispersed seeds (Agee 1993). More frequent fire will decrease abundance of avoider species, including those with thin bark, and slow invaders after fire (e.g., grand fir [*Abies grandis*] and white fir) (Chmura et al. 2011). If fire-sensitive species are unable to reestablish after fire because of short fire-free intervals, competition, or harsh establishment conditions, they can be lost from a site.

Climate Change and the Wildfire Deficit

Fire exclusion during most of the 20th and early 21st centuries drastically reduced burned area, leading to a widespread east-side wildfire deficit. As the climate continues to warm and dry, area burned will sharply increase but level off prior to mid-century because of the area already burned and reburned.

Fire history studies throughout eastern Oregon and Washington (e.g., Hagmann et al. 2013, 2014, 2017, 2019; Heyerdahl et al. 2001, 2008; Johnston 2017; Johnston et al. 2016, 2018; Merschel et al. 2014, 2018; Wright and Agee 2004) demonstrate high historical abundance of lightning and aboriginally ignited wildfires. Such fires maintained significant nonforest area on forest-capable sites in dry, moist, and cold forests (Hessburg et al. 2019) and low surface fuel levels and open canopy conditions in most dry and some moist mixed-conifer forests (Hessburg et al. 2007, Johnston et al. 2018).

Box 9

Increases in Nonforest Area and Transitions of Dry Forests to Savannahs and Grasslands and Moist Mixed-Conifer to Dry Mixed-Conifer Forests

In Pacific Northwest forests, warming climate and changing disturbance regimes will lead to changes in forest structure and species composition (Case et al. 2019, Kerns et al. 2017, Raymond et al. 2014). Within forest stands, more frequent fire will decrease tree density in dry forests, and some dry, moist, and cold forests will be converted to grassland or shrubland as a result of drought stress, increased burn severity, or reburning (Halofsky et al. 2013; Hessburg et al. 2016, 2019; Prichard et al. 2017). Increasing water deficits (fig. 11) will likely increase transitions of moist to dry mixed-conifer forests and contribute to decreased forest growth rates (Restaino et al. 2016), slowing development of large tree structures. Water deficits will likely produce significant and ongoing tree mortality, especially via bark beetle eruptions, but also from other physiological stresses (Bentz et al. 2010, Kolb et al. 2016, Stephens et al. 2018), and mature trees may be more vulnerable than younger trees (Allen et al. 2010, Kwon et al. 2018, van Mantgem et al. 2009).

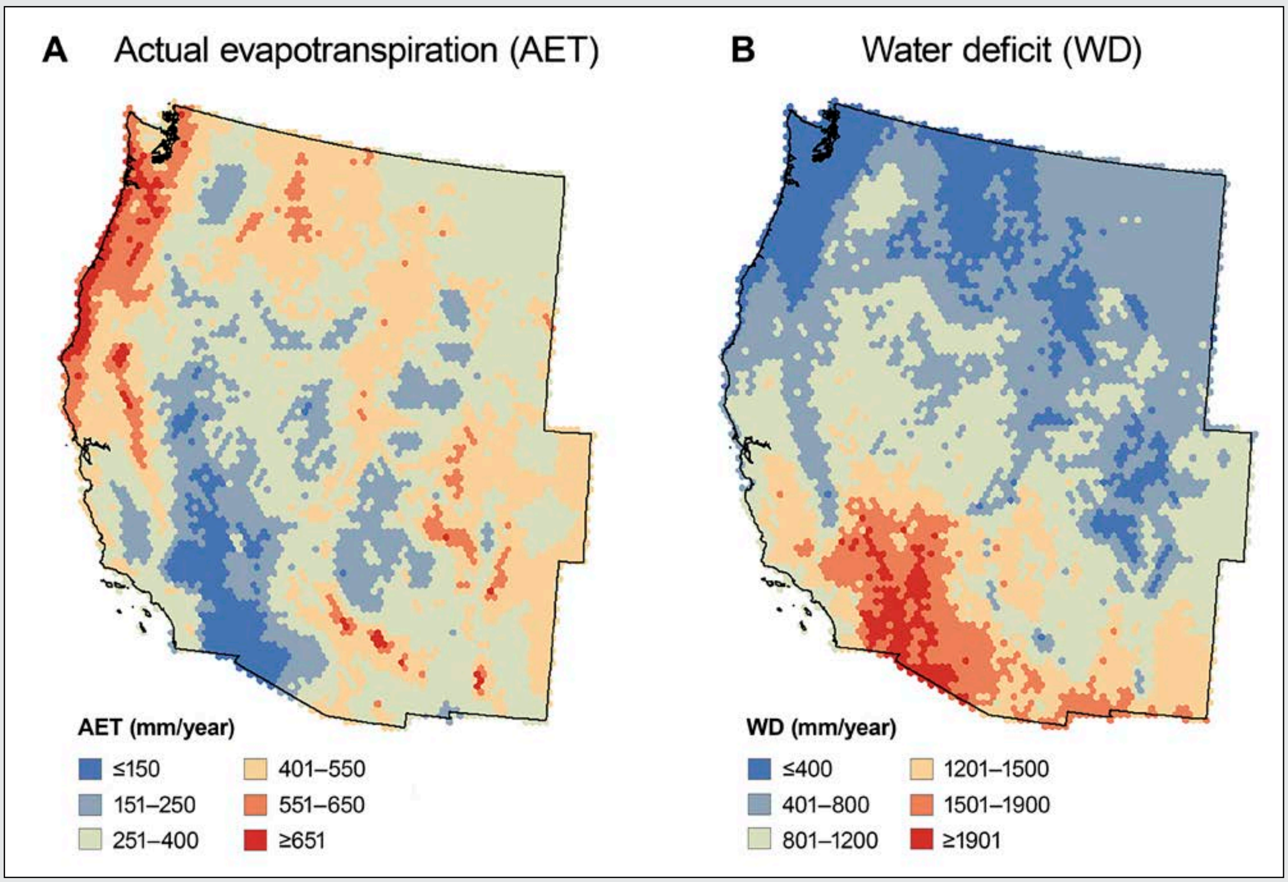

Figure 11—(A) Mean actual evapotranspiration and (B) water deficit per 50 000 ha (123, 552 ac) hexel (hexagonal pixel) for the Western United States from 1984 to 2010. Adapted from Parks et al. (2014).

Wildfire suppression began in earnest in the United States in the early 20th century (1934–1935) with the advent of the "10 a.m. rule" (Dombeck et al. 2004), although some fire history studies noted suppression successes before that time as early as the 1890s and 1900s. Fire suppression efforts today result in effectively dousing 97 to 98 percent of all U.S. fire starts each year (Dombeck et al. 2004, North et al. 2015), and the Western United States remains in a wildfire deficit (Parks et al. 2015). However, the remaining 2 percent of wildfires that escape control typically burn under extreme fire weather conditions in forests with high fuel-loads, accounting for 97 percent of all firefighting costs and area burned (Calkin et al. 2005).

Fire suppression has steadily declined in efficacy since around 1985, as the climate in the West began a prolonged period of warming and drying.

Without a significant infusion of proactive prescribed burning, as in decades past, fire suppression has steadily declined in efficacy since around 1985, as the climate in the West began a prolonged period of warming and drying (Littell et al. 2009, North et al. 2015, Westerling et al. 2006), and acres burned are steadily rising. Predictive models suggest that area burned in the West will double or triple by mid-century (McKenzie et al. 2004, Westerling et al. 2011), and then decline later as fuel availability declines because of the area burned and reburned (McKenzie and Littell 2017, Prichard et al. 2017).

Expanded wildland fire use (syn. managed wildfires) and proactive prescribed burning, coupled as needed with thinning pretreatment (e.g., Prichard et al. 2010), will be useful for improving forest conditions affected by large wildfires (Fernandes and Botelho 2003, Reinhardt et al. 2008); followup maintenance treatments will be needed at regular intervals to maintain effectiveness. Because the land area treatable by prescribed burning is restricted by operational, social, and ecological issues (Schultz et al. 2018), a fire and fuel management strategy based primarily on thinning and prescribed burning will have a slim chance of succeeding (Schoennagel et al. 2017, Schultz and Moseley 2019). The increasing incidence of human ignitions (Balch et al. 2017) and exurban development in forests greatly constrains prescribed fire use (Schultz and Moseley 2019), suggesting more strategic use of prescribed burning to protect the WUI and to establish and harden anchor point and control locations to predate expanded use of managed wildfires under moderate fire weather conditions (Prichard and Kennedy 2014, Thompson et al. 2016, Wei et al. 2019).

Box 10

The Fire Deficit Will Diminish, But Resulting Conditions Will Often Be Uncharacteristic for the Forest Types

The effect of increasing burned area will likely diminish the fire deficit in time, but fire effects resulting from the large area burned under extreme fire weather events will little resemble the more characteristic fires of the pre-Euro-American settlement era (North et al. 2015). Likewise, forests resulting from escaped large wildfires will bear little resemblance to the forests that evolved under a more characteristic wildfire regime (Hessburg et al. 2005, Keane et al. 2002).

Future Considerations

> Fuel treatments, including managed wildfire, prescribed burning, and forest thinning coupled with prescribed burning, can decrease fire severity locally and regionally. To have an impact, treatment extent will need to be enough to tip dynamics in favor of more beneficial future fires.

More frequent and larger wildfires in east-side forests will be the major challenge facing resource managers. Managers will not affect broad trends in increasing area burned resulting from climatic changes, but fuel treatments (including mechanical treatments and wild and prescribed fires) can decrease fire severity in local and regional landscapes, where their extent is sufficient to tip dynamics in favor of more benign fire behavior and effects (Ager et al. 2010, 2013, 2020).

In dry and moist east-side mixed-conifer forests, especially in the drier topoedaphic settings (e.g., south slopes and ridgetops), reducing smaller tree density and layering can decrease negative effects of drought on tree growth and the likelihood of severe fires (Halofsky et al. 2019a, Sohn et al. 2016). Decreases in forest density, coupled with hazardous fuel treatments, can also increase forest resilience to fire (fig. 12) (Agee and Skinner 2005) and mitigate losses to large tree structures (Halofsky et al. 2014). Fuel treatments maintained over time will be more effective (Agee and Skinner 2005). Such methods are not needed everywhere for wildfire and climate adaptation. For example, moist valley bottom settings and cool-moist north-facing aspects would ordinarily support denser and more layered forest conditions under the native fire regime; a consequence of reduced fire frequency (Merschel et al. 2014). These would be areas of complex forest that would burn severely when touched by fire and are examples of climatic and wildfire refugia, some of which can provide future old-forest characteristics and habitats for associated species.

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John Marshall

Figure 12—Widespread encroachment of (A) small- (9 to 15.9 inches diameter at breast height [dbh]) and (B) medium-size (16 to 24.9 inches dbh) Douglas-fir and grand fir into once-open canopy (primarily) ponderosa pine and Douglas-fir (secondarily) stands. Most trees in the shown area have filled in during the 20th-century period of fire exclusion. In both photos, note the live and dead fuel ladders that accompany this conifer encroachment.

Silviculture Research: Stand Development and the Role of Large Trees

Resistance, Resilience, and Landscape Heterogeneity

> Silvicultural methods can aid in reducing stand density, increasing the abundance of early-seral tree species, protecting and promoting large-tree recruitment and survival, increasing patch and landscape spatial heterogeneity, and improving landscape resistance and resilience in the face of escalating disturbances and climate change.

Forest restoration objectives for the east side commonly include increasing landscape resilience to fire, insects, drought, and other disturbances, and increasing landscape heterogeneity at multiple spatial scales. Objectives also include the need to maintain critical wildlife habitats, and restore underrepresented vegetation types, such as old forests dominated by early-seral tree species, open forest, and early-seral conditions (Franklin and Johnson 2012, Hessburg et al. 2016, Lehmkuhl et al. 2015, Stine et al. 2014). These landscape-level restoration objectives are typically achieved through stand- or patch-level silvicultural and fuels management treatments that seek to modify forest structure, species composition, and forest fuels.

Growing and retaining large live trees of early-seral species (e.g., ponderosa pine, western larch) is critical for achieving many restoration objectives (Franklin and Johnson 2012, Franklin et al. 2018). With their thick bark and elevated canopies, large, early-seral trees are typically more resistant to fire, insects, and drought than smaller and shade-tolerant tree species. Large trees concentrate stand biomass and carbon and often contain a high proportion of above- and belowground biomass in a stand. By resisting the effects of disturbances, large early-seral trees provide a persistent local seed source that promotes forest regeneration when favorable climate and establishment sites allow. When they die, large early-seral trees generate large snags and fallen logs that provide important wildlife habitat and persist longer than smaller snags and logs and many large dead shade-tolerant trees.

Large Versus Old Trees

Old trees, even smaller ones, have high ecological values.

Trees more than 150 years old are known to be more valuable for forest restoration and wildlife conservation than simply large trees.

Older trees—those more than 150 years old—are known to be more valuable for forest restoration and wildlife conservation than simply large trees. Old trees develop unique morphological and pathological traits (Castello et al. 1995) that provide additional wildlife habitat features. Having stood the test of time, they are likely to withstand future disturbances and add to forest genetic diversity. Old trees of early-seral species also provide information about historical forest density, forest patch sizes, tree spatial patterns, and stand dynamics (Larson and Churchill 2012). Recent research shows, however, that old trees are not always large, and large trees are not always old (Brown et al. 2019, Van Pelt 2008); selection of old trees for retention based on field identification of morphological traits and site environments has been shown to be effective in addition to selection based on diameter alone (Riling et al. 2019, Van Pelt 2008). Some current diameter limits require retaining large trees, regardless of age, shifting harvest to smaller trees (including old smaller trees) to generate revenue. Combining age- and size-based metrics to retain adequate densities of large trees along with old trees featuring desirable traits could allow younger large trees to be managed more flexibly, depending on stand conditions and local restoration objectives.

Box 11

Silviculture for Wildfire and Climate Adapting Dry and Moist Mixed-Conifer Forests

Reducing stand density, altering species composition to promote increased dominance of early-seral tree species, promoting development of larger size trees, and increasing stand and landscape heterogeneity are some of the primary silvicultural objectives for improving landscape resilience to wildfires and climate change in many east-side forest landscapes (Franklin and Johnson 2012, Stine et al. 2014). Reducing stand density and increasing dominance of large trees of fire-resistant species helps to increase forest resistance (Agee and Skinner 2005) and resilience (Hessburg et al. 2019) to fire and, along with increasing spatial complexity, may increase resistance and resilience to drought and insect disturbances (Fettig et al. 2007, Sohn et al. 2016). Shifting species composition toward increased dominance by early-seral tree species, such as ponderosa pine and western larch, may be important where past harvesting practices or forest succession have largely eliminated these species from stands or landscapes on which they would characteristically dominate (Stine et al. 2014).

Some recent retention and removal strategies to promote climate- and wildfire-adapted dry and moist mixed-conifer forests could include the following (adapted from Hessburg et al. 2005, 2016; Johnston et al. 2018; Merschel et al. 2018; North et al. 2009a, 2009b; Stine et al. 2014):

- Retaining large-size western larch, ponderosa pine, western white pine, and sugar pine, and recruiting more trees into these size classes
- Retaining old (>150 years old) trees of these same species, regardless of size
- Retaining old and overmatured (>200 years old) grand fir, white fir, and Douglas-fir (>250 years) as habitat trees, especially those with well-developed heartrot, e.g., *Echinodontium tinctorium,* rust red stringy rot, in grand and white fir (fig. 13); *Laricifomes officinalis,* brown trunk rot, *Porodaedalea pini*, red ring rot (fig. 13); *Phaeolus schweinitzii,* brown cubical butt rot, or *Fomitopsis cajanderi*, rosy top rot, in Douglas-fir; and other obviously significant bole or very large branch defects, e.g., massive mistletoe (*Arceuthobium douglasii*) brooms (fig. 14)
- Retaining Douglas-fir, white fir, and grand fir forest patches in relatively dense and layered arrangements, especially on the moistest mixed-conifer sites
- Rebuilding representation of ancient tree cohorts (>400 years) of early-seral species
- Removing medium- and large-size grand fir and white fir where they are growing in direct competition with preferred and retained medium, large, and old ponderosa, western white, or sugar pine and western larch
- Removing medium- and large-size and smaller grand fir, white fir, and Douglas-fir, where they interfere with wildfire and climate adaptation goals to reconstitute early-seral ponderosa pine, western white pine, sugar pine, western larch, or hardwoods (fig. 15).

Examples like these can be further shaped by local knowledge, expertise, and site-specific objectives.

Box 12 shows example forest plan components based on retention and recruitment of live, large, and old early-seral trees. Another plan component can address retention and recruitment of dead, large, and old early-seral trees.

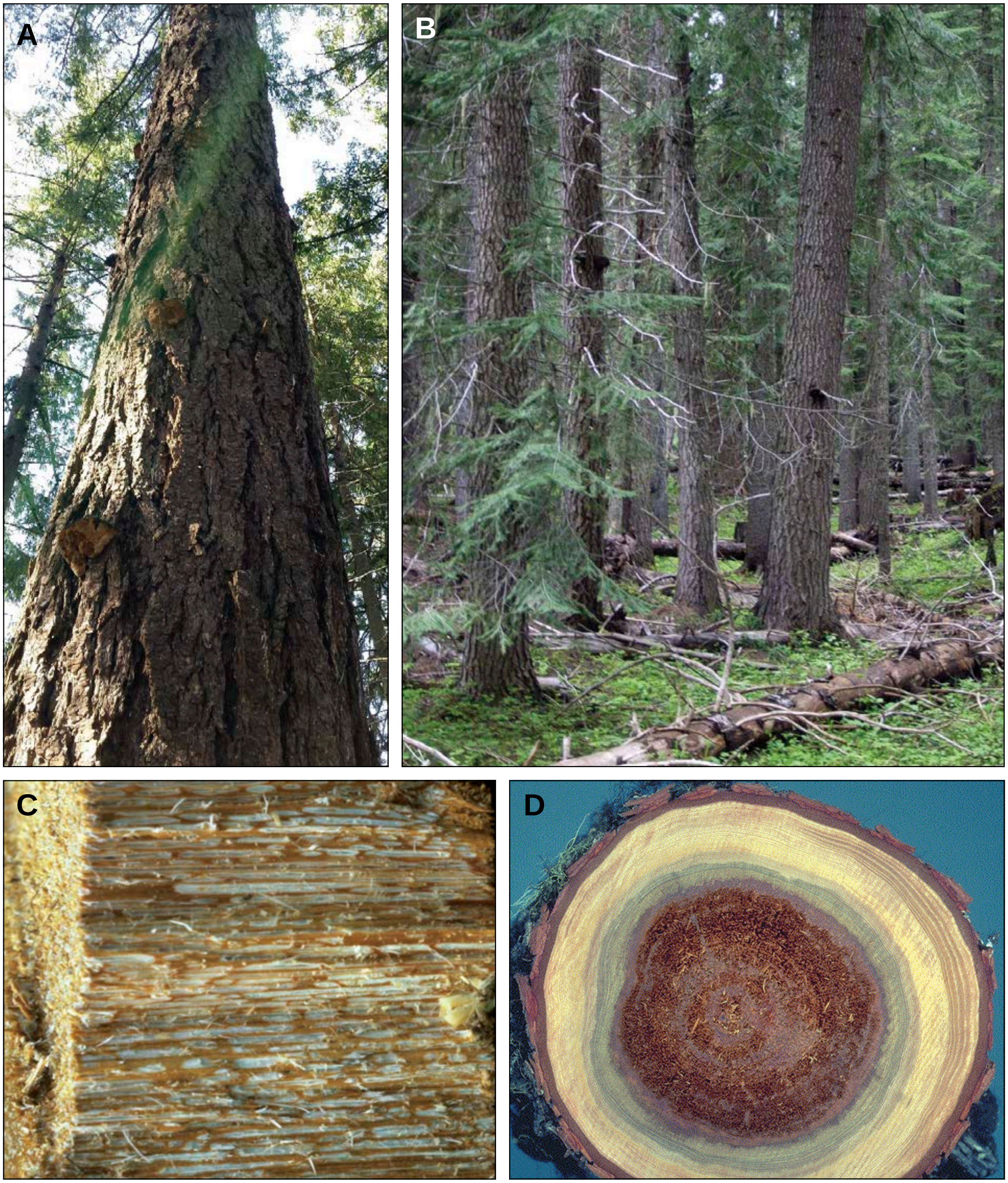
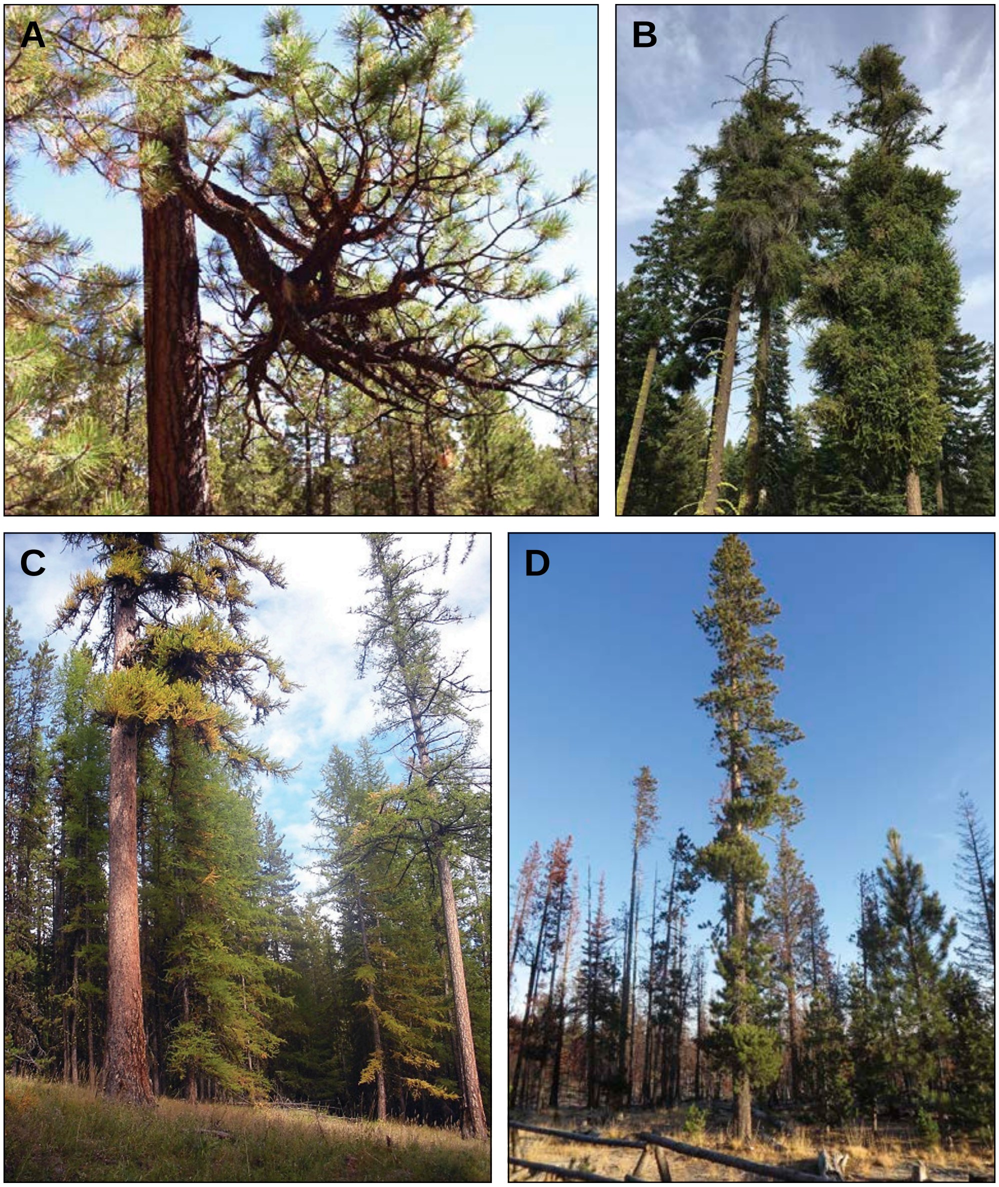


Figure 13—Heartwood decay softens the heartwood of living trees, making them suitable for cavity excavation by a variety of wildlife species. Broken-topped trees with heartrot make suitable nesting platforms as well. (A) Red ring rot and conks caused by *Daedalea pini*. (B) Rust red stringy rot and conks of immature grand fir cause by the heartrot pathogen *Echinodontium tinctorium*. (C) Typical heartwood decay of *D. pini*. (D) Typical wood decay associated with *E. tinctorium*. Many grand fir—especially when shaded as young saplings or poles—develop dormant infections in vascular traces of juvenile needles and in lateral branchlets of the interior crown that are later enveloped by heartwood. These quiescent infections release from dormancy when trees are injured (Etheridge and Craig 1976). This defect is common today in stands that were selectively harvested for their overstory ponderosa pine or western larch, and the injured grand fir were left to grow as replacement trees.



Jim Worrall

https://wyeastblog.org/2019

http://coniferousforest.com

Michelle Agne

Figure 14—Massive dwarf mistletoe brooms in (A) ponderosa pine, (B) Douglas-fir, (C) western larch, and (D) lodgepole pine, caused by the host-specialized parasitic seed plants *Arceuthobium campylopodum, A. douglasii, A. laricis,* and *A. americanum*, respectively. Mistletoe brooms make excellent nesting platforms for a large variety of birds and small mammals.

Figure 15—Conifers encroaching on aspen clones. Ultimately, the conifers grow taller and shade out aspens, which begin to decline in vigor after 55 to 75 years. Aspen clones will release if conifers are removed in time and older aspen are felled or killed by fire.

Box 12

Example Forest Plan Components to Guide Restoration and Retention of Large and Old,[7] Live, Early-Seral Trees

Desired condition: The abundance and spatial arrangement of large and old, live, early-seral trees is within the natural range of variability (NRV) for mid-21st century climate (e.g., considering RCP 4.5 and 8.5 GHG emission scenarios) at patch, local landscape, and forest levels.

Standard: Large and old, live, early-seral trees (western larch, ponderosa pine, western white pine, and sugar pine) are retained and recruited.

Guideline: Management activities will retain and emphasize recruitment of large and old, live, early-seral trees across the landscape. Exceptions in which individual large or old, live, early-seral trees may be removed can include the following:

- Trees that pose an imminent threat to public safety
- Trees removed to facilitate an emergency response
- Cases in which the abundance of large and old, live, early-seral trees exceeds the NRV for the mid-21st-century climate at each of the three levels

[7] Old trees are identified using the criteria of Van Pelt 2008.

Early-Seral Tree Species

Silvicultural treatments such as thinning and prescribed burning can help increase underrepresented early-seral species.

Where large trees of early-seral species are underrepresented in east-side forests, because of past harvesting or fire exclusion, active management can support restoration objectives. Severe fires or regeneration harvests may be needed (Spies et al. 2018a) to remove local seed sources and competition from undesirable shade-tolerant trees to help certain patches of forest adapt to fire and climate change. Under these conditions, successional trajectories after disturbance will be more likely to lead to stands dominated by shade-tolerant tree species than under historical fire regimes. Removal of shade-tolerant tree species may be followed by planting to establish local populations of the desired early-seral tree species. Following establishment, individual tree growth can be accelerated through active density management (Cochran and Barrett 1998, 1999).

Where large trees of early-seral species are underrepresented in east-side forests, because of past harvesting or fire exclusion, active management can support restoration objectives.

If the desired early-seral tree species are already present, thinning treatments can be used to reduce stand density, promote the growth and retention of large early-seral trees, and increase spatial tree clump and gap complexity within stands. Where increasing resilience to fire and other disturbances is the primary management objective, thinning treatments typically focus on removing smaller shade-tolerant trees and reducing canopy fuels. Removal of some young large trees may also be needed to achieve stand density index targets or permit retention of a broader range of tree sizes. Spatial complexity can be enhanced during thinning through marking procedures that promote generation or preservation of openings, open-grown individual trees, and clumps of trees (Churchill et al. 2013, Larson and Churchill 2012). Prescribed burning treatments are commonly used to reduce surface fuels and understory regeneration, but may require prior overstory thinning or understory vegetation modification treatments to be effective and safe.

Although removal and suppression of shade-tolerant tree species is an important management objective across a large portion of east-side forests (fig. 16), these species are important for generating multilayered forest structures that were historically found in fire refugia (Camp et al. 1997, Krawchuk et al. 2016), and that currently support important wildlife species like the northern spotted owl (*Strix occidentalis caurina*) and its prey species (Lehmkuhl et al. 2006, 2015), or the northern goshawk (*Accipiter gentilis*). In many cases, large and old, early-seral tree species formed the overstory component of these old forests, and smaller shade-tolerant tree species formed the layered understories.

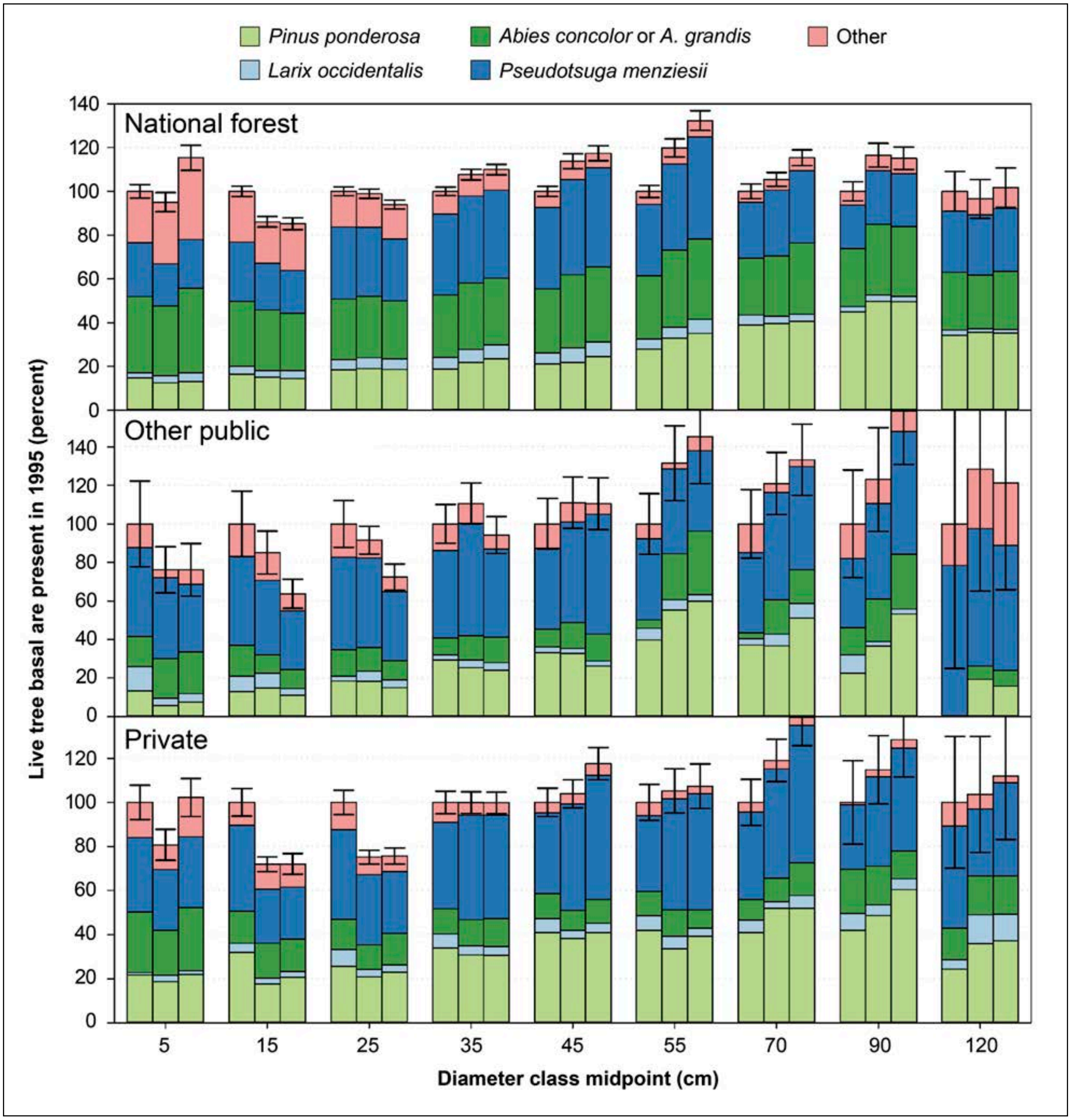


Figure 16—Changes in species abundances in mixed-conifer forests of eastern Oregon and Washington. Forest inventory and assessment data (FIA) show basal area per hectare (BAH) by landowner group. For each diameter class, bars from left to right represent estimates for midpoint inventory years 1995, 2004, and 2014. Species include the fire-tolerant *Pinus ponderosa* and *Larix occidentalis*, the shade-tolerant *Abies concolor* or *A. grandis* and *Pseudotsuga menziesii*, and all other remaining coniferous species combined (Other). Species proportions are relative to the totals in 1995 for each diameter class. Results indicate overall increases in BAH greater than 40-cm diameter, with *Abies concolor* or *A. grandis* and *Pseudotsuga menziesii* increasing more than *Pinus ponderosa*, and *Larix occidentalis* generally declining, particularly on national forest lands. Total BAH for trees >40 cm diameter in 1995 differed by owner, with 10.6, 9.5, and 6.2 m2/ha on national forest, other public, and private lands, respectively.

It is well known that frequent fires lead to low-density stands of medium- and large-size ponderosa pines and other fire-tolerant tree species on dry sites. However, recent research shows that frequent fire also occurred on moister sites where it created low-density stands of large, old ponderosa pine and Douglas-fir (Hessburg et al. 2007; Johnston et al. 2016, 2018; Wright and Agee 2004). Although shade-tolerant trees were somewhat more common on moist mixed-conifer than dry mixed-conifer sites, the fire regimes were often quite similar, and exclusion of fire has led to a dramatic increase in density of shade-tolerant trees there as well (Everett et al. 2000; Heyerdahl et al. 2001, 2008; Johnston et al. 2016, 2018; Merschel et al. 2014, 2018). Thus, removal of shade-tolerant trees is not just a restoration issue on dry mixed-conifer but also on moist mixed-conifer sites (Hessburg et al. 2016, Stine et al. 2014)**.**

Box 13

Fire Refugia—Key Components of Fire-Prone Landscapes

Fire refugia reside in unique topoedaphic settings that increase the likelihood that a patch of forest will avoid fires. The refugial setting **and** the surrounding forest and fuel conditions work together to increase the fire-free interval and reduce the likelihood of high-severity fires within refugia (Camp et al. 1997). Fire refugia are often late-successional or old-forest habitats, and they provide seed rain to regenerate new patches of forest after wildfires (Downing et al. 2019). In large high-severity patches, this effect is likely smaller but present. Active silvicultural treatment is not recommended for high-value habitat areas such as fire refugia but may be important in surrounding stands to help spatially isolate high-value habitats from wildfire, or to limit fire flow to that of low-severity fires (Lehmkuhl et al. 2015). Low-density stands with large, early-seral trees can potentially serve as "habitat-in-waiting," as they can develop multilayer characteristics within a few decades to replace more complex habitat lost to wildfire (Lehmkuhl et al. 2015).

Old-Growth Definitions

Interim old-growth definitions based on stands that were a byproduct of fire exclusion may be inadequate for east-side, frequent fire, ponderosa pine, dry and moist mixed-conifer forests.

Definitions of old growth can be confusing, especially in east-side forests where fire controls succession and the structure and composition of forests (Spies et al. 2018b). Where fire has been absent for many decades or centuries, late-successional forests dominated by shade-tolerant and fire-intolerant species often develop. Where fire is more frequent and less severe, forests are dominated by older, shade-intolerant and fire-tolerant species. The latter are typically termed old growth or old forest, while the former are often termed "late-successional." However, people sometimes use these terms interchangeably, which can cause confusion.

For east-side forests in Oregon and Washington, interim definitions of old growth developed 25 years ago were based on the number of large trees and snags, the amount of down woody debris, the number of tree canopy layers, and other components (Merschel et al. 2019). Large tree density was a critical component of definitions. These interim definitions were based on the structure of existing stands of older trees that were the byproduct of decades of fire exclusion, i.e., late-successional

If diameter limits apply to all tree species, it may not be possible to restore forest stands and landscapes.

Box 14

Diameter Limits

Applied to all tree species, diameter limits may not be effective at meeting restoration objectives and may even be counterproductive.

If diameter limits apply to all tree species, it may not be possible to restore forest stands and landscapes. This will be especially true where fire exclusion and past selection cutting have allowed shade-tolerant tree species to grow beyond the diameter limit. Such trees can be moderately fire resistant when large, provide a persistent seed rain for maintaining a local tree population that competes with early-seral tree species for site dominance, and limit fuel treatment effectiveness and longevity. In the Blue Mountains, for example, a recent study found that the abundance of large trees was at least as great today as in the 19th century, but the proportion of large trees of shade-tolerant species had increased (Johnston et al. 2018) (fig. 17). Such trees may not be desirable from a restoration perspective but cannot be removed (except by forest plan amendment) because they exceed the current 21-inch diameter threshold.

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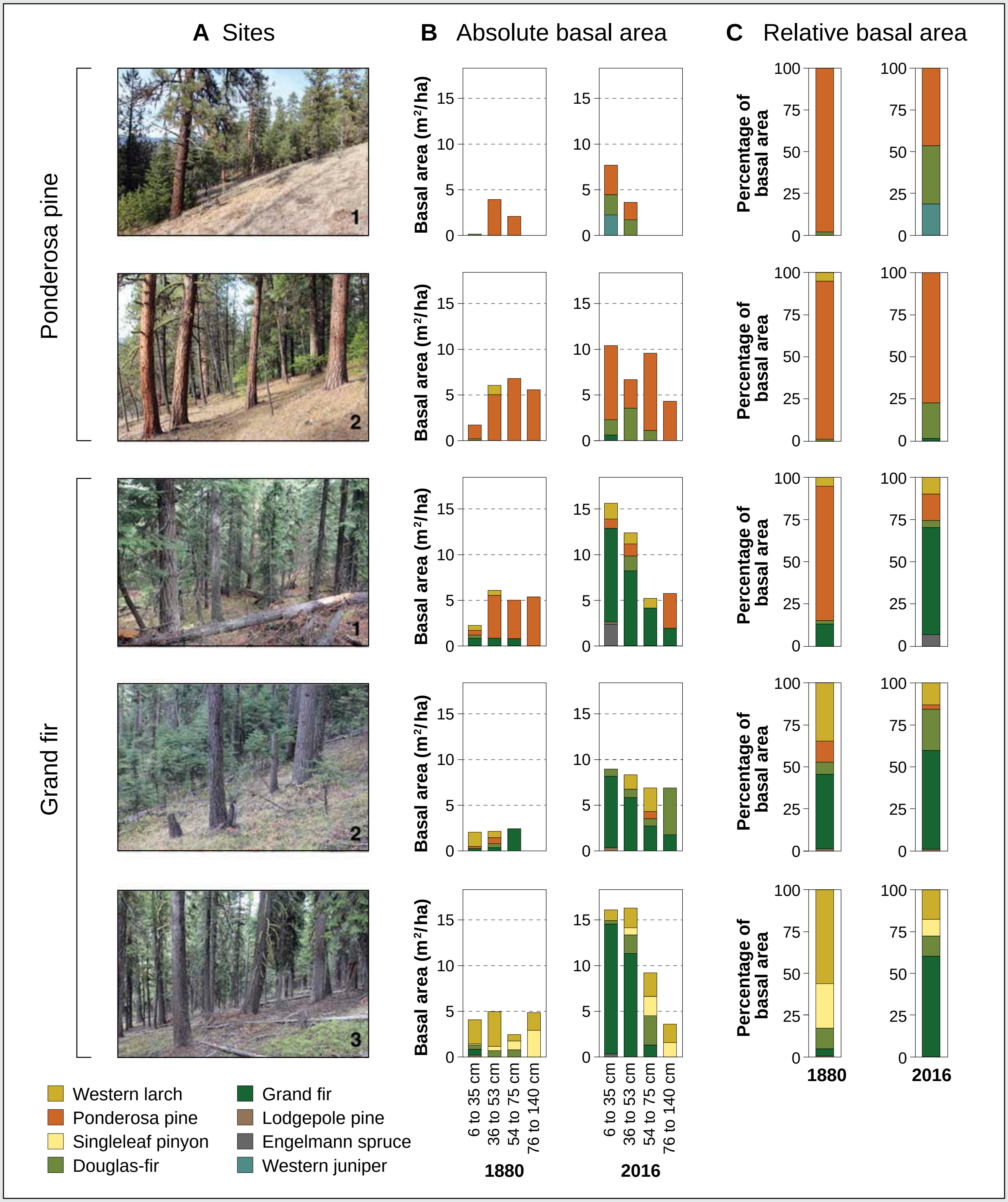


Figure 17—Basal area of reconstruction plots in 1880 and 2016. (A) Two ponderosa pine and three grand fir sites, respectively (each site consisted of three plots); (B) basal area in 1880 (left) and 2016 (right) in four diameter categories; and (C) species basal area, shown as a percentage of total basal area in 1880 (left) and 2016 (right). As examples, photos are somewhat unrealistic (lower density than actual) to allow a view into stands. Illustration adapted with permission from Johnston et al. 2018.

forests. Consequently, they may not be applicable to the goal of restoring old-growth conditions that developed under frequent-fire regimes of dry and moist mixed-conifer forests, or for enhancing resilience to drought and other stressors (Merschel et al. 2019, Spies et al. 2018a). New old-growth definitions have been suggested for east-side, frequent-fire, ponderosa pine, dry mixed-conifer, and moist mixed-conifer forests (Spies et al. 2018a). These definitions are best based on knowledge of historical stand and landscape structure and composition, with likely adjustment needed for climate change adaptation.

Large-tree and old-forest effects on microsite conditions—
Relatively closed-canopy old forests can modify the microclimate of sites they occupy. This can be important to plants and animals that thrive under these conditions.

Large trees and old forests often provide an insulating effect on environments compared to younger forests, reducing maximum springtime air temperatures near the ground by as much as 4.5 °F (Frey et al. 2016). Variable-intensity restoration thinning of older ponderosa pine stands followed by prescribed burning shows distinct patch-level variation of understory microclimates (Burnett and Anderson 2019). This effect on understories can be important for certain plants and animals that thrive in cool and moist environments (Muscolo et al. 2014).

Nonnative or invasive plant species are also found less often and with lower cover and constancy in stands with larger tree cover (Gray 2005, McIver et al. 2013). Fuel and restoration treatments can promote invasive plants, though the timing of burning, cover type, and pretreatment abundance are important factors (Kerns and Day 2017, Kerns et al. 2020, McIver et al. 2013).

Contributions of Large Trees to Biodiversity and to Aquatic and Terrestrial Habitats

Biodiversity

> Protection of old-forest patches and large, old-tree legacies has long been recognized as a means of conserving biodiversity in the Pacific Northwest.

Large, old trees make important contributions to the structural and spatial diversity of forests, which in turn provide for the development and persistence of diverse plant, fungal, lichen, and animal communities (Lindenmayer and Franklin 2002, Marcot et al. 2018). The Interagency Special Status/Sensitive Species Program list includes 425 species found on east-side national forests, including 257 vascular and 44 nonvascular plants, 6 fungi, 65 invertebrate, and 63 vertebrate animals. Of the vertebrates, 48 are terrestrial, including 27 birds, 15 mammals, 4 amphibians, and 2 reptiles, and many are associated with large, old trees (https://www.fs.fed.us/r6/sfpnw/issssp/agency-policy/). Despite emphasis placed on addressing sensitive species information gaps in previous planning assessments, the ecology, distribution, and status of many rare and sensitive species remains poorly known (Marcot et al. 2018).

Wildlife Habitat Functions

> Large trees provide food, security, shelter, and thermal refugia for forest wildlife.

Large trees contribute a variety of habitat functions for forest wildlife, including food, shelter, and security from predators or competitors (Brown 1985, Bull et al. 1997, Johnson and O'Neil 2001, Thomas 1979). For example, large trees provide food resources, including cones and nuts (e.g., ponderosa pine and Douglas-fir seeds), arboreal lichens, and foraging substrates for insectivores (e.g., Gaines et al. 2007, 2010), foliage and bark gleaners (Lyons et al. 2008), and are frequently associated with productive fungal communities (e.g., truffles and mushrooms) (Carey et al. 2002, Lehmkuhl et al. 2004). Large trees provide shelter for wildlife, including critical nesting, resting, and denning structures in the form of cavities, platforms, and exfoliating bark (Bull et al. 1997). These structures also provide thermal refugia that are enhanced by the deep shading and cool-moist microclimates provided by large trees with complex canopy structures.

Box 15

Role of Diseases and Defects of Large, Old Trees

As trees age, they increasingly host an array of forest diseases. Wood decay pathogens and the defects they create play a vital role in the development of wildlife habitat components of large trees.

Forest diseases play a critical role in the development of wildlife habitat in large and old trees. The older a tree, the more likely that forest diseases are increasing in influence, especially when root, bole, or top injuries have occurred over the life of a tree (Castello et al. 1995). Injuries can include those occurring during prior timber harvests, wildfires, and weather or climatic events. Bull et al. (1997) identified five distinct large tree structures that provide unique habitat values: (1) living trees with decay (such as internal heartrot) (fig. 13), (2) hollow trees, (3) trees with brooms (misshapen branches) (fig. 14), (4) snags, and (5) fallen logs. Cavities are particularly important structures that provide nesting, resting, and denning functions for animals ranging from bats to bears (Mellen-McLean et al. 2017). Primary cavity excavators (e.g., woodpeckers) play a fundamental ecological role by creating tree cavities they use that are later exploited by a variety of other users, including many birds and small mammals.

Large trees also provide security functions for wildlife in the form of screening vegetation, vertical escape structures, and horizontal canopy connections for canopy dwellers (e.g., northern flying squirrels [*Glaucomys sabrinus*]) (Gaines et al. 2010) (fig. 18). Gaines et al. (2007, 2010) found that when medium- and large-size trees were retained in a manner mimicking historical tree patterns, there were positive responses by many bird species. When large trees die, they also provide habitat functions for animals and plants. The website DecAID (https://apps.fs.usda.gov/r6_decaid/views/index.html) provides information on the functions of dead trees in Oregon and Washington forests.

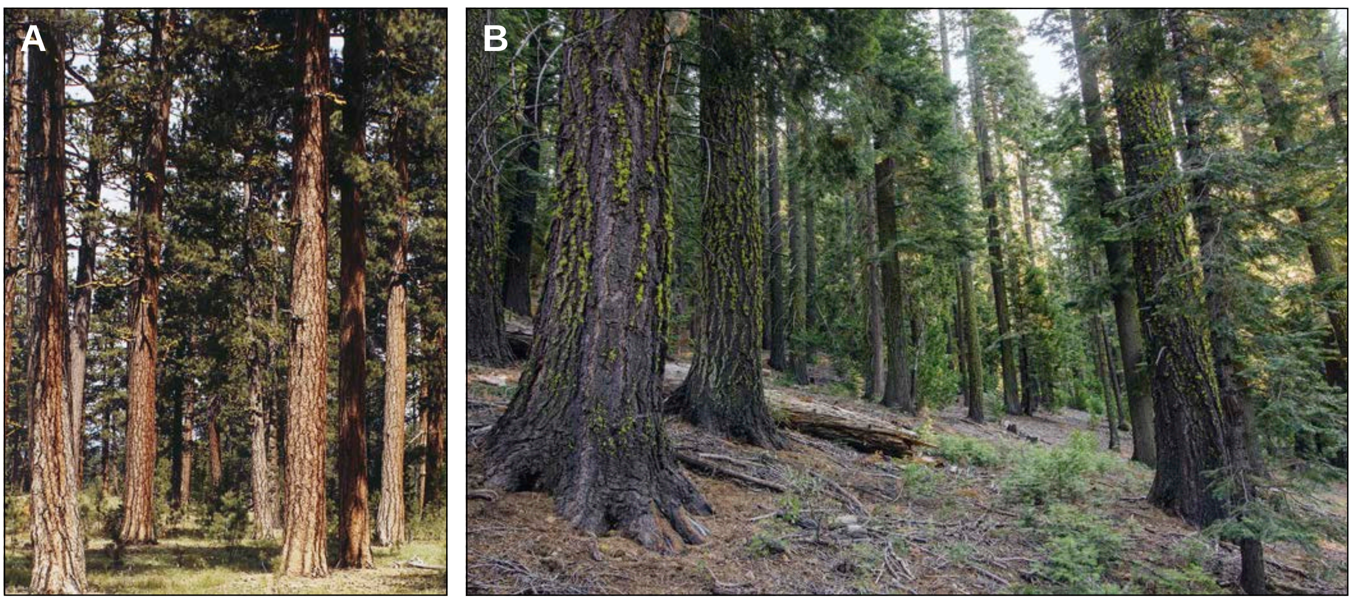


Figure 18—Large trees provide screening vegetation, vertical escape structures, and horizontal canopy connections for canopy dwellers. (A) Stand of virgin ponderosa pine, 1942, Malheur National Forest, Grant County, Oregon. (B) Douglas-fir old forest that provides suitable nesting and roosting habitat for northern spotted owl and goshawk.

Both tree size and interior wood decay processes contribute to the suitability of a tree for cavity excavation and nesting (Lorenz et al. 2015, Raphael and White 1984). Large hollow trees and logs 20 to 80 inches dbh can provide unique denning and resting structures for larger animals, including lynx (*Lynx canadensis*), wolverines (*Gulo gulo*), black bears (*Ursus americanus*), American marten (*Martes americana*), and fisher (*M. pennanti*) (Witmer et al. 1998). Dwarf mistletoe brooms provide shelter and visual screening from predators when large, old trees with cavities are limited. Host-specialized dwarf mistletoes occur on most Pacific Northwest conifers; the older the infections, generally the more massive the brooms (fig. 14). For example, out of 276 northern spotted owl nests documented in the Cle Elum spotted owl demographic study area, 90 percent were on platforms formed by large Douglas-fir dwarf mistletoe brooms (Sovern et al. 2011).

Owing to disease processes, older large trees frequently have more desirable structural habitats for wildlife than younger trees of similar size (Gaines et al. 2007, 2010; Van Pelt 2008). Moreover, early-seral species like ponderosa pine, Douglas-fir, sugar pine, and western larch tend to have both sound wood and wood-decay characteristics that are favorable to development of these values. Late-seral species like grand and white fir have a “live fast, die young” growth strategy that, under certain circumstances, can help them achieve large sizes faster than early-seral species, but comparatively they do not live as long, and snags and logs decay faster (Van Pelt 2008: 143). However, when large and old early-seral trees are unavailable, late-seral grand or white fir may provide the only available large tree habitat structures.

Owing to disease processes, older large trees frequently have more desirable structural habitats for wildlife than younger trees of similar size.

Biological Legacies

> Large trees of early-seral species are typically more resistant to fires and can persist for centuries as important wildlife habitat legacies.

Large trees of early-seral species (ponderosa pine, western larch, Douglas-fir, and in some locations, western white pine and sugar pine) can provide biological legacies that contribute wildlife habitat functions spanning numerous disturbance events, forest development stages, and centuries (Thomas 1979). For example, large trees that persist through disturbance events, such as mixed-severity fire or timber harvest, can provide unique structures in those postdisturbance communities, and influence successional pathways that accelerate development of late-successional habitats (Thomas 1979). In this manner, large trees provide habitat components for both disturbance-adapted and old-forest-dependent wildlife species.

Findings From Population Viability Assessments

> Population viability of species associated with large- and old-tree habitats has declined.

Several wildlife population viability assessments have been undertaken over the past three decades to assess the relationship between live, large-tree forest habitats and wildlife population viability in east-side forests (Gaines et al. 2017, Lehmkuhl et al. 1997, Wisdom et al. 2000). Lehmkuhl et al. (1997) conducted an opinion survey for population viability analysis among wildlife biologists associated with the Interior

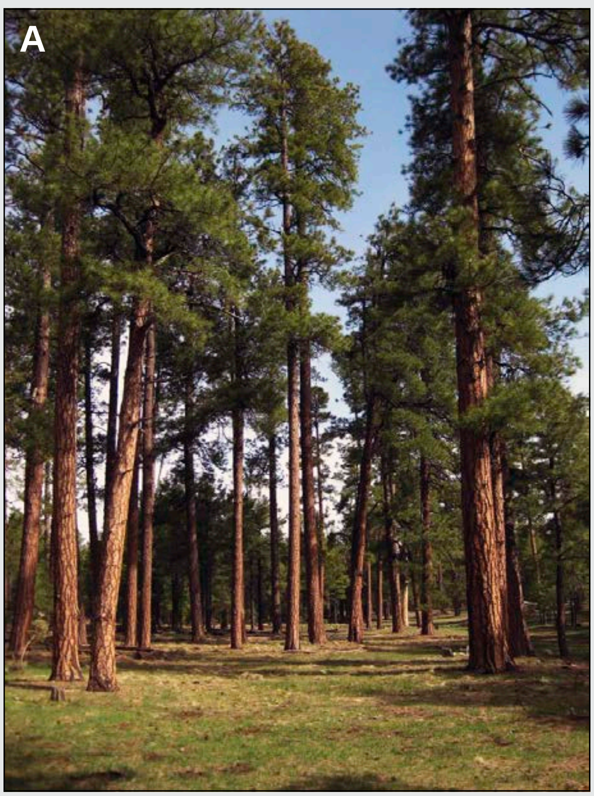
Box 16

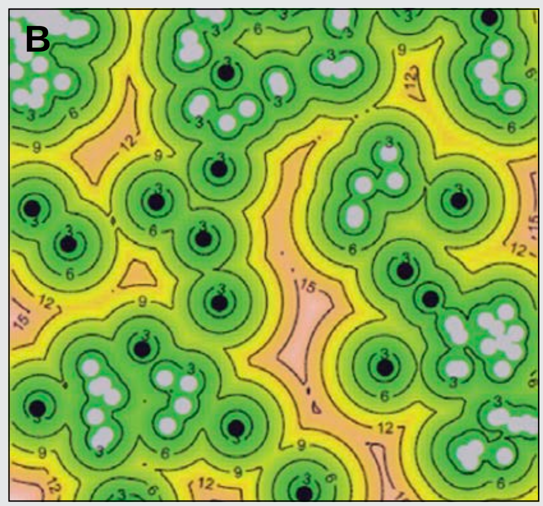
Importance of Large-Tree Spatial Patterns

Spatial patterns of large and old trees determine their resistance to wildfires, resilience in the face of repeated disturbances, and suitability as habitat.

Recent science reviews highlight the importance of spatial configuration at multiple landscape scales for ecological functions of large trees (Hessburg et al. 2015, 2019; Lehmkuhl et al. 2015; Lesmeister et al. 2018; Marcot et al. 2018; Stine et al. 2014). Spatial heterogeneity in the distribution of large trees, characterized by individuals, clumps, and openings (sensu Churchill et al. 2013, Larson and Churchill 2012) (fig. 19) has important implications for ecological functions of large trees and their resilience to wildfires and other disturbances (Churchill et al. 2013, Larson and Churchill 2012). For example, both large trees and a clumped forest pattern were found to be important habitat characteristics for white-headed woodpeckers (Latif et al. 2015). At a meso scale, patterns of fragmentation, isolation, and connectivity between patches that can provide a varied suite of forest wildlife habitat components determine the overall ability of a landscape to support viable populations of sensitive wildlife species (Gaines et al. 2007, 2010, 2017).

Derek Churchill





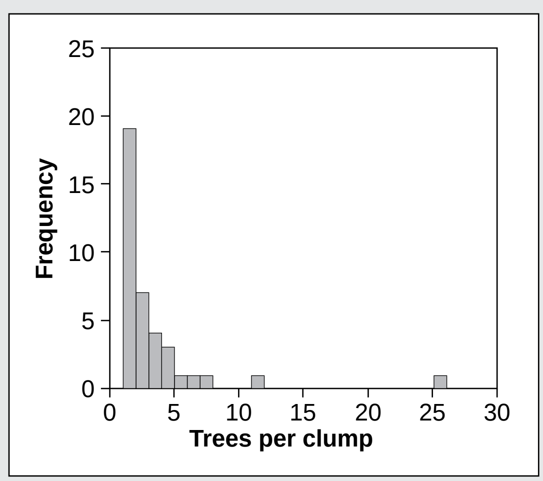


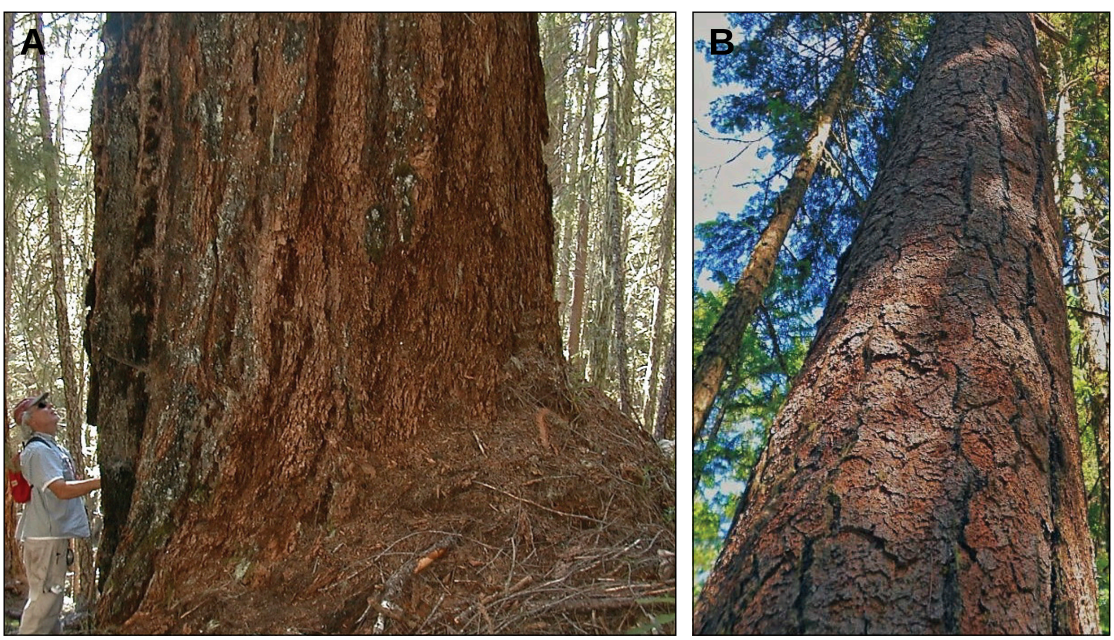
Figure 19—(A) Naturally clumped and gapped patterns of ponderosa pine in Oregon. (B) Example of a frequency-size distribution of ponderosa pine in a sampled plot; adapted from Larson and Churchill (2012).

Columbia Basin Ecosystem Management Project (ICBEMP). They highlighted that large early-seral tree habitat for nearly all species was more favorable under historical conditions than at the time of the assessment. Wisdom et al. (2000) further conducted a systematic assessment of source wildlife habitats for the ICBEMP. They found that species favoring low-elevation, early-seral old-forest habitats (e.g., white-headed woodpecker [*Dryobates albolarvatus*], white-breasted nuthatch [*Sitta carolinensis*], pygmy nuthatch [*S. pygmaea*], Lewis' woodpecker [*Melanerpes lewis*], and western gray squirrel [*Sciurius griseus*]) had experienced the greatest decline of any of the groups analyzed. They stressed that broad-scale loss of large (>21-inch) early-seral trees and snags was the most important issue for these species (Wisdom et al. 2000: 73). Likewise, they found that another suite of species that favored low- to mid-elevation, old-forest habitats (e.g., fisher, flammulated owl [*Psiloscops flammeolus*], northern goshawk, pileated woodpecker [*Dryocopus pileatus*], boreal owl [*Aegolius funereus*], northern flying squirrel, and black-backed woodpecker [*Picoides arcticus*]) also experienced substantial habitat losses of large early-seral trees, mostly in lower elevation areas (Wisdom et al. 2000: 76).

The loss of live, early-seral, large trees simply described as >21 inches dbh masks an even greater loss of very old, ancient, early-seral trees, some more than 50 inches dbh, and virgin forest refugial patches that had survived for centuries longer than the surrounding old trees and forests. These were remnants of bygone forests; occasional trees and small patches of trees that had lived for 400 to 800 years (fig. 20), and which possessed defects and habitats unknown to younger large trees. An example

The loss of live, early-seral, large trees simply described as >21 inches dbh masks an even greater loss of very old, ancient, early-seral trees.

Francis Eatherington



Evan Frost

Figure 20—(A) Ancient (>400 years old) Douglas-fir and (B) ponderosa pine trees. Note presence of old fire scar in the outer bark on the left side of the Douglas-fir, and its flaking outer bark. Ponderosa pine exhibits broad orange bark plates, shallow bark fissures, and flaking outermost bark when ancient.

of this type of loss occurred when ancient ponderosa pine from eastern Oregon and Washington, the Black Hills, and the pine forests of Arizona and New Mexico were extracted during the earliest logging days. At that time, a heartrot fungus that was well known to foresters and mill owners contributed to significant losses in cull volume. The fungus was *Polyporus anceps,* the cause of red ray rot (Andrews 1955). In the forest, this decay would have created opportunities for cavity excavation of very large, ancient trees that would survive for centuries, then become snags and fallen denning logs. Similar stories can be told for scattered ancient western larch, Douglas-fir, white pine, Shasta red fir, noble fir, sugar pine, and other ancient trees that would live for centuries with major heartrot and butt defects, and then serve as snags and fallen logs for a few centuries more (Wagener and Davidson 1954). Reestablishing such ancient trees to east-side forests is both a worthy ecological and social goal.

Gaines et al. (2017) identified six surrogate species for assessing live, medium and large, early-seral tree forest communities: northern goshawk (fig. 21) and Cassin's finch (*Haemorhous cassinii*) for medium to large trees in all forest community types; Larch Mountain salamander (*Plethodon larselli*), pileated woodpecker (fig. 21), and American marten for medium to large early-seral trees of cool-moist forests; and white-headed woodpecker for medium to large, live, early-seral trees in dry forests (Gaines et al. 2017: 37). All groups experienced a decline in viability relative to historical conditions, with northern goshawks showing the least decline, and white-headed woodpeckers the greatest (Gaines et al. 2017: 38). Gaines et al. (2017: 238) suggested that "to increase viability outcomes, managers could identify and protect large tree and snag habitat within all forest types." All

Iosto Doneddu

Figure 21— (A) Northern goshawk and prey. In eastern Oregon and Washington, goshawks often build stick nests on large mistletoe brooms or on other abandoned stick nest platforms. (B) Pileated woodpecker feeding on wood borers in decayed sapwood. Pileated woodpeckers excavate cavities in medium- to large-size conifers of most species, especially where there is internal tree decay. Like many woodpeckers, they feed on wood borer larvae (Cerambycid and Buprestid) that infest dead trees or dead parts of live trees.

viability assessments have in fact come to similar conclusions—that viability for wildlife populations associated with live and dead, large-tree and old, early-seral-tree forest habitats, particularly those at lower elevations, has been reduced compared with historical conditions.

Box 17

Large Trees as Thermal Refugia

With warming, large and old, live and dead trees will become even more important as thermal refugia.

Climate vulnerability assessments for east-side forests highlight that large-tree forest structures will become increasingly important for wildlife population persistence given expected climate change impacts (Gaines et al. 2012, Raymond et al. 2014, Singleton et al. 2019). Habitat components provided by large trees, particularly high-quality nesting, resting, and denning structures (fig. 22), and thermal refugia, will become increasingly important for animals responding to worsening disturbance regimes and thermal stress under climate change. Forests with large and old live and dead trees, especially those with cavities, have long been recognized as providing important protected environments during cold weather (Holthausen and Marcot 1991). Mediated environmental conditions provided by large-tree forests will become increasingly important as periods of extreme high temperatures become more frequent (Gaines et al. 2012, Raymond et al. 2014, Singleton et al. 2019).

Tammi Zynski

Figure 22—Large, old trees with root and butt rot or heartrot make excellent down logs and often provide denning structures once they die and have fallen.

Large-tree forest structures will become increasingly important for wildlife population persistence given expected climate change impacts.

Coarse and Fine-Filter Management

> Adapting local and regional landscape patterns to climate change is essential to conserving characteristic patterns of all forest seral stages. Species of concern may benefit from additional fine-filter mitigations that are not provided through coarse-filter strategies. Retaining an extensive backbone of medium- and large-size trees is critical for conserving and recruiting diverse forest wildlife habitat components.

Management based on reference conditions using historical and climate change analogues (Keane et al. 2009) can be a strong foundation for multispecies conservation (Stine et al. 2014). Knowledge of historical or natural range of variability of landscape conditions (HRV, NRV) (Agee 2003, Landres et al. 1999, Morgan et al. 1994, Wiens et al. 2012) improves managers' understanding of the direction and magnitude of changes in forest and nonforest successional patterns over the period of management (Hessburg et al. 2015). Climate change analogue references, or future ranges of variability (sensu Hessburg et al. 2013, Keane et al. 2009), will be even more useful for predicting the associated variability under predicted climate changes, which can aid managers in adapting or transitioning current landscapes as they consider expected changes.

Regardless of forest structural class, retaining a "backbone" of medium- and large-size trees is fundamental to conserving and developing the variety of forest structures and spatial patterns that support diverse wildlife communities (Gaines et al. 2007, 2010; Lyons et al. 2008; Spies et al. 2018; Stine et al. 2014). Where forest disturbance processes are dynamically distributed, as on the east side, it is impossible to predict where late-successional and old forests will occur (Keane et al. 2009, Wiens et al. 2012), or to hold them in place. However, if remnant large and old trees comprise a backbone of many forest patches, many more late-successional forest patches can occur in a relatively short span of time after disturbances because the habitat component that takes the longest to recruit is already present (Hessburg et al. 2015).

Managing for a shifting mosaic of habitats, while recognizing that some areas with unique habitat values may be at high risk of disturbance, is an important principle for forest wildlife habitat management. These values would include large live trees, standing dead snags and fallen logs, productive herbaceous openings, and other features or patterns that provide food, shelter, and security to wildlife. Lehmkuhl et al. (2015) proposed a landscape-scale management concept that retained existing areas with important late-successional habitat values. These values could be conserved while promoting fire-resilient conditions in nearby areas that could develop owl or goshawk habitat values within a few decades and replace those where habitat values would be compromised by high-intensity wildfire. They also suggested identifying nonhabitat areas that could be managed as crown fire breaks, where surface fires were more likely, to improve landscape-scale fire resilience. This sort of multiscale, multitemporal planning framework for conserving and recruiting large-tree forest structures will be increasingly important going forward. Monitoring and adaptive management given known climate-driven trends in disturbance regimes and species distributions will be essential accompaniments (Lehmkuhl et al. 2015).

Large trees, snags, and fallen logs provide immense habitat value for forest wildlife, but environmental and disturbance context are important. They did not occur everywhere historically; nor will they in the future because of forest reburning (Prichard et al. 2017), and they are more apt to persist when their abundance is discontinuous.

Large trees, snags, and fallen logs provide immense habitat value for forest wildlife, but environmental and disturbance context are important.

Large Trees, Physical Processes, and Fish Habitats

> Large trees play a vital role in creating instream structure by adding complexity to stream channels and providing shade, and they are essential to creating high-quality, durable fish habitats.

Large trees and old forests affect the distributed hydrology of forests in general and of their associated streams and riparian areas (Bisson et al. 2003). Large dead wood originating from old forests ends up in streams by either falling in from riparian zones proximal to streams or by being delivered in landslides, mass failures, and debris torrents after moderate and severe disturbance events such as wildfire (fig. 23) (Dunham et al. 2007, Miller et al. 2003, Wondzell and King 2003).

Large wood physically structures stream habitat for a wide range of aquatic organisms, including anadromous fish (Gurnell et al. 2002, Sedell et al. 1988). Large wood from debris torrents can produce logjams that can span a channel (fig. 24). Later, a peak-flow event can redistribute wood locked in jams over great distances, providing structures for habitat creation and channel evolution. Larger stems with root wads are more effective at creating deep plunge pools and side channels than smaller wood pieces because they can deflect greater hydraulic power and they last longer in streams. Smaller stems are often removed by spring floods and other channel reorganizing events.

Across a continuum of habitats from the headwaters to the mouth, large wood plays a functional role in the creation of complex habitats for native fish (Fausch

Rhys Logan

Rhys Logan

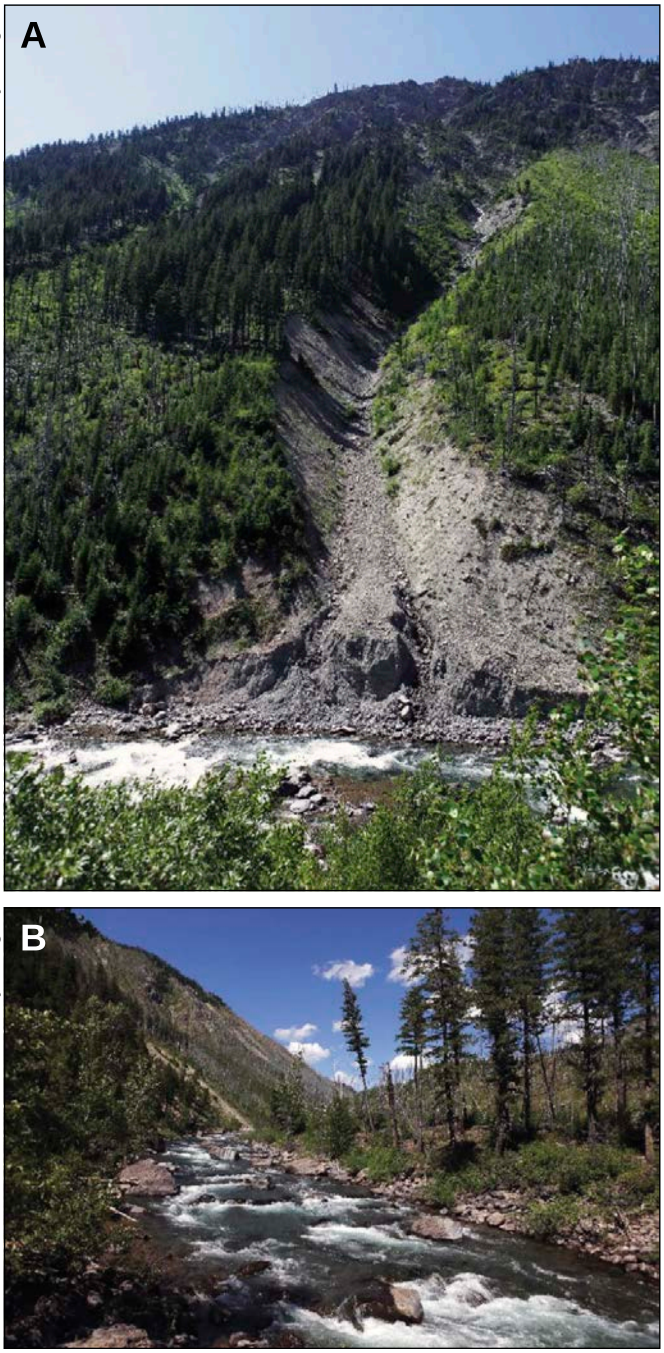


Figure 23—(A) Landslide after a reburn in lodgepole pine and Douglas-fir forest, Dry Fork of the Blackfoot River, Bob Marshall Wilderness Complex, Montana. Soil cohesion was lost within 10 to 15 years as dead tree roots rotted. Saturated soils triggered the landslide. (B) Instream habitat below the landslide area.

et al. 2002, Vannote et al. 1986). Thus, management strategies like the Aquatic Conservation Strategy (ACS) of the NWFP were designed to enhance ecological processes on federal lands through changes in tree harvest and protection of riparian areas (Reeves et al. 2006). The positive effect of increased large tree abundances in riparian areas and improved salmon habitat as a result of the ACS and the PACFISH and INFISH strategies (Dombeck 1996, Williams and Williams 1997) have been described for the interior Columbia River basin (Roper et al. 2019) and elsewhere in the NWFP jurisdiction (Miller et al. 2017).

Highly complex habitat mosaics reflect a combination of dead trees, log and debris jams, sediment, varying hydraulic power, and stream width (fig. 24). In headwater areas, habitat is often characterized by high-gradient streams. In such environments, large wood moderates flow, providing cold water and microrefugia for fish, salamanders, and other aquatic species (Vannote et al. 1986). Farther downstream, large wood contributes to the complexity of floodplain habitats by slowing water, depositing sediment, and facilitating seasonal flooding. These floodplain environments are highly productive for native fish and are enhanced by the presence of embedded large wood (Jeffres et al. 2008).

Large water-adjacent trees often contribute to the complexity of aquatic habitats (Reeves et al. 2018). Large wood slows water, resulting in the deposition of sediment that is then sorted into microhabitats by the water column (Montgomery and Buffington 1997). Instream cover provided by large wood offers critical refuge from predation, particularly for rearing juvenile fishes (Dolloff and Warren 2003). Large wood is also important to the life histories of dozens of species of fish that associate with large wood for cover, spawning, and feeding. Other aquatic organisms such as crayfish, freshwater mussels, and turtles also use large wood during part of their life cycles (Dolloff and Warren 2003). Further, large wood with root wads tends to survive

Figure 24—Diverse instream habitat relies on inputs of wood and sediments. Source material is often associated with adjacent riparian areas or nearby hillslopes with hydrologic connectivity. Natural disturbances such as wildfires, floods, debris flows, shallow mass failures, and landslides can be instrumental in the delivery of source material necessary for instream habitat complexity.

longer in the stream channel, providing cover and moderating sediment and flow dynamics more effectively than small wood, which tends to be washed downstream.

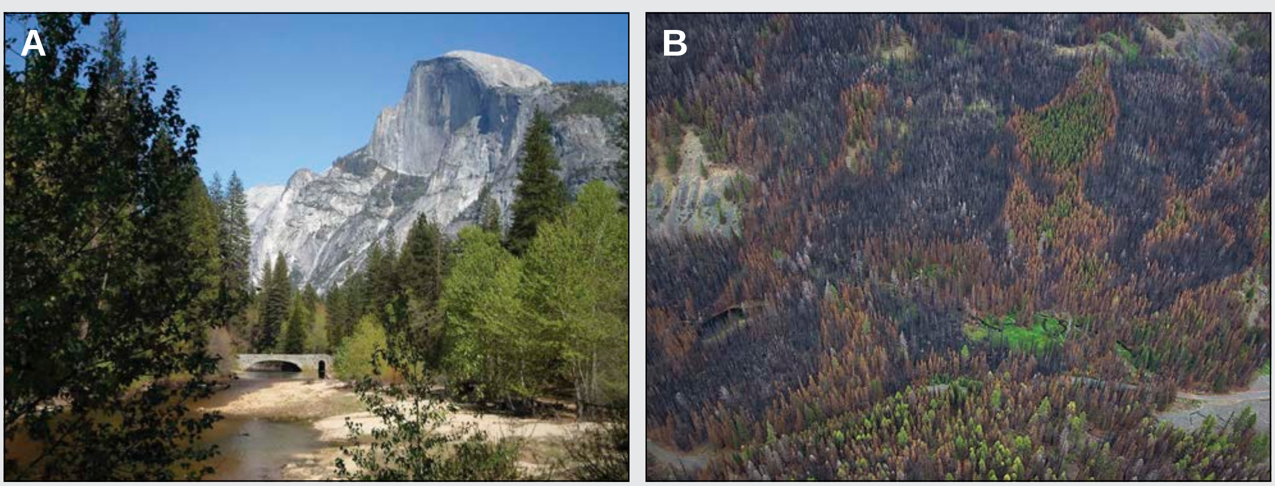
Shading of stream channels by large riparian trees provides critical thermal regulation and cooling (Poole and Berman 2001), particularly for cold-water-dependent fish such as bull trout (*Salvelinus confluentus*) (Dunham et al. 2011). As climate changes, anticipated thermal stress will further reduce habitat available to native coldwater fish (Falke et al. 2016, Isaak et al. 2017, Rieman et al. 2007). Large trees throughout the river network are important regulators of thermal condition.

Box 18

Large Trees and Floodplains

Floodplains are maintained (i.e., refilled) by pulsed hillslope disturbances (surface erosion, debris torrents, landslides, mass failures) that convey soil, cobble, boulders, and trees to streams (Wondzell and King 2003) and temporarily fill the channel. Disturbances often originate at higher elevations in the watershed, in steep-sloped face drainages, and in steep tributary drainages (Wondzell and King 2003). Wildfires are the disturbance process that eventually catalyze these hillslope processes (Bisson et al. 2003). Large trees are important to retain in areas with hydrologic connectivity to streams because they provide longer duration habitat structure in floodplains than small trees (Bisson et al. 2003). In contrast, floodplains most often experience hydrologic rather than wildfire disturbances, which come in the form of ice floes at break-up and spring peak-flow events that can scar and regenerate hardwood trees and redistribute some hardwood shrubs. Occasional pulsed large depositional events are essential to maintaining floodplain functionality. Otherwise, annual flows downcut the channel, ultimately dewatering the floodplain and making it easier for conifers to encroach (fig. 25).

National Park Service



John Marshall

Figure 25—(A) Conifers encroaching on the floodplain of the Merced River, Yosemite National Park, California, looking east toward Stoneman Bridge and Half Dome. (B) The 2012 Wenatchee Complex Fire removed many conifers that, in the absence of fire, had encroached on the Mission Creek floodplain, formerly occupied by hardwood shrubs and wet meadow sedges and grasses in the Wenatchee National Forest, Washington.

Large Trees Influence Carbon Sequestration, Hydrology, and Ecosystem Services

Carbon Sequestration

> Understanding the carbon balance of forests is complex, involving many variables. The relative importance of managing for larger versus smaller trees is a yet unresolved topic.

Maintaining or increasing carbon sequestered by forests is of growing interest owing to concerns about the effect of increasing atmospheric carbon dioxide (CO_2) and other greenhouse gases (GHG) on the Earth's climate. The role of large and old versus small and young trees in sequestration, especially when considering long-term carbon storage in wood products, is an ongoing and still-unsettled topic of study. Carbon in forests is stored in several **stocks**, including in above- and below-ground live and dead trees and other vegetation, down wood, litter and duff on the forest floor, and in mineral soil. The amounts of carbon in each of these stocks varies by vegetation type, environmental setting, geographic area, and time since disturbance (McKinley et al. 2011).

The role of large and old versus small and young trees in carbon sequestration is an ongoing and still-unsettled topic of study.

Carbon **flows** or moves between stocks through time (Kurz and Apps 1996, Kurz et al. 2008). For example, carbon from live trees flows to standing dead tree (snag) stocks, then on to down wood stocks, then to the forest floor, and eventually to mineral soil stocks, with decay fungi and bacteria emitting CO_2 to the atmosphere at every stage of decomposition. When considering carbon accounting, these flows are critical to understanding terrestrial carbon sequestration because the processes affecting carbon storage differ significantly by stock.

Carbon **flux** is the exchange between an ecosystem and the atmosphere, with live vegetation absorbing or fixing carbon in plant parts via photosynthesis, and decomposition or combustion of dead vegetation, which emits carbon (Cohen et al. 1996, Goward et al. 2008). Decomposition and combustion processes differ significantly in their rate of carbon emission, with combustion being the process that more rapidly emits carbon to the atmosphere (Williams et al. 2012).

To understand the carbon **balance** of a forest, one must consider the amounts and fates of carbon removed in the form of harvested wood products, where the carbon in tree stems flows to lumber, paper, energy production, landscaping, and landfills, and eventually returns to the atmosphere (Hayes et al. 2012). Carbon assessments also consider the effects of **substitution**, where using wood (e.g., for energy or construction) may result in fewer emissions than using other materials (e.g., oil, aluminum, or steel) (Baral and Guha 2004, Hall and House 1994, Hudiburg et al. 2011). Another consideration is **leakage**, where increases in forest stocks in one region are offset by increased harvest in another in order to supply wood products (e.g., Di Maria and Van der Werf 2008, Haim et al. 2016, Kuik and Gerlagh 2003).

The Role of Older Forests

> Forests, especially old-growth forests, are important carbon sinks.

Although forest sector stocks and flows are well understood, substantial disagreement persists on methods to account for biomass energy, substitution, and leakage. Vigorous young forests accumulate carbon at a faster rate than older forests, but when considering carbon flux, what is important for atmospheric carbon calculations is the average stock stored in forests over large spaces and long timeframes. Confounding these calculations is the harvest of wood products over a forest rotation, and across landscapes where there is high variability in space and time in forest rotations. Although state-level estimates for Oregon and Washington are still under development, analyses from California show that stocks in harvested wood products represent but 8 percent of the in-forest carbon component (Christensen et al. 2019). Although sensitive to assumptions, analyses suggest that accounting for harvested wood products, substitution, and biomass energy does not compensate for harvesting carbon-dense old forests (McKinley et al. 2011). However, we note that full accounting for leakage is highly problematic.

Comparing Current and Presettlement Era Conditions

> Carbon stocks increased in eastern Oregon and Washington from 1994 to 2007, but calculating the level of stocks associated with earlier conditions is difficult to impossible.

Stand reconstructions suggest that many presettlement era east-side, dry, and some moist mixed-conifer forests exhibited open canopies, with grass- or shrub-dominated openings, large areas of low overstory tree cover (10 to 30 percent) and low to nonexistent understory tree cover (Hessburg et al. 2016, Stine et al. 2014). About 40 percent of the trees were in medium and large tree size classes historically, with large (>21 inches dbh) fire-tolerant tree stocking averaging 18 to 38 trees per hectare (tph) (about 7 to 15 trees per acre) (table 3 in Stine et al. 2014). Currently, measured overall tree density is at least twice as high in these types, with 7 to 10 percent of the trees in large size classes, and large fire-tolerant trees at 13 to 25 tph (table 3 in Stine et al. 2014). Because individual large trees store much more carbon than small trees, it is not clear that current dense stands with fewer large trees store more carbon than presettlement era fire-maintained forests. For example, studies that compared mixed-conifer forests in California found that current unmanaged old forests stored 28 percent less live tree carbon than presettlement era reconstructed forests (Fellows and Goulden 2008, North et al. 2009a). Fewer large early-seral trees may provide more carbon storage when accounting for disturbances.

It is difficult to reconstruct presettlement dead tree density and associated carbon storage because large snags and fallen logs last longer than small ones, and smaller dead trees are more readily consumed by frequent fire. Here, the fire ecology and fire history literature provides good clues, showing that frequent fires made these ecosystems generally woody-fuel-limited (Agee 1993, Agee and Skinner 2005, and references therein), where primary surface fuels were grasses and shrubs, hence increasing or maintaining a high likelihood of low-intensity future fires. Carbon stocks increased in east-side forests from 1994 to 2007, though, as in California, the increase in dead wood stocks was greater than that of live trees (Gray and Whittier 2014) (fig. 26).

Fuel-Reduction Treatments and Carbon Storage

> Restoration to climate change adapted frequent-fire forests would likely promote stands with large trees; however, tree densities would likely be lower than historical densities.

Fuel treatments to reduce future wildfire severity also reduce forest carbon stocks in the short term, but there is important debate as to whether wildfire in untreated stands results in even greater reductions in live and dead forest carbon. New research is exploring this question in detail.

The issue of carbon and wildfire dynamics, when coupled with management, is complex and not readily characterized. For example, recent landscape simulations in the eastern Cascade Range of Oregon showed that more carbon was stored on a simulated forest landscape without management than with management, despite the occurrence of some large, high-severity wildfires (Spies et al. 2017). In simulations, high-intensity wildfires covered less than 1 percent of a large (>1 million ac) landscape. The mean annual proportion of the forested landscape burned with high-severity fire over 50 years (15 replications) varied from <0.01 to 1.0 percent, across all scenarios modeled. The reference period for predicting fire occurrence was 1992–2009, a period of relatively frequent large fires. The other 99 percent of the landscape was covered by aggrading forests where carbon gains were offsetting losses to wildfire. Where losses to high-severity fires are greater than this, it is unclear whether gains will outstrip losses to wildfire. For example, models suggest that the proportion of forested area that is highly suitable for fires >100 ac will increase three- to fourfold

The issue of carbon and wildfire dynamics, when coupled with management, is complex and not readily characterized.

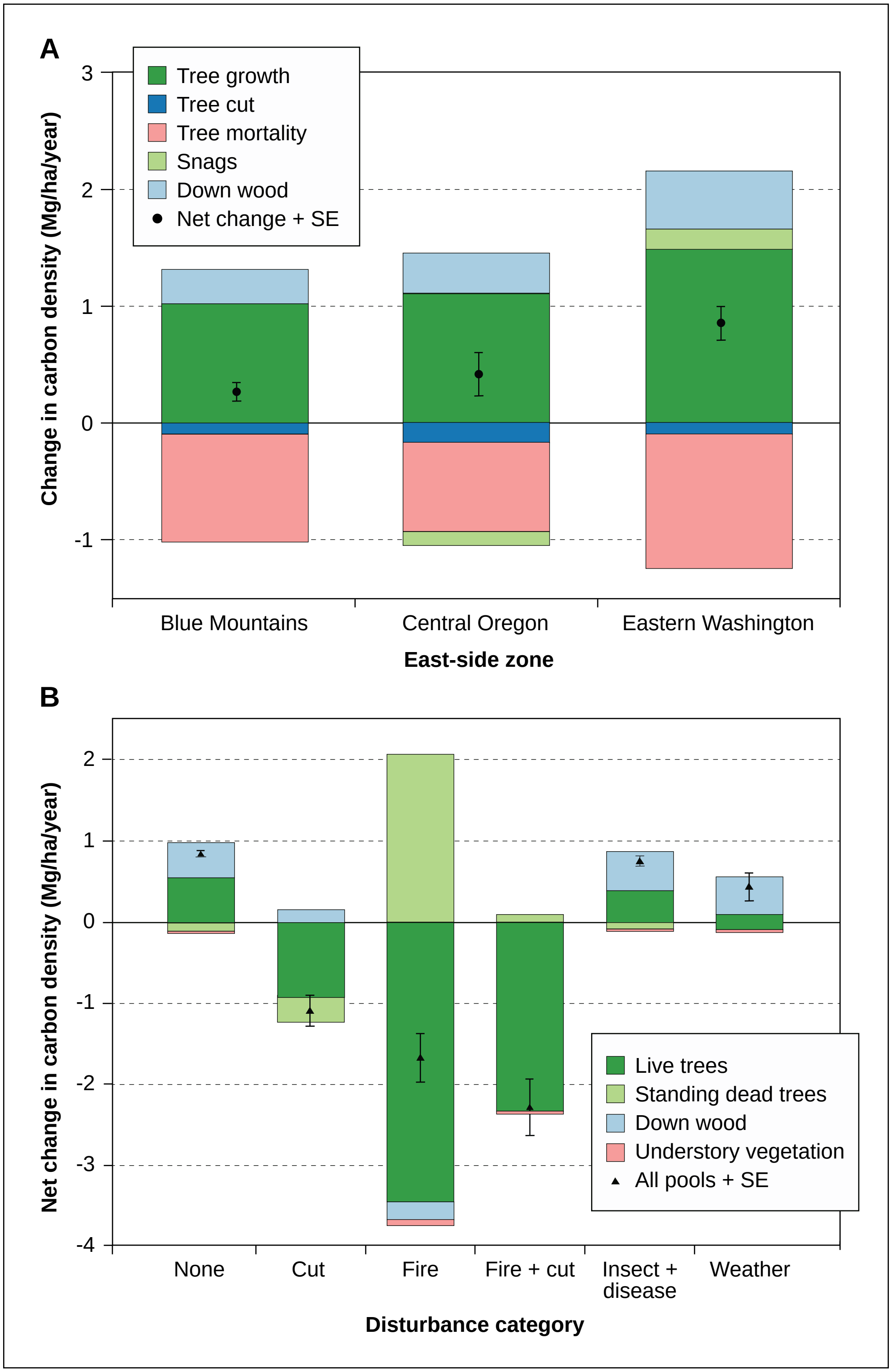


Figure 26—(A) Net annual change in carbon density over the period 1994–2007, by pool, and for all pools, on national forests in eastern Oregon and Washington. Standard error (SE) bars are for net change, based on results in Gray and Whittier 2014. (B) Net annual change in carbon density, 1994–2007, by pool, and for all pools, on national forest lands in eastern Oregon and Washington, combined by disturbance category. SE bars are for total flux.

Box 19

Treated and Untreated Stands and Their Carbon Storage

At the stand or patch scale, simulations of eastern Oregon ponderosa pine and California mixed-conifer stands showed that understory thinning with followup prescribed burning, followed by an uncontrolled wildfire, retained more carbon than wildfire alone in unmanaged stands (Harmon et al. 2009, Hurteau and North 2009) (fig. 27). On large landscapes, this is a critical consideration because burned area and wildfire severity are on the rise, and even greater burned area and fire severity are projected for coming decades. Scenarios in which the likelihood of low- or moderate-severity fire is increased with proactive management, modeled after the presettlement era fire regime, would have a higher likelihood of promoting stands with greater numbers of medium- and large-size trees of early-seral species. These conditions would provide more stable live tree carbon stocks. Alternatively, large and severe fires would result in slower forest recovery and longer periods of reduced live tree carbon stocks (Hurteau et al. 2011).

Continued on next page

in the Blue Mountains, and fourfold in the eastern Cascades by the end of the century (Davis et al. 2017b). Process models project similar increases (Case et al. 2019, Cassell et al. 2019, Halofsky et al. 2013, Kerns et al. 2017, Kim et al. 2018). Under these increases, losses may in fact outstrip gains in carbon storage.

Since 1997, large wildfires have increased in frequency, often covering much larger areas (100,000 to 250,000+ ac) (NIFC 2020), and these fires are releasing significantly more carbon to the atmosphere. Where these much larger fires occur, it is unclear whether carbon gains in the remainder of the landscape offset losses to wildfire. It is clear, however, that where fuel treatments have a small footprint, most will not overlap wildfires or impede their progress (sensu Prichard et al. 2017), and in the short run will result in carbon losses to the atmosphere. These results do not mean that fuel treatments should not be used to restore landscapes, but that complex tradeoffs and the scale of potential disturbances should be considered.

Where fuel treatments have a small footprint, most will not overlap wildfires or impede their progress, and in the short run will result in carbon losses to the atmosphere.

Hydrologic Processes

Large-tree cover with small openings can improve snow accumulation and snow-pack shading, reduce sublimation losses, delay snowmelt, and increase flows.

Compared to closed-canopy forests, forest canopies that are somewhat opened via moderate-intensity harvest or wildfire disturbances can increase snow accumulation (Lundquist et al. 2013; Storck et al. 1998, 1999), reduce rates of snowmelt via

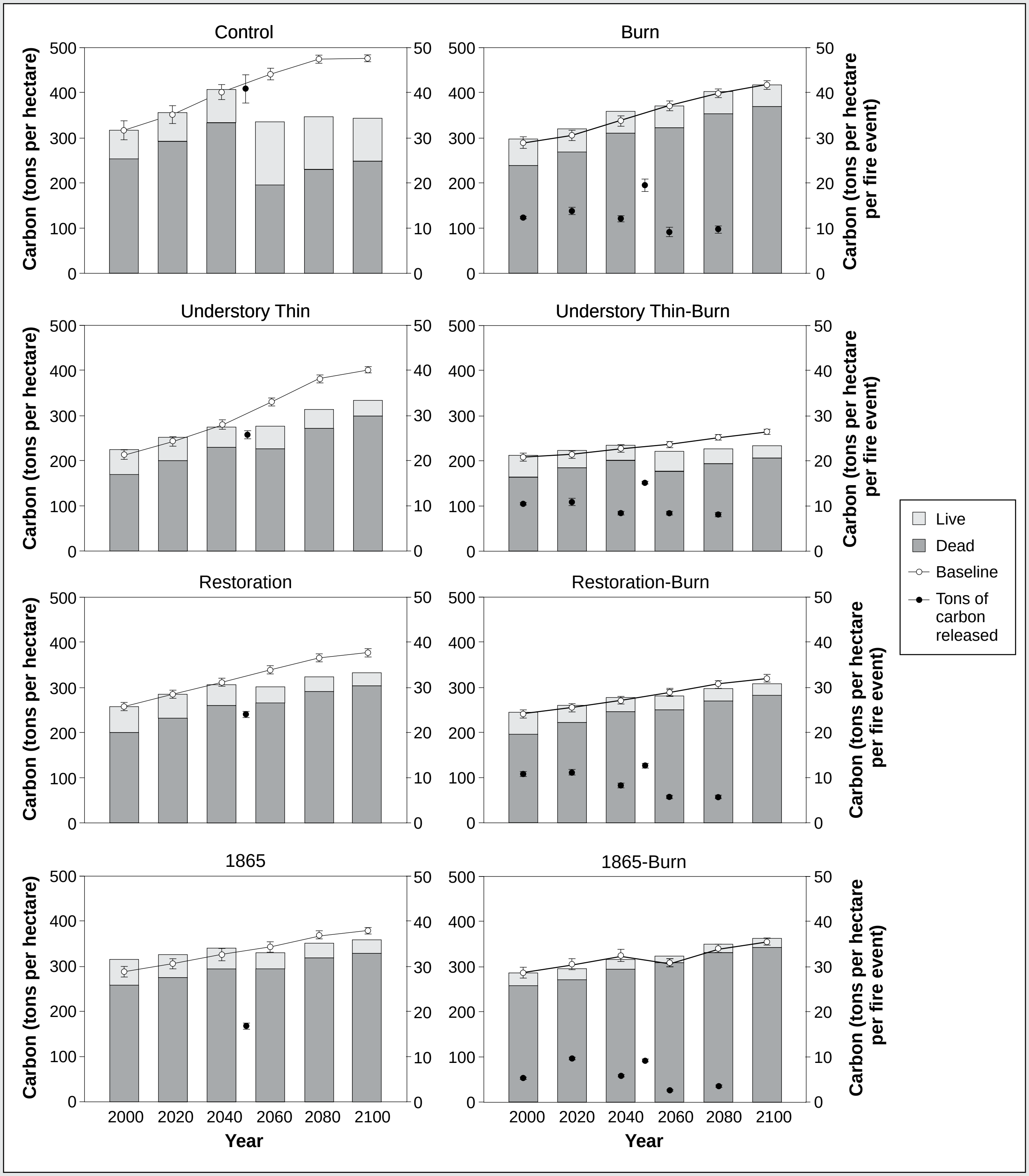


Figure 27—In wildfire-prone forests in the Southwestern United States, tree-based carbon stocks were best protected by fuel treatments that produced a low-density stand structure dominated by large fire-resistant pines. Graphs show tons of carbon per hectare stored in live- and dead-tree biomass and released by fire in eight fuel treatments: (A) control, (B) burn only, (C) understory thin, (D) understory thin and burn, (E) restoration thin, (F) restoration thin and burn, (G) 1865 reconstruction, and (H) 1865 reconstruction and burn. Black dots indicate the tons of carbon per hectare released in the 2050 wildfire and during each prescribed burn event. Baseline dots with standard error bars represent the total aboveground live and dead biomass, starting from the posttreatment stand condition, if the forest did not burn. Baselines in B through H can be compared with the control's baseline to assess total changes in carbon stocks from pretreatment condition. Adapted from Hurteau and North (2009).

improved snowpack shading (Dickerson-Lange et al. 2017, Schneider et al. 2019), and increase total annual and late-season low flows (Troendle et al. 2001, VanShaar et al. 2002, Waichler et al. 2005, Woods et al. 2006). For example, Sun et al. (2018) showed that canopy gap treatments hold considerable potential for enhancing late-season low flows and noted later snowmelt in small- to medium-size canopy gaps (the ratio of gap radius (r) to canopy height (h) $\leq$ 1.2). They also noted that snow melt rates were more sensitive to changing canopy gap size in a medium range of gap sizes ($0.5 \leq r/h \leq 1.2$).

In some cases, summer low flows can be more affected by riparian vegetation than the surrounding forest (Moore and Wondzell 2005); context matters significantly (VanShaar et al. 2002, Whitaker et al. 2002). Depending on leaf area, evapotranspiration by young trees can be greater than by older trees, which could reduce soil moisture and result in reduced streamflows from younger stands.

The impact of forest management or disturbances on hydrologic processes is variable, depending on local climate; soil type, texture, and depth; vegetation type, height, leaf area, and canopy cover; topographic setting, dominant basin orientation and aspect; and the spatial patterns of disturbance. Distributed hydrology models that are able to simulate energy, mass, and water balance simultaneously are vitally important tools for estimating how changes in forest management practices and tree cover can affect moisture states such as canopy interception and storage, snow water equivalent and soil moisture, and fluxes such as evapotranspiration, sublimation, and streamflow (Wigmosta et al. 1994).

How Has Our Scientific Understanding of East-Side Oregon and Washington Forests Changed?

Over the past three decades, much has been learned about the landscape, fire, wildlife, plant and forest ecology of inland Northwest forests and the social systems and processes that frame their management. Landscape assessments and field and experimental research from numerous laboratories have provided new insights about the structure and function of local and regional landscapes and the human dynamics that influence them.

Changes in Fire Regimes and Forest Conditions

> Many forests are now more homogenized. Consequently, disturbance regimes have become more severe, causing widespread ripple effects and further uncharacteristic landscape alterations. Activities that are intent on restoring large-tree and old-forest abundance will be aided by considering 20th- and 21st-century changes to forest and nonforest landscapes and the east-side climate, and how those changes have altered wildfire regimes.

Removal of characteristic wildfires from dry, moist, and cold forests has brought about a cascade of changes to east-side landscapes.

Throughout the east side, large trees, old forests, and their associates have been substantially influenced by early-selection cutting and regeneration harvests (Hann et al. 1997, Hessburg et al. 2005, Lehmkuhl et al. 1994, Wisdom et al. 2000). Present-day stand structure of these previously harvested areas is predominantly even-aged where regeneration harvesting was practiced, and uneven-aged where selection cutting was more common (Hessburg and Agee 2003), with tree species and genetic compositions that are often poorly adapted to the environment. Most forest conditions and their wildfire regimes have been unwittingly but significantly altered by a combination of factors (Hessburg and Agee 2003, Johnston 2017). Removal of characteristic wildfires from dry, moist, and cold forests has brought about a cascade of changes to east-side landscapes; and factors working along with removal of fire have created conditions without precedent in historical records (Hessburg et al. 2005, Johnston 2017).

Forest conditions no longer resemble those that managers first inherited, and forest dynamics are nothing like those of prior centuries (Lehmkuhl et al. 1994). Along with fire exclusion came the establishment of many shade-tolerant species that are now larger than 21 inches dbh. Such increases in shade-tolerant tree densities have made forests less resilient to fire (Huff et al. 1995). Forests are now more homogenized, with little evidence of the former seral-stage diversity of local and regional landscapes (Hessburg et al. 2000, Johnston 2017, Lehmkuhl et al. 1994). For example, early- (rather than late-) seral conditions—in the form of grasslands, shrublands, sparse-woodlands, and recently burned bare ground—no longer reflect the patchiness and grain of the topography, but instead occur in 4-, 8-, and 16-ha (10-, 20-, and 40-ac) clearcuts, lending a high degree of fragmentation to affected landscapes (Hann et al. 1997, Hessburg et al. 2000, Lehmkuhl et al. 1994). Early in forest management, no one knew how important early-seral patches were to native plants and animals and the species that depend on them (Swanson et al. 2011). Similarly, recent research shows the vital importance of nonforest patchworks as broadscale habitat and fire delivery context for forest successional patchworks. Expanded forest area (Reilly et al. 2018) and densification of once open forests have reduced the area and patch sizes of nonforest at virtually all scales of observation, from portions of an acre to thousands of acres (Hessburg et al. 2016, 2019).

Research also reveals that historical forest landscapes ordinarily showed much burned area and that the patchiness and severity of area burned varied by forest type and geographic area (Leenhouts 1998). For example, dry ponderosa pine and dry mixed-conifer forests were characterized by frequent fires (Hagmann et al. 2013, 2014, 2017; Johnston et al. 2016, 2018). Frequent fires (every 5 to 25 years) maintained open-forest conditions with less than full canopy cover (Hagmann et

Box 20

Adapting Moist Mixed-Conifer Forests to Changing Climatic and Wildfire Regimes

In moist mixed-conifer forests, fires of all severities occurred, but there was a strong tendency for low- and moderate-severity fires to be most influential (Hagmann et al. 2013; Hessburg et al. 2007; Johnston 2017; Merschel et al. 2014, 2018, 2019) and geographic variation could be significant. Like dry forests and their frequent fires, moist mixed-conifer forests experienced frequent to moderately frequent (every 30 to 50 years) ignitions. Ignition frequency, interacting with productive growing conditions, characteristically yielded low- and moderate-severity fires (Johnston 2017; Johnston et al. 2016, 2018). These "take some and leave some" fires promoted a medium- to coarse-textured mosaic of tree clumps and gaps of various sizes.

Moderate-severity fires regularly consumed woody fuels in gaps and thinned out small trees and dead wood under larger tree clumps (Hagmann et al. 2019). These effects increased the likelihood that most future fires would be of either low or moderate severity as well (Merschel et al. 2018). Feedbacks such as these in dry and moist forests were critical to landscape resilience and fuel maintenance at a level that enabled forest persistence at a broad scale. A century of wildfire exclusion effectively eliminated these feedbacks, and today many moist mixed-conifer forests are no longer climate or wildfire resilient. Recreating these medium- to coarse-textured mosaics would go a long way toward increasing the likelihood of future moderate-severity disturbance. As with dry forests, more severe fire weather conditions would occasionally contribute to more severe fires, and more benign conditions or adjacency to dry forests would foster fire regimes that resembled those of dry forests.

al. 2019), and this openness increased the likelihood that future fires would be low in severity (<20 percent of the dominant basal area or canopy cover killed by a fire) (Merschel et al. 2014, 2018). More severe fire weather conditions would occasionally foster moderate- (20 to 70 percent of the dominant basal area or canopy cover killed) or high-severity (>70 percent of the dominant basal area or canopy cover killed) fires, but open conditions, dominated by medium- to large-size trees, typically fostered more benign fire behavior and effects.

Perhaps most surprisingly of all, landscape science reveals that cold forests have also been significantly affected by fire exclusion (Hann et al. 1997; Hessburg et al. 1999a, 1999b, 2000; Merschel et al. 2018). Prior plot-based studies gave the impression that because fire frequency was infrequent (50 to 100 years) to very infrequent (75 to 150+ years) in cold forest plots, there was no impact from the loss of infrequent fires via fire exclusion (e.g., see Schoennagel et al. 2017 and references therein). These conclusions were found to be incomplete. Large landscape assessments, because they examine large wall-to-wall areas, revealed that seral-stage patchworks of cold forests and those of nearby nonforests had changed significantly via fire exclusion. A new inference emerged, that cold forests were burning somewhere each year because lightning ignitions were ongoing, occurring on high ridges, benches, and in other cold forest environments (Hessburg et al. 1999a, 2000, 2007). These ignitions yielded primarily moderate- and high-severity fires (Agee et al. 1990) that continually pockmarked the landscape, yielding broad variation in burned patch sizes and subsequent cold forest successional conditions. Burned cold forests yielded much broader variation in successional conditions and biotic diversity than expected.

Findings From Resilience Research

Landscape assessments improve understanding of how resilience works.

Looking more broadly, research into resilience and resistance mechanisms of local and regional landscapes has yielded new insights into the landscape ecology of fire (McKenzie et al. 2011). From the theoretical literature, we understand that landscape patterns were hierarchically structured (Allen and Starr 2017): fine-scale landscapes nested within mesoscale landscapes, nested within broad landscapes. Landscape scaling is subjective, however, depending upon the questions one asks (Allen and Hoekstra 2015). For forest landscapes and their multiscale structure, can we devise an approximate landscape hierarchy that makes a modicum of sense? Scaling depends on the magnification of the lens with which one looks at landscapes, and the questions one asks; no one set of hierarchies will do. Moreover, within a hierarchy, are we able to understand approximately how such a hierarchy works? For example, how does it provide resilience and resistance to disturbances and the capacity to maintain or rebuild landscape structure and organization? Landscape assessments and large landscape studies give us some insight into the structure, organization, and roles of hierarchical resilience mechanisms.

Across western North America, we find that a handful of emergent properties conferred forest resilience and resistance to historical disturbances and climatic changes. The following sections, excerpted and paraphrased from Hessburg et al. (2019), summarize that new understanding.

Scale-Dependent Spatial Controls on the Landscape Ecology of Forests and Their Disturbances

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Landscape resilience and resistance appear to be multilevel, dynamic, and subject to top-down and bottom-up drivers. Activities that are intent on restoring large-tree and old-forest abundance will be aided by understanding the multilevel nature of resilient landscapes and how those multilevel characteristics change in the context of changing wildfire and climatic regimes.

Wildfires were historically influenced by broad-, meso-, and fine-scale factors (Moritz et al. 2011, Peterson et al. 1998). Top-down, broad-scale factors included a wide range of climatic, weather, geologic, or geomorphic events. Mesoscale factors of local landscapes included spatial patterns of forest and nonforest, fuel and successional conditions, productivity and topoedaphic settings. Bottom-up, fine-scale factors included fine-scale surface fuel loading, microsite conditions, tree density variation, endemic insect and disease incidence and severity, topographic variations, and local continuity of tree canopies and ladder and understory fuels.

These broad-, meso-, and fine-scale factors together influenced biotic and wildfire conditions, and their relative contribution likely varied by event size. Bottom-up factors spatially controlled the sizes and effects of smaller fires, while top-down factors likely drove or constrained occurrence of the largest fires (Moritz et al. 2011). Fires in the middle range of sizes were likely driven by a tug-of-war—played out in real time—among top-down and bottom-up factors. Because forcing by top-down drivers can be highly influential, we suggest that forest resilience and resistance are mutable rather than static system properties (Millar and Woolfenden 1999). Hence, the study of historical ecology over varying climatic regions and periods helps us understand components and some plausible configurations of resilient ecosystems (Swetnam et al. 1999).

The study of historical ecology over varying climatic regions and periods helps us understand components and some plausible configurations of resilient ecosystems.

From a survey of highly varied ecoregions from British Columbia, Canada, to the Baja Peninsula of Mexico (Hessburg et al. 2019), we learn that historical wildfires influenced and were influenced by cross-connections between broad physiognomic patchworks of nonforest and the mix of extant forest successional conditions (sensu Wu and Loucks 1995). Nonforest types had surface fuels, typically grasses and dry or moist-site shrubs, that often supported, and were supported by, moderate- or high-frequency fires. Historical ignitions often spread quickly when they contacted these nonforest fuelbeds, and because of flashy fuel conditions, fire rates of spread were relatively fast, but flame lengths and fireline intensity were low. The primary fire behavior that was delivered to many patches of dry and moist forest was surface- rather than crown-fire driven. But nonforest patches were not restricted to low-productivity sites. Some occurred in topoedaphic settings that readily supported forest. Thus, the potential extent of forest area fully supported by climate and environmental conditions (the carrying capacity) was seldom realized (Bond and Keeley 2005).

Box 21

Cross-Connected Broad- and Meso-Scale Landscapes

Wildfires and other processes are influenced by interactions among forest and nonforest patchworks. Activities that are intent on restoring large-tree and old-forest abundance will be aided by understanding that maintenance of nonforest patchworks is essential broad-scale context to forest successional patchworks that reside within them. Activities that create long-term exceedance in either the amount of forest or area of complex forest will likely be met with wildfire or insect disturbances that significantly diminish forested area or structural complexity.

Similarly, forest successional patches in drier environments were open canopy with flashy fuels, typically favoring continued spread of surface fires, while those in cool-moist settings were more complexly layered, favoring mixed surface and crown fire, or crown fire alone. Carrying capacity of forest successional landscapes was also seldom realized because of extensive, fire-maintained, open-canopy conditions. During cool-moist climatic periods of lower than average fire frequency, tree densities would increase, and patches of nearby forest or woodland would expand, encroaching on and reclaiming areas of grassland or shrubland. However, during hot-dry climatic periods with elevated fire frequency and severity, grass, shrub, and woodland areas would again expand (e.g., see Beaty and Taylor 2009), often in new locations, and tree densities would decline.

Cross-Connected Meso- and Fine-Scale Landscapes

Heterogeneous patchworks of forest successional conditions and patch sizes are nonstationary, but they lead to similar future patchworks under conditions of modest climatic variation. Where variation in the climate is more intemperate, patchwork similarity declines. Activities that are intent on restoring large-tree and old-forest abundance will be aided by understanding that planning for dynamically shifting patchworks with large trees and old forests will reduce uncertainty of outcomes in comparison with static reserve planning.

Across many surveyed ecoregions, we find cross-connections and interactions whereby wildfires historically shaped and were shaped by fine-grained vegetation patterns within and among patches (Harvey et al. 2017). Fire interacted with patches of intermingled nonforest, dry, moist, and cold forests, maintaining high spatial variability in fire frequency and severity, and resulting in a multiscaled mosaic of seral stages and associated fuelbeds. For example, frequent surface fires would spread from dry forests into adjacent moist or cold forest patches, thereby maintaining lower surface fuel loads and structures that were atypical for the forest type. These spatial interactions explain the presence of open-grown lodgepole pine trees with multiple fire scars and historical subalpine ribbon forests interspersed with extensive wet and dry meadows (fig. 28). Historical forest successional landscapes were seldom at carrying capacity in either forested area or density because of disturbance-mediated feedbacks at meso and fine scales.

Box 22

Species Adaptations to Fire

Species traits and adaptations drive within-patch structure, composition, and response to disturbances. Activities that are intent on restoring large-tree and old-forest abundance should consider exploiting these traits when developing fine- and meso-scale management prescriptions.

Within patches, physiological traits and adaptations of species such as serotiny, thick bark, and vegetative reproduction strategies are critical not only to species persistence, but to the maintenance of a characteristic structure, composition, and fire severity. In historical frequent-fire forests, we learn that medium- and large-size ponderosa pine, western larch, and Douglas-fir displayed elevated crown bases that prevented fire from climbing into the canopy, and thick bark that insulated them from most basal scorching. Shrubs resprouted from deep root systems or via seeds long buried in soils. Native grasses were fire adapted, and some formed sods, which were available to reburn within a year. Bunchgrasses grew in individual tufts and tussocks, which provided fine-scale fuel discontinuities, while also making them resistant to fire-caused mortality. Patch-level structures such as clumped and gapped tree distributions were also supported by recurrent fires (Churchill et al. 2013, Larson and Churchill 2012). Clump and gap sizes varied predictably with seed-dispersal distances, in-filling rates, and patchy tree mortality driven by surface and ladder fuels and other physiological species-level traits.

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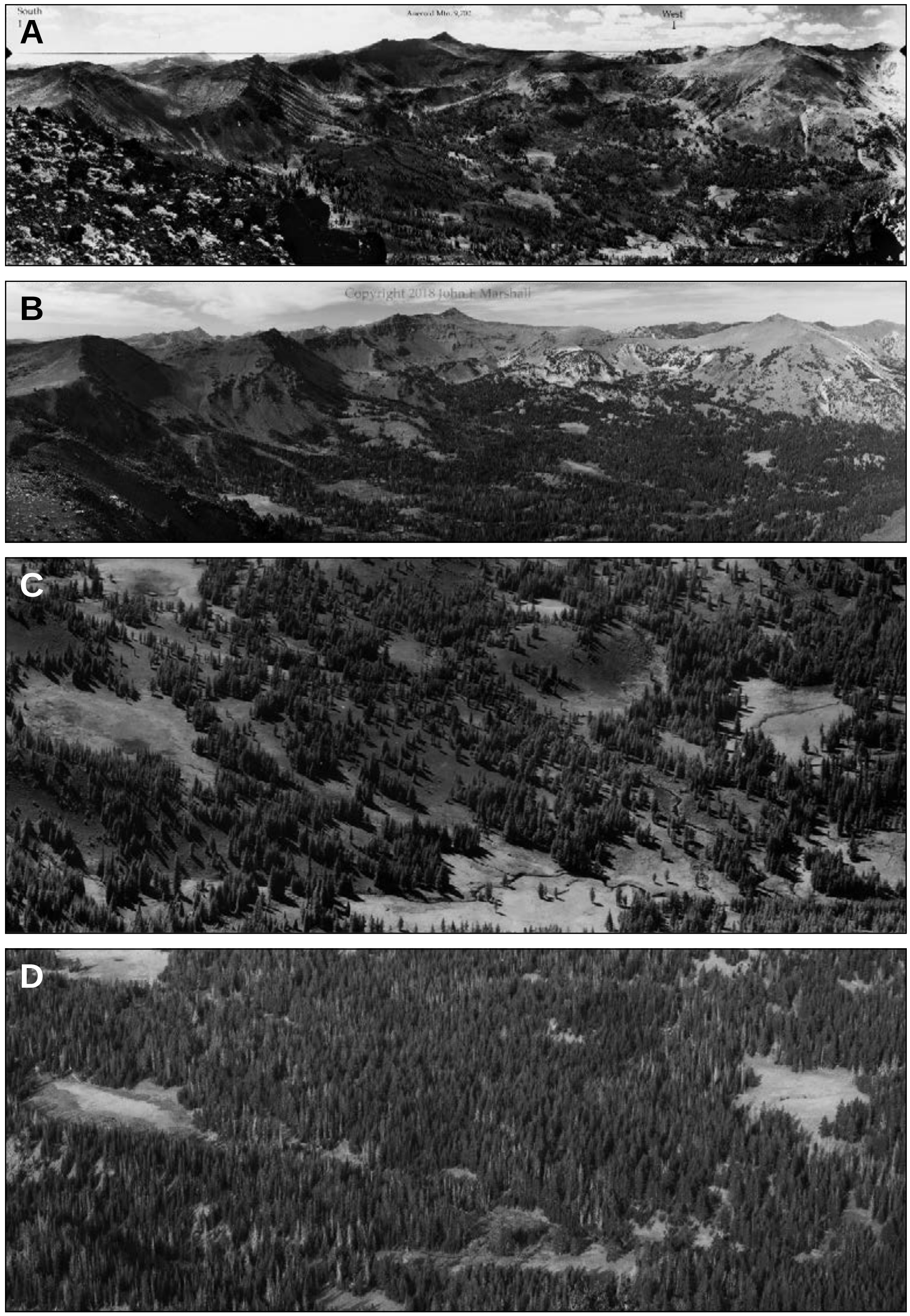

Figure 28—(A and B) Panoramic comparison (1936 vs. 2018) of forest conditions in McCully Basin, Eagle Cap Wilderness, Wallowa Mountains, Oregon. In photo A, note the presence of low-, moderate-, and high-severity fire patches in small sizes. In photo B, in the absence of fires, trees have widely encroached on both wet and dry meadows. (C and D): Close-up view of McCully Creek, Eagle Cap Wilderness, 1936 vs. 2018, showing a shift in dominance from wet and dry meadows (1936) to dense cold forest (2018). Forest species are lodgepole pine, subalpine fir, and Engelmann spruce. Bark beetle mortality in lodgepole pine, subalpine fir, and spruce is widespread in the overstocked 2018 conditions.

Likely Responses Under Climate Change

Extreme disturbances or climatic events can disrupt current landscape conditions at multiple levels. Restoring more characteristic multilevel pattern conditions should improve landscape resilience to climate change, restore more characteristic patterns and abundances of large trees and old forests, and improve landscape capacity to continually adapt to the coming changes.

Cross-connections between broad-, meso-, and fine-scale landscapes afford us clues for what is to come with the expected warming and drying of western North America (Davis et al. 2019b, Keane et al. 2013, Kitzberger et al. 2007). A steadily warmer and drier climate will likely contribute to decline in forest area to levels that are less than occurred historically. Conifer regeneration will likely vary along elevational gradients, with tree regeneration being poorest at warm and dry low-elevation sites (Dodson and Root 2013). Grasses, sometimes including nonnatives, will likely dominate lower elevation dry forest and woodland sites after fires, which may reduce conifer seedling establishment because of increased moisture stress from competition for water (Kerns et al. 2020).

A steadily warmer and drier climate will likely contribute to decline in forest area to levels that are less than occurred historically.

As nonforest area grows, area burned will likely increase across flashy-fuel-connected landscapes. This may have the effect of increasing fire frequency not only in dry forests, but also in some moist and cold forests, especially as they intermix with dry forests near ridgetops, valley bottoms, and south-to-north aspect transitions. In more rugged terrain, topography will continue to influence fire size and severity (Povak et al. 2018), but with continued warming and increasing fire sizes, we may see an erosion of topographic controls. Increased fire frequency will reduce canopy cover and tree density while favoring plant species with traits that allow them to survive or colonize quickly following fire. These trends may ultimately increase the amount of low- and moderate-severity fire that historically was associated with each forest type, thus redefining their fire ecology and associated forest and nonforest successional conditions.

We find that similar resilience mechanisms are shared across a wide range of western North American environmental conditions. Resilience arises through adaptations from physiological traits at the species level to physiognomic patterning at the ecoregion level; simultaneously, all levels are incrementally adapting to the prevailing climate. During periods of modest climatic variation, multilevel patterns support a system that appears to be stable but is not truly stable (metastable). When fueled by extreme disturbance or climatic events, this apparent stability can shift abruptly, mutating and changing the dominance and distribution of landscape conditions at all levels.

Box 23

Increases in Nonforest Under Climate Change

With predicted increases in temperature and declining precipitation throughout the 21st century, moisture stress will likely become a limiting factor to conifer establishment after severe wildfires in these drier forests (Dodson and Root 2013). The most continually moisture-limited sites may fail to regenerate for protracted periods after severe disturbance, and some may be a priority for reforestation where maintaining open forest or sparse woodland cover is important, whereas some will be a priority for conversion to healthy nonforest communities.

Effects of Climate Change and Invasive Species

Large trees of early-seral species play an expanding role in climate change and wildfire adaptation. Methods that favor recruitment and maintenance of large-tree populations and older forests will also favor expansion of some invasive and nonnative plant species, especially in drier forests. Invasive and nonnative species will continue to expand under climate change.

In the context of climate change adaptation, large trees of early-seral species play an even greater role. Under a warmer and drier climate, we can expect more area burned in larger fires, and more area burned severely. Large trees of fire-adapted species have a greater chance of persisting and are better adapted to these expected future climate and wildfire scenarios. That adaptability is greatly diminished where large trees are growing in high densities and where forests are multilayered with closed canopies. Numerous silvicultural and prescribed burning remedies are available to protect individual large trees and stands with concentrations of large trees (see “Silviculture Research: Stand Development and the Role of Large Trees” above).

Invasive and nonnative plant species concerns will expand under climate change, but methods are available to diminish their influence (Kerns et al. 2006, 2017). Many invasive plants of concern are ruderal or pioneering species after disturbances, with a high capacity to invade damaged or stressed sites. Preferring management tactics that minimize the likelihood of high-severity fires in dry and moist mixed-conifer forests will help to minimize more extreme influences of invasive plant species. Within cold forests, there is much less concern.

Several exotic plant species can invade after fire and other forest thinning disturbances (Kerns et al. 2006, 2017). These species can interfere with conifer recruitment and reduce some positive effects of restoration treatments by increasing fine-fuel continuity and fire spread rates (Kerns et al. 2017). For example, in some dry pine and dry mixed-conifer forests, and in nearby woodlands and grasslands, introduction of nonnative cheatgrass (*Bromus tectorum),* medusahead (*Taeniatherum caput-medusae*), and North Africa grass (*Ventenata dubia*) produce more abundant fine fuels than the native bunchgrass communities (e.g., *Festuca idahoensis*), which can then support more high-frequency burning than can be tolerated by native perennial grasses. Thus, there can be a management tension associated with some fuel-reduction projects that show an increased potential for expansion of invasive plants and concurrent weed management needs (Kerns et al. 2020). Even so, methods that reduce the likelihood of high-severity fire will tend to favor reduced impacts. Despite great efforts to implement these preferred methods, invasive and nonnative species will continue to expand under climate change.

Nonnative animals continue to increase in importance (Witmer and Lewis 2001). Of prime interest in eastern Oregon and Washington forests is the invasive barred owl (*Strix varia*), which is not native to the east side and is rapidly ramifying preferred forest habitats of the northern spotted owl. Experimental removal projects are planned and underway, but it is unclear how successful they may be in maintaining viable populations of spotted owls (Diller et al. 2016, Gutierrez et al. 2007). Nor is it clear how niche partitioning will settle out under removal and nonremoval scenarios (Lesmeister et al. 2018, Singleton 2015, Singleton et al. 2010).

Box 24

Forest-Human Community Resilience

Promoting forest resilience or resistance to climate change, wildfires, and other disturbances is a broad charge that will require planning for surprises, including extreme events and unexpected feedbacks. It will also necessitate being mindful and inclusive of species-level traits; patch-level tree clump and gap distributions, tree sizes, densities, and canopy layers; meso-scale, seral-stage, and fuelbed heterogeneity; and broad-scale forest and nonforest patchworks. This effort will likely require preemptively adapting landscapes in areas with anticipated future water deficits before abrupt changes occur from disturbance- or drought-related mortality events. Examples of preparing landscapes for the coming wildfire and climatic regime include reducing forest area, expanding woodland or grassland area, reducing canopy cover and layering, and increasing the areal extent of fire-tolerant species. In these ways, managers can also better prepare human communities for future uncertainty by reducing the likelihood of abrupt broad-scale changes.

Managing for Social-Ecological Systems

> Active management can support forest resilience, restoration of old trees and old forests, adaptation to ongoing disturbance, and active, ongoing, and meaningful engagement with human communities.

Numerous studies have demonstrated that purposeful and proactive land management requires active and ongoing engagement of human communities that depend on these landscapes.

Broad-scale and abrupt changes in landscape structure and organization can be difficult for native plants, animals, and human communities to withstand (Liu et al. 2007, Spies et al. 2014). Accordingly, a task for current-era managers is to manage for change with uncertainty in mind. Methods that narrowly focus on rebuilding late-successional and old forests cannot restore integrity or resilience to landscapes, nor can they bring about climate change and wildfire-adapted landscapes. However, they are an important piece of the puzzle.

Numerous studies have demonstrated that purposeful and proactive land management requires active and ongoing engagement of human communities that depend on these landscapes (Fischer et al. 2016). Social science research finds high levels of public support for some proactive forest management, such as thinning and prescribed burning on public lands with a high fire risk (Burns and Cheng 2007, McCaffrey et al. 2013).

Mitigating Risks to Forests From Invasive Plants

> Many surveyed family forest owners in the Western United States are concerned about invasive plants, try to control them on their property, and are aware of the need to cooperate with neighboring landowners to do so effectively.

Social science research regarding public support for mitigating risks to forests posed by invasive plant species focuses on individual and collective mitigation behavior among private forest owners. Among family forest owners surveyed in eastern Oregon, about half were familiar with invasive plant species that natural resource professionals consider problematic, two-thirds were concerned about potential negative impacts of invasive plants on their properties, and roughly half had treated invasive plants on their private parcels (Fischer and Charnley 2012). Awareness and concern predicted landowner treatment behavior, as did having biodiversity or wildlife habitat conservation as important forest management goals. In addition, some landowners were sensitive to the risks posed by invasive species on neighboring ownerships and were aware of the need to cooperate with neighbors to reduce these risks (Fischer and Charnley 2012).

In western Montana, surveys found a high level of awareness of invasive plants regarded as weeds by private forest owners, as well as high levels of weed control by landowners surveyed (78 percent) (Yung et al. 2015). Seeds coming from neighboring properties were reported as the biggest barrier to effectiveness in this study, and cooperation among neighbors was thought to be critical for effective weed control. Based on research undertaken elsewhere in the West, additional factors that drive individual landowner willingness to control invasive plant species include sense of community, the desire to be a good neighbor (Fischer and Charnley 2012, Yung et al. 2015), and beliefs that weed control is a collective problem and that groups of landowners can successfully control them (Lubeck et al. 2019). Although these studies have focused on private family forest lands, the findings that many forest landowners are aware of and concerned about invasive plants, and that they need to cooperate with adjacent landowners to control them effectively, suggest that landowners would strongly support measures to mitigate the risk of invasive plants on western public lands. Collaborative efforts to control invasive plants across land ownerships, and to restore ecosystems affected by them, have emerged in many parts of the West from local to state levels (Schelhas et al. 2012).

Public Support for Managing Invasive Insect Pests

> Public support for managing invasive insect pests, and the methods used to do so, varies by local community context—depending on variables such as severity and timing of the outbreak, nature of community connections to nearby forests, and level of public awareness and understanding of the problem.

Social science research about public support for managing invasive insect pests on public U.S. forest lands focuses on how people respond and adapt to invasions once they have occurred, and management actions proposed to address them. We are not aware of research on this topic from the east side of Oregon or Washington. However, in the Eastern United States, research finds high levels of support for controlling the exotic insect pest, hemlock woolly adelgid *(Adelges tsugae)*, in eastern hemlock *(Tsuga canadensis)* forests, although awareness of the issue among the public is low (Poudyal et al. 2016). There was less social agreement about how to control the pest, with use of chemical insecticides and biological control measures being more controversial than other remedies (Poudyal et al. 2016).

In the West, few studies have investigated the social acceptability of specific management responses to bark beetle outbreaks (Morris et al. 2016). However, research about public support for forest management interventions to address insect outbreaks more generally has found that it differs by community, depending on the recency and severity of the disturbance, level of economic dependence on the forestry sector, and level of knowledge and concern about their impacts. In general, the more recent and severe the outbreak and greater the dependence, knowledge, and concern surrounding the outbreak, the greater the support for intervention (Qin et al. 2015). For example, Colorado residents who were asked about a recent widespread outbreak of the mountain pine beetle *(Dendroctonus ponderosae)* had different views of forest management responses (Flint et al. 2012).

In lower amenity communities (having greater employment in forestry and agriculture) with higher tree mortality, residents were less satisfied with, and less trusting of, the management responses of government land managers (especially the Forest Service and city government) than residents of higher amenity communities where tree mortality was not as severe. The latter were less likely to support aggressive or industrial timber harvest as a solution to the problem because of the high value they placed on recreation and scenic quality (Flint et al. 2012). But views of forest management in response to insect disturbance can change over time.

In Colorado, residents of higher amenity communities, resurveyed 10 years later, had become more supportive of addressing beetle-kill impacts with proactive management, including harvesting live and dead trees affected by the beetle and clearcutting because of concerns about wildfire risk and desire for a healthy forest (Vickery et al. 2020). Similarly, public satisfaction with forest manager responses to spruce beetle *(Dendroctonus rufipennis)* infestations on Alaska's Kenai Peninsula (including those on Forest Service lands) also improved overall with time (Qin et al. 2015). This literature indicates that forest management actions to mitigate insect disturbances on public lands should not be one-size-fits-all but different in different places, depending on local community context, in order to garner public support.

We suggest that managing for resilient forest landscapes is a construct that strongly depends on scale and social values. It involves human community changes and adaptations that are concordant with the ecosystems they depend on. It entails exploiting factors and mechanisms that drive dynamics at each level as a means of adapting landscapes, species, and human communities to climate change, and maintaining core ecosystem functions, processes, and services. Finally, it compels us to prioritize management that incorporates ongoing disturbances and the anticipated effects of climatic change and supports dynamically shifting patchworks of forest and nonforest. Doing so will make these shifting forest conditions and wildfire regimes more gradual and less disruptive to individuals and society.

A New Vision for Landscapes

The National Forest Management Act requires the U.S. Forest Service to abide by federal regulations that guide the development and implementation of land and resource management plans, also known as forest plans. In 2012, the Forest Service amended these regulations.

The 2012 planning rule represents an important shift in national forest management policy (Schultz et al. 2013). It emphasizes whole ecosystem thinking in planning and management, and the critical role of biodiversity and forest productivity conservation, but not to the exclusion of, or with persistent damage to, other ecosystem values and services. The 2012 rule also emphasizes collaborative planning and climate change adaptation as primary goals; outcome- rather than output-based management as the overarching intention of forest- and project-level plans; and managing for native species, including those at risk and their habitats within the broader context of whole systems thinking. In addition, the 2012 rule emphasizes reducing wildfire vulnerability and increasing landscape resilience using lessons learned from historical landscape conditions as a way of guiding planning. The 2012 planning rule is a new vision for federal forests, and it will be influential to forest plan revision in Oregon and Washington.

Recent ecological and social science research establishes the need for understanding and planning for ecosystems as social-ecological systems (Spies et al. 2014, 2018b). This is a large and highly relevant breakthrough in managing ecosystems. If no humans were present—making demands or having expectations of ecosystems—only ecological data and inferences would be relevant. However, that is not our circumstance. People have lived in, influenced, managed, and had expectations of western ecosystems for more than 10,000 years, and as populations expand, expectations and anthropogenic effects are on the rise, as is public interest in federal forest management.

Recent ecological and social science research establishes the need for understanding and planning for ecosystems as social-ecological systems.

In these social-ecological systems, research shows that federal land managers and policymakers could benefit by broadening their social networks beyond those that are characteristically in play (Fischer et al. 2016), potentially bridging institutional barriers, moving toward co-planned and co-managed outcomes, and increasing people's understanding of the social and ecological tradeoffs of different management conditions and actions, including no action. This likely leads to an even larger role for collaborative groups and stakeholders in forest- and project-level planning, implementation, and monitoring. Only in this context can all involved in planning have a clear understanding of the available data, extant conditions, likely changes and dominant influences, tradeoffs among goals, and likelihoods and uncertainties surrounding outcomes. Understanding and sharing the weight of uncertainties is a large part of trust development and maintenance among partner organizations and stakeholders, and for managing dynamic landscapes.

Box 25

Long-Term Utility of the 21-Inch Rule

The 21-inch rule is a policy tool that was designed to meet a need in 1994 (Powell 2013): halting the logging of large trees in eastern Oregon and Washington. Large and old trees have both ecological and social values, and there is widespread public support for protecting them and for active management to restore them on the landscape. However, the 21-inch rule may not allow managers to consider advances over the past 25 years in ecology, conservation science, and social science, and may limit manager flexibility to reach evolving restoration goals where flexibility is key. For example, it does not provide protection for older but smaller trees that may play an important ecological role. Nor does it allow for removal of young but large shade-tolerant trees that are maladapted to the existing fire regime and that developed over the period of fire exclusion. Hence, retaining these trees can be inconsistent with the desired future conditions recommended by new climate- and wildfire-adaptation science.

The science to date shows us that the only constant in social-ecological systems is change—given evolving climate, landscape, and social dynamics. In this light, the ability to operationalize adaptive management—literally, rapid learning by doing, and rapid doing based on learning—is essential to success. Inflexible plans and administrative or operational constraints will likely fail over time.

In addition, the 21-inch rule does not provide for managing complex social-ecological systems in which scale, feedbacks, and cultural values of old trees are important. Tree diameter alone is an insufficient guide for restoration, and for managing landscapes for resilience to climate change and related stressors. Focusing on a single scale (the tree) does not address stand- (within patch heterogeneity) and landscape-scale (forest successional patchwork) considerations that are critical for meeting multiple ecological and social goals.

Tree diameter alone is an insufficient guide for restoration, and for managing landscapes for resilience to climate change and related stressors.

The past 25 years of research illustrate the importance of numerous considerations and tradeoffs in management of east-side forests. Goals to achieve include but are not limited to the following:

- **Human values**—Engaging real opportunities to realize ecological and social values centered around conservation of large and old trees and old forests.

- **Collaboration**—Building and maintaining trust and a common vision for landscapes and ecoregions among stakeholders.
- **Economic values**—Restoration efforts are consistent with and restorative of nondeclining yield conditions.
- **Nonforest** (grass, shrub, herb, sparse woodland, savannah) —Current and future nonforest abundance and patch sizes are characteristic for this landscape and ecoregion, and suitable for the evolving climate.
- **Forest**—Current and future abundance and patch sizes of forest successional patches are characteristic for dry, moist, and cold forests, for this landscape, for the ecoregion, and for the evolving climate.
- **Relations with topography**—The current distribution of dry, moist, and cold forest successional patches aptly considers topoedaphic setting.
- **Disturbance**—Current and future expected wildfire, and insect and pathogen disturbances and their variability, will be characteristic for this landscape and ecoregion, considering ongoing climatic changes and supportive forest and nonforest successional patterns.
- **Tree species**—Choices of tree species are adequately adapted to the coming climatic and disturbance regimes.
- **Tree age**—Patterning tree age across patches and landscapes with sustainable landscape genetics, wildlife habitats, improved wildfire and climate vulnerability, and biotic diversity in mind.
- **Invasive species**—Management choices will minimize or mitigate further spread of invasive or nonnative species.
- **Carbon sequestration**—Management choices value potential carbon sequestration by forests and reduced uncertainty of long-term storage.
- **Wood in streams**—Manage terrestrial landscape patches that have direct hydrologic continuity with rivers and streams in a manner that delivers future large wood.

A rule that considers all such factors is not possible, given the complexity and variability of forests and landscapes and their rapidly evolving dynamics. However, the new research summarized here outlines areas of evolving science that could be incorporated into east-side forest planning. For example, forest plan guidance on stand and landscape management could include advice on tree species and ages to prefer, provisions that incorporate increased understanding of climate resilience goals, and consideration of social values such as those listed above.

The science presented here underscores the importance of management strategies that allow for consideration of multiple criteria, multiple spatial and ecological scales, social-ecological and geographic variation, and the opportunity for a degree of flexibility.

Plant Species Identified in This Report

Scientific name	Common name
Abies concolor (Gord. & Glend.) Lindl. ex Hildebr.	White fir
Abies grandis (Douglas ex D. Don) Lindl.	Grand fir
Abies lasiocarpa (Hook.) Nutt.	Subalpine fir
Arceuthobium americanum Nutt. ex Engelm.	American dwarf mistletoe
Arceuthobium campylopodum Engelm.	Western dwarf mistletoe
Arceuthobium douglasii Engelm.	Douglas-fir dwarf mistletoe
Arceuthobium laricis (Piper) H. St.	Larch dwarf mistletoe
Bromus tectorum L.	Cheatgrass
Festuca idahoensis Elmer	Idaho fescue
Juniperus spp.	Juniper
Larix occidentalis Nutt.	Western larch
Picea engelmannii Parry ex. Engelm.	Engelmann spruce
Pinus contorta Douglas ex Loudon	Lodgepole pine
Pinus lambertiana Douglas	Sugar pine
Pinus ponderosa	Ponderosa pine
Pseudotsuga menziesii (Mirb.) Franco	Douglas-fir
Quercus kelloggii Newberry	California black oak
Taeniatherum caput-medusae (L.) Nevski	Medusahead
Tsuga canadensis (L.) Carrière	Eastern hemlock
Vaccinium membranaceum Douglas ex Torr.	Thinleaf huckleberry
Ventenata dubia (Leers) Coss.	North Africa grass

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Metric Equivalents

If you have:	Multiply by:	To get:
Inches	2.54	Centimeters
Acres (ac)	2.47	Hectares
Tons per acre	2.24	Metric tons per hectare
Degrees Fahrenheit	.56(°F – 32)	Degrees Celsius

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