Climate and wildfire adaptation of inland Northwest US forests

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After a century of intensive logging, federal forest management policies were developed in the 1990s to protect remaining large trees and old forests in the western US. Today, due to rapidly changing ecological conditions, new threats and uncertainties, and scientific advancements, some policy provisions are being re-evaluated in interior Oregon and Washington. The case for re-evaluation is clearest where small- to large-sized, immature, fast-growing, fire-intolerant trees have filled in forests after both a long period of fire exclusion and the harvest of large, old trees. This infilling has created abundant fuel ladders that increase patch and landscape vulnerability to severe wildfires, which now threaten many forests. As climate change continues to alter fire regimes, we recommend that landscape-level planning is needed to determine where fire-tolerant and intolerant forest successional conditions are best retained on the landscape. Critical to our proposal are effective public engagement, collaboration, and tribal consultation.

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In this review, we describe how a progression of 19th- to 21st-century sociopolitical activities and management practices affected dry pine forests as well as dry and moist mixed-conifer forests of the US Pacific Northwest (PNW) region. These forests reside east of the crest of the Cascade Mountain Range in Oregon and Washington, and in the Klamath Mountains of southwestern Oregon and northern California. We propose that landscape-level adaptation is warranted to address broadly altered forest successional conditions, 20th- and 21st-century climatic changes, and rapidly changing wildfire regimes in the modern era.

In a nutshell:

- Sociopolitical and forest management activities of the past two centuries have dramatically altered inland Northwest forest area, density, and species composition
- Harvest of large, old, fire-tolerant trees and fire exclusion associated with multiple factors were chief among the influences
- These changes have predisposed modern forests to severe drought, insect, and wildfire events
- Removal of fire-intolerant trees and fuel ladders that accumulated during the period of fire exclusion is vital to adapting current landscapes to altered fire regimes and climatic warming
- Resulting conditions will be better adapted to wildfires and more resilient to climatic changes

During the hot, dry summer of 1910, hurricane-force winds drove lightning-ignited August wildfires throughout northwestern Montana, northern Idaho, and northeastern Washington, burning over three million acres and causing the deaths of 85 people, 78 of whom were firefighters. After this "Big Burn", wildfires were perceived to be purely destructive to western landscapes, and policies were developed to ensure that future fires were quickly extinguished (Pyne 2001). Thereafter, federal forest management featured aggressive fire suppression to protect people, infrastructure, and forests. High suppression efficacy and reduced burned area were clearly noticeable by 1935 (Westerling *et al.* 2006) with the advent of the "10-am rule" (that is, fires were extinguished by 10 am the morning after detection; Hessburg and Agee 2003). At the same time, large, old, fire-tolerant trees – which contained the most timber volume – were logged in pine forests as well as in dry and moist mixed-conifer forests of Oregon and Washington (Langston 1995). Harvests intensified greatly after World War II as a result of the increased availability of gas-powered engines.

Public aversion to the removal of large trees and old forests increased during the 1970s and 1980s as the pace and consequences of 20th-century logging became apparent (Langston 1995). Societal values had shifted; industrial-style logging had fallen from favor, and there was limited social license for further harvest of large, old trees, much less expansive harvest units. To many, old forests became iconic and were valued for their intrinsic, recreational, aesthetic, spiritual, and ecological importance (Hessburg *et al.* 2020). To Native American tribal

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groups, old forests were valued for providing foods and medicines, maintaining spiritual and cultural practices, and creating intergenerational connections to traditional practices (Long *et al*. 2018).

Due to the increased magnitude of public concern over large tree harvesting on public lands, new policies were crafted in the early 1990s that applied harvest restrictions to large and diverse ecoregions (Powell 2013). In inland PNW forests, one new policy would prohibit the harvest of any tree larger than 53-cm diameter at breast height (dbh), which was considered at the time to be a convenient lower cutoff for old trees and forests. Policy application improved the transparency of forest management as applied to large trees while easing public concerns over large tree harvesting; however, it took neither tree age and species nor rural community needs into account (Hessburg *et al*. 2020). Accordingly, application of the large tree policy would further contribute to increasing density and layering of shade-tolerant but fire-intolerant trees, which increased the likelihood of crown fire initiation and spread in forests where this had historically been an infrequent occurrence (Hessburg *et al*. 2020; Hagmann *et al*. 2021).

Similarly, the large tree policy did not anticipate the effects of rapidly unfolding changes on the PNW landscape. Wildfire suppression practices were highly effective until around 1985, when burned area unambiguously increased, despite growing suppression effort (Calkin *et al*. 2015). Increasing burned area after 1985 was driven by climate warming, reduced mountain snowpack, earlier spring warming, longer fire seasons, and increasing likelihood of drought (Westerling *et al*. 2006). Aggressive fire suppression – undertaken to facilitate community, resource, and habitat protection – steadily became more costly and ineffective (North *et al*. 2015) as forest fuel build-up and drought conditions worsened.

In the meantime, logging had removed many of the largest and oldest fire-tolerant trees, while intolerant, shade-loving trees with low canopy bases filled in gaps left by removals (Hessburg *et al*. 2005; Hagmann *et al*. 2021). After more than 170 years of fire exclusion, many fire-intolerant trees have grown into larger size classes (Figure 1). Fire exclusion did not begin with fire suppression, but began in the mid-1800s with widespread domestic livestock grazing and the loss of Native American cultural burning (Otis 2014). Indigenous peoples had settled this region at least 10 millennia prior to European colonization; during this period, they augmented natural ignitions with intentional burning. In many areas, the effects of intentional burning were substantial, typically reducing the likelihood of large and severe events (Pyne 1997). For example, Taylor *et al*. (2016) showed that tribal burning in California's Sierra Nevada mountains was of such frequency and extent that it minimized the influence of climate controls on burned area and severity. Their research demonstrated that the effects of fire exclusion were first amplified by the removal of extensive intentional burning and illustrated the potential benefits of intentional burning on reducing uncertainty, increasing likelihood of fires of lesser severity, and reducing burned area. In addition, it underscores the need to improve the accuracy of fire–climate model predictions by incorporating human influences on ignitions, the built environment, and vegetation management.

At present, in most forested patches affected by the loss of intentional burning and fire exclusion, there are many more shade-tolerant trees overall (Larson and Churchill 2012; Churchill *et al*. 2013), and many more trees >40 cm dbh than occurred under historical disturbance regimes (Figure 1; a "patch" is any subunit of a landscape with relatively homogeneous structure and composition). Consequently, affected forests are often homogeneous, dense, and layered (Figure 2), with fuel ladders that extend from near ground level to the crowns of remaining large, old, fire-tolerant trees (Figure 3; Agee and Skinner 2005). Wildfire and drought are continuing threats to these forests (Stephens *et al*. 2020; Hagmann *et al*. 2021) and the biota that depend on them, a situation that compels many fire and climate scientists to ask (eg Krofcheck *et al*. 2018; Stephens *et al*. 2020) whether broad modification of 21st-century landscape structure and composition – if that is even possible – is necessary to promote climate- and wildfire-adapted conditions and restore habitat diversity.

What have we learned?

Much has transpired in research and management since the 1990s. As knowledge has advanced and ecological conditions have changed, new threats and uncertainties have emerged. For example, we now recognize that fire exclusion has created conditions that are more vulnerable to disturbances (Hagmann *et al*. 2021), especially given projected increases in severe wildfires and droughts (Westerling 2016). We also better understand that while many large trees provide important ecological functions, all large trees are not ecologically equivalent (van Pelt 2008), and that tree size is an unreliable indicator of tree age, as some old trees are small and some young trees large in diameter (Brown *et al*. 2019). Likewise, the tree species retained influences adaptability to wildfires and climatic warming (Stevens *et al*. 2020).

As indicated above, many large (≥50-cm dbh) trees are relatively immature, fast-growing, shade-tolerant, and fire-intolerant, and established in response to early selective harvests (Figure 1), the shade offered by residual trees, and >170 years of fire exclusion; examples include grand fir (*Abies grandis*), white fir (*Abies concolor*), and immature Douglas fir (*Pseudotsuga menziesii*) (North *et al*. 2015; Stephens *et al*. 2018). Note that these three species are fire-intolerant until they mature and acquire a relatively thick bark. They were historically minor associates of most frequently burned inland PNW forests (eg Lillybridge *et al*. 1995); however, their retention today maintains a continuous seed rain of these species, allowing them to quickly regenerate after moderate- or light-touch fires or active management (Spies *et al*. 2018a, 2019). Moreover, dense areas of these species are vulnerable to certain bark beetles, dwarf mistletoes, and tree-killing root diseases (Fettig *et al.* 2007; Sturrock *et al.* 2011). Consequently, some of the most inherently fire-resilient species, including ponderosa pine (*Pinus ponderosa*), western larch (*Larix occidentalis*), and large-sized Douglas fir, are now threatened with overcrowding from fire-intolerant tree species and size classes that have encroached across the landscape (Figure 3; Stephens *et al.* 2018).

Fire-tolerant species generally require exposed mineral soil created by fires (or other site disturbance) to regenerate from seed (Burns and Honkala 1990). As these trees age in the presence of fires, they develop flame-avoidant, progressively elevated crown bases and thickening flame-resistant bark (Agee and Skinner 2005). Fire-tolerant species at lower densities are also better adapted to changing climatic and wildfire regimes, prescribed burning, and managed wildfires (Stevens *et al.* 2020; Prichard *et al.* 2021), and exhibit reduced vulnerability to insect attacks (Fettig *et al.* 2007). While wildfires in fire-frequent forests of the past minimized overcrowding and favored larger tree sizes and fire-tolerant trees (Agee and Skinner 2005), wildfires today often result in stand replacement, with many patches struggling to regenerate in the absence of nearby early seral seed trees (eg see Stevens-Rumann *et al.* 2018; Davis *et al.* 2019; Coop *et al.* 2020).

Numerous studies have shown that reducing tree density and favoring fire- and drought-tolerant species can reduce stresses on residual trees (Stephens *et al.* 2020; Prichard *et al.* 2021) and increase forest resistance and resilience (sensu Hessburg *et al.* 2019) to wildfire, insects, and extreme climatic events (Bradford and Bell 2017). Landscape-level planning is needed to determine where fire-tolerant and intolerant species, as well as open and closed canopy forest successional conditions, are best retained on the landscape as the modern climate continues to alter biophysical environments and wildfire regimes (Spies *et al.* 2019). For existing and recruited old forests, it will be important to manage them in the larger context of wildfire-resilient conditions to improve their protection (Spies *et al.* 2019; Meigs *et al.* 2020; Prichard *et al.* 2021). In fire-excluded pine, and dry and moist mixed-conifer patches – where most of the largest fire-tolerant trees were harvested – removing immature fire-intolerant trees (including large ones) and

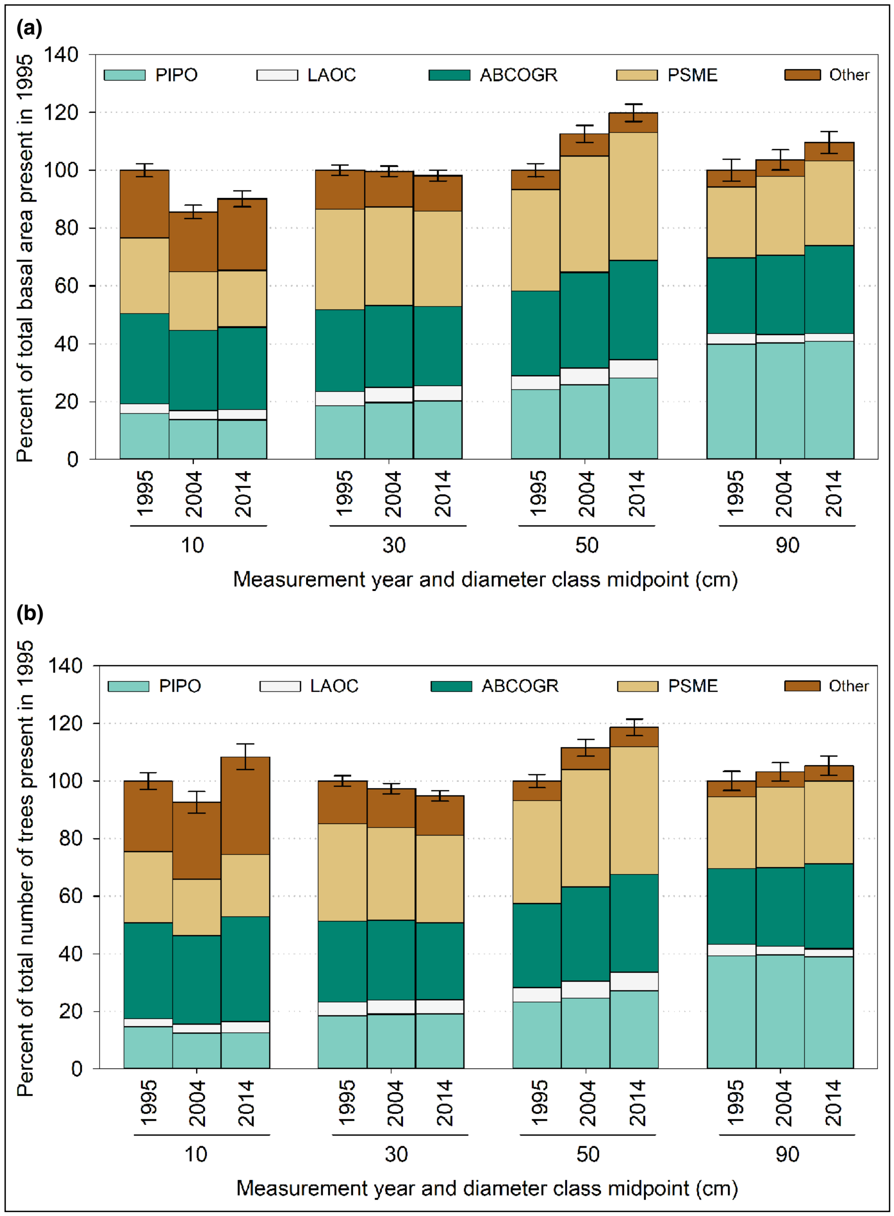

Figure 1. Species abundance has changed in mixed-conifer forests in recent decades. Estimates of (a) basal area per hectare (BAH) and (b) trees per hectare (TPH) on public lands in eastern Oregon and Washington were compiled from national and regional forest inventories. For each diameter class, bars from left to right represent estimates for midpoint inventory years 1995 (1990–1999), 2004 (2000–2007), and 2014 (2010–2017). Species include the fire-tolerant and shade-intolerant ponderosa pine (*Pinus ponderosa*, PIPO) and western larch (*Larix occidentalis*, LAOC); the shade-tolerant white fir (*Abies concolor*) or grand fir (*Abies grandis*) (together, ABCOGR) and Douglas fir (*Pseudotsuga menziesii*, PSME); and all other tree species combined (Other). Species proportions in 2004 and 2014 are relative to the BAH (above) and TPH totals (below) in 1995, for each diameter class. Diameter class midpoints are (left to right) 10 cm (range 2.5–20 cm), 30 cm (range 20–40 cm), 50 cm (range 40–60 cm), and 90 cm (range 60–120 cm). Results show overall increases in BAH and TPH for tree diameters >40 cm, with ABCOGR and PSME increasing more than PIPO, and LAOC generally declining. Error bars represent the standard error of the mean estimate. (a) The proportions of total BAH in the 10-cm, 30-cm, 50-cm, and 90-cm midpoint classes in 2014 were 16.9%, 35.1%, 27.5%, and 20.6%, respectively. (b) The proportions of total TPH in the 10-cm, 30-cm, 50-cm, and 90-cm midpoint classes in 2014 were 79.9%, 14.5%, 4.2%, and 1.4%, respectively. Note that the largest increases in BAH and TPH occurred in the 50-cm class.

(a) RR Sarlin; National Archives and Records Administration, Seattle

Figure 2. A densely treed landscape emerges. Panoramic photographs – taken from Duncan Hill, Washington, looking southeast along the Entiat River drainage to the Columbia River – show the majority of the 238,000-ha Entiat drainage in (a) 1934 and (b) 2012. Fire exclusion and selection cutting broadly homogenized successionally diverse pine forests, and dry and moist mixed-conifer forests. In the absence of wildfires, bark beetles kill trees, increase fuels, and synchronize large areas for burning.

Figure 3. View near Tronsen Ridge (in the background), 2013, Okanogan-Wenatchee National Forest, Washington. Only a handful of trees in this scene were present 125 to 150 years ago. The largest ponderosa pines are 300 to 400 years old, developing under a frequent fire regime. Most other trees are fire-intolerant grand fir and Douglas fir that established over the period of livestock grazing and fire exclusion. A few dwarf mistletoe infested younger western larch are dead in this scene owing to extreme intertree competition for soil moisture and nutrients, and mistletoe infection severity.

replanting with an adequate stocking, along with a clumped and gapped arrangement of fire-tolerant trees (sensu Larson and Churchill 2012; Churchill *et al.* 2013; LeFevre *et al.* 2020), would increase forest resilience to wildfires and insect outbreaks (Stephens *et al.* 2018). Restoring and/or recruiting open canopy conditions and medium to large fire-tolerant tree sizes in dry aspect and ridgetop pine and mixed-conifer forest patches would also promote habitats for sensitive wildlife species like the white-headed woodpecker (*Picoides albolarvatus*; Gaines *et al.* 2007) and other associates of frequent-fire open canopy forests (Russell *et al.* 2007).

Research has also demonstrated that policies in which the same rules are applied uniformly are inconsistent with management for both resilient landscapes and the processes that support them (eg Spies *et al.* 2018a, 2019). Across montane PNW landscapes, we find varied environmental gradients that can support wide-ranging forest structural and compositional patterns and their associated biodiversity (Hessburg *et al.* 2016, 2019). For example, mixed-conifer forests historically coexisted with frequent low- and moderate-severity fires, which produced widespread open forest conditions, whereas in cool and moist forest patches, environmental conditions readily supported denser forests with a wide variety of tree species, sizes, and densities, including those with large fire-intolerant trees and infrequent moderate- and high-severity fires (Perry *et al.* 2011). Management guidelines that provide flexibility to build on this topo-edaphic template will more likely succeed at providing the structural and compositional complexity that contributes to biodiversity, restored wildfire regimes, and other forest values (Taylor and Skinner 2003; Hessburg *et al.* 2016, 2020). Balancing multiple objectives, including conserving habitat for wildlife species that use shade-tolerant trees, continues to be important (Raphael *et al.* 2001; Singleton *et al.* 2010).

Improving the fire and climate resilience of certain patches

The context of rapidly changing climatic and wildfire regimes necessitates shifts in 21st-century management. Most western US forest scientists contend that extensive changes have occurred over the past 170+ years and recommend the use of mechanical treatments, prescribed fire, managed wildfire, and combinations of these – as appropriate to specific conditions – to restore forests' resilience to wildfire and climate (Stephens *et al.* 2013, 2020; Prichard *et al.* 2021). However, there are major financial, personnel, and infrastructure barriers to achieving widespread landscape adaptation, issues requiring attention by high-level managers and policy makers and that are beyond the scope of this review. There is also growing recognition that many contemporary wildfires and prescribed ignitions can be used to restore key aspects of forest resilience if fire behavior can be managed (North *et al.* 2012; Boisramé *et al.* 2017; Barros *et al.* 2018). Yet many areas within present-day wildfires burn with uncharacteristic severity (Parks and Abatzoglou 2020), killing large fire-tolerant trees and old forests that would have survived the lower severity fires of the past (Stephens *et al.* 2013, 2018). Continued success at suppressing fires of low and moderate intensity contributes to this dynamic (North *et al.* 2015).

Much research-based evidence supports the view that changes in forest condition and fire regimes compound the effects of climatic warming and ongoing aggressive fire suppression. These combined effects are leading to regional vegetation conditions that often pass a resilience tipping point, beyond which large-scale conversion to a nonforest state occurs (see Tepley *et al.* 2017; Davis *et al.* 2019; Coop *et al.* 2020). Forests have indeed expanded into numerous fire-maintained historical nonforest areas; many historical nonforest patches could readily support either hardwood or conifer forest growth in the absence of fire. As a result, forested patches are denser and more extensive in many parts of western North America than they were historically (Hessburg *et al.* 2019). The variety of current forest successional conditions is likewise monotonous in many regions and forest types (Figure 2; Prichard *et al.* 2017). Adapting landscapes to climate change and future wildfires will require opening seasonally dry pine and dry and moist mixed-conifer forest canopies, especially on ridges and dry aspects, favoring fire-tolerant and larger-sized trees (Agee and Skinner 2005). It will also involve removing many small- to large-sized (20–60 cm dbh; Figure 1) and immature fire-intolerant tree species that create fuel ladders for crown fires and directly compete with fire-tolerant trees for moisture, nutrients, and growing space (Agee and Skinner 2005; Figure 3). Ultimately, promoting adaptation to climate change and future wildfire will require restoring broad variety in forest successional and nonforest conditions based on appropriate landscape setting (Perry *et al.* 2011; Hessburg *et al.* 2016, 2019). This more heterogeneous patchwork would regulate rate of fire spread, fireline intensity, flame length, and crown fire potential, and promote greater beta habitat diversity (Perry *et al.* 2011; Figure 4).

While halting the harvest of large and old trees was a critical initial step in conserving native forest biodiversity, the primary threats on public lands have changed from timber harvest to expanding areas of severe fire, invasive species, drought-stressed trees, and increasing insect and disease disturbances (Dale *et al.* 2001). Simple rules of protecting all large trees or keeping treatment patches small are insufficient to support a scientifically based adaptation policy, requiring the development of a more nuanced approach (Hessburg *et al.* 2020). Better definitions of old trees and forests in wildfire-prone regions are also needed (Spies *et al.* 2018a), crucial to which are knowledge of site and geographic conditions and fire history (Stephens *et al.* 2015; Spies *et al.* 2018a).

Federal managers could markedly improve the sustainability of native biodiversity and forest resilience to wildfires and future climate by: (1) *employing whole landscape thinking and geographically varying topo-edaphic templates* to determine where on the landscape fire-tolerant and intolerant successional conditions, including old forests, may best reside as climate and weather patterns alter the fire regime and redefine biophysical settings (Taylor and Skinner 2003; USFS 2012; Hessburg *et al.* 2015); (2) *using fire through prescribed burning or managed wildfire* (under appropriate fuel and weather conditions) to promote desired stand structure and species composition (Graham *et al.* 2004; North *et al.* 2012; Barros *et al.* 2018); (3) *reintroducing fire-tolerant early seral species* by hand planting or direct seeding where they have been removed from the landscape and no longer dominate the seed rain (Agee and Skinner 2005; Hessburg *et al.* 2005, 2020); (4) *retaining existing old trees and forests and developing more of them* in appropriate biophysical settings, where harvests have diminished their presence (Churchill *et al.* 2013; Stephens *et al.* 2015); (5) *using age rather than diameter as a criterion* in tree retention decisions (van Pelt 2008; Franklin and Johnson 2012; Franklin *et al.* 2018) (van Pelt [2008] provides distinctive visual clues for rapid age estimation in the field); (6) *creating treatment patches that match the topographic template*, with feathered edges as was typical after historical wildfire and insect disturbances (Matlack and Litvaitis 1999; Taylor and Skinner 2003; Hessburg *et al.* 2015); (7) *favoring removal of fire-intolerant species* across the size classes that contribute to fuel ladders and overly dense forests, and where they dominate the seed-rain on

(a) RL Cooper; National Archives and Records Administration, Seattle

Figure 4. Young shade-tolerant and fire-intolerant trees rapidly fill in the forest. Over 150 years, a dry and moist mixed-conifer forest landscape has become densely filled with Douglas fir and grand fir pole-, small-, medium-, and large-sized trees. View looking southwest into Stafford Creek, North Fork Teanaway River watershed, Cle Elum, Washington, in (a) 1934 and (b) 2013.

pine and mixed-conifer sites (Graham *et al*. 2004; Agee and Skinner 2005); (8) *thinning smaller ponderosa pine and western larch in places with ample stocking*, and where current tree density is vulnerable to severe disturbances under future climate and wildfire conditions (Pollet and Omi 2002; Graham *et al*. 2004); and (9) *developing and funding transparent implementation and effectiveness monitoring protocols* so that stakeholders and partners can see how managers are implementing new policies, practices, and demonstration projects, and observe the manner of ecosystem responses (White *et al*. 2015).

There is support for active management on public lands to confront current anthropogenic climate and wildfire regime changes (Burns and Cheng 2007; McCaffrey *et al*. 2013); however, there remain social and ecological challenges to widely applying forest adaptation treatments. For example, with increasing 21st-century global greenhouse-gas emissions comes public and scientific support for *increasing* forested area and forest density to improve carbon (C) sequestration (see Domke *et al*. 2020). Yet in many western forests, increased forest density and area contribute to current wildfire vulnerability and instability of forest sector C stocks (eg Hurteau *et al*. 2008; Liang *et al*. 2018). Similarly, although prescribed burns and managed wildfire can reduce future wildfire vulnerability in many forests, such actions generate smoke emissions harmful to human health (Penman *et al*. 2011; Higuera *et al*. 2019), though they pale in comparison to those of wildfires. Clear understanding of these interactions, in addition to their relative magnitude and associations with positive and negative feedbacks, is essential to public debate and policy decisions. Nevertheless, social acceptability of forest adaptation treatments is greatest when members of the public perceive high wildfire risk and poor forest health, are familiar with the proposed treatment techniques, believe treatments will achieve desired outcomes and avoid undesirable ones, and trust the implementing agencies (McCaffrey *et al*. 2013).

Several Native American tribes express support for harvesting large trees. For example, Klamath Tribes in southern Oregon and northern California have expressed concern that young, fast-growing, fire-intolerant trees are displacing hardwoods in areas of cultural value (Kimmerer and Lake 2001; Lake 2007; Johnson *et al*. 2008). In their plans to restore pine and oak (*Quercus* spp) woodlands, they recommend that age- rather than size-based thresholds serve as the basis for protecting large trees. Adding large immature trees to harvest mixes improves the financial viability of adaptation treatments on public lands, but if stakeholders perceive that treatments are being driven by commercial rather than ecological considerations, they will likely be contested, especially where large-sized tree harvests are involved (Stidham and Simon-Brown 2011).

Whether large or small fire-intolerant trees might be removed ultimately depends on context, clear consideration of management and fire histories, adaptation goals, site and local landscape conditions, multi-party monitoring, and social license (Spies *et al*. 2018b). Objectives like maintaining habitats for sensitive species are also key considerations (Franklin and Lindenmayer 2009). Simple solutions are rarely ecologically or socially sound (Spies *et al*. 2018b). Removing large fire-tolerant trees like western larch, ponderosa, western white pine (*Pinus monticola*), and sugar pine (*Pinus lambertiana*) can reduce forest resilience to wildfire and climate change (Agee and Skinner 2005). Where necessary, however, removing small to large immature shade-tolerant species like grand fir, white fir, or immature Douglas fir can increase resilience (Graham *et al*. 2004; Stephens *et al*. 2018, 2020; Figure 3).

A potential path forward

Seasonally dry forests of pre-management era landscapes were dynamically shifting, diverse patchworks of forest and nonforest successional conditions; no two landscapes were alike (Keane *et al*. 2009; Wiens *et al*. 2012). Native plants and animals persisted in this dynamism. Climate and wildfire adaptation of modern landscapes will require creating a forward-looking form of this dynamism (Keane *et al*. 2009; Wiens *et al*. 2012), and there will be continuing trade-offs, just as in the pre-management era landscape, where changes in climate and biophysical settings gave rise to changes in processes and the shifting patterns that supported them.

In wildfire environments, climate-adapted landscapes will generally not maximize forest cover, growth, density, or C storage (Hurteau *et al*. 2008; Liang *et al*. 2018). A continual tug-of-war wages between (1) biogeoclimatic factors that promote forest development and (2) climatic, environmental, and disturbance factors that eliminate forest cover. Disturbances of all sizes and intensities will occur, including those that kill or remove large and small trees, thereby reducing forest cover, shading, and microsite buffering at some sites. The continued ebb and flow of disturbances across space and time, and the resulting forest conditions, will favor certain species, habitats, and processes, and not others; however, this too is in constant flux. For example, in some seasonally dry Oregon and Washington forests, the northern spotted owl (*Strix occidentalis caurina*) often relies on habitats where young Douglas fir and grand fir have encroached into formerly open-canopy maturing or park-like old forest patches of ponderosa pine (Figure 3; Hagmann *et al*. 2017). These more open-canopy pine-dominated conditions historically supported the white-headed woodpecker, now a sensitive species (Buchanan *et al*. 2003; Gaines *et al*. 2007).

Managing for resilient forest landscapes is a construct that strongly depends on scale and social values, and that involves human community changes and adaptations that are concordant with the ecosystems they depend on (Hessburg *et al*. 2020). It entails building on ecological and social factors and mechanisms that drive dynamics as a means of adapting landscapes, species, and human communities to climate change, while maintaining, to the best extent practicable, core ecosystem processes and services (Spies *et al*. 2014, 2018b). It compels us to prioritize management that incorporates appropriate active disturbance regimes to each forest and nonforest type. It obliges us to anticipate the effects of wildfires and climatic changes so as to support dynamically shifting patchworks of forest and nonforest that are in synchrony with the climate and the affected biophysical settings. Doing so will make the transformation of forest conditions and wildfire regimes less disruptive to species and society (Spies *et al*. 2014).

Decision making in public forest management involves weighing and balancing trade-offs (Fischer *et al*. 2016). Potentially contentious issues like tree removal or managed wildfire call for dialogue, skill development, and trust building among managers, collaborators, and stakeholders (Davis *et al*. 2017). Forest collaborative groups consisting of diverse managers, regulators, tribes, nongovernmental organizations, and citizens have become prevalent over the past two decades throughout the western US. Collaboration provides an avenue for forging agreement about harvest and burning activities in specific locations, but the details matter (Fischer *et al*. 2016). Among some collaborative groups, agreement has been reached about harvesting small to large white fir, while protecting large and old pines (eg Davis *et al*. 2018). In addition, there is greater support for removal treatments if harvested trees are processed locally, thereby generating economic benefits for local communities. Whether to promote climate and wildfire adaptation in inland PNW forests will be as much a social question as an ecological one (Fischer *et al*. 2016). Our suggested climate and wildfire adaptation actions will be difficult to embrace for those who simply oppose tree harvesting or prefer retaining shade-tolerant species.

At present, federal land management agencies struggle with garnering the social capital needed to adapt forests and increase resilience to wildfire, climate, and environmental changes. Policies and planning standards that clearly articulate adaptation goals will aid in forging agreements. Effective public engagement, collaboration, and tribal consultation to develop a shared vision for management are critical to building and maintaining trust. Demonstration projects that build "zones of agreement" and assent to adaptation and monitoring goals can facilitate progress. If agencies move ahead to adapt existing policies without forging trust and agreement, success is unlikely.

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