esa

ECOSPHERE

INNOVATIVE VIEWPOINTS

The nature of the beast: examining climate adaptation options in forests with stand-replacing fire regimes

Joshua S. Halofsky,[1,]† Daniel C. Donato,[1,2] Jerry F. Franklin,[2]
Jessica E. Halofsky,[2] David L. Peterson,[3] and Brian J. Harvey[2]

[1]*Washington State Department of Natural Resources, 1111 Washington Street SE, Box 47014, Olympia, Washington 98504 USA*
[2]*School of Environmental and Forest Sciences, University of Washington, P.O. Box 352100, Seattle, Washington 98195 USA*
[3]*U.S. Forest Service, Pacific Northwest Research Station, 400 N 34th Street, Seattle, Washington 98103 USA*

Citation: Halofsky, J. S., D. C. Donato, J. F. Franklin, J. E. Halofsky, D. L. Peterson, and B. J. Harvey. 2018. The nature of the beast: examining climate adaptation options in forests with stand-replacing fire regimes. Ecosphere 9(3):e02140. 10.1002/ecs2.2140

Abstract. Building resilience to natural disturbances is a key to managing forests for adaptation to climate change. To date, most climate adaptation guidance has focused on recommendations for frequent-fire forests, leaving few published guidelines for forests that naturally experience infrequent, stand-replacing wildfires. Because most such forests are inherently resilient to stand-replacing disturbances, and burn severity mosaics are largely indifferent to manipulations of stand structure (i.e., weather-driven, rather than fuel-driven fire regimes), we posit that pre-fire climate adaptation options are generally fewer in these regimes relative to others. Outside of areas of high human value, stand-scale fuel treatments commonly emphasized for other forest types would undermine many of the functions, ecosystem services, and other values for which these forests are known. For stand-replacing disturbance regimes, we propose that (1) managed wildfire use (e.g., allowing natural fires to burn under moderate conditions) can be a useful strategy as in other forest types, but likely confers fewer benefits to long-term forest resilience and climate adaptation, while carrying greater socio-ecological risks; (2) reasoned fire exclusion (i.e., the suppression component of a managed wildfire program) can be an appropriate strategy to maintain certain ecosystem conditions and services in the face of change, being more ecologically justifiable in long-interval fire regimes and producing fewer of the negative consequences than in frequent-fire regimes; (3) low-risk pre-disturbance adaptation options are few, but the most promising approaches emphasize fundamental conservation biology principles to create a safe operating space for the system to respond to change (e.g., maintaining heterogeneity across scales and minimizing stressors); and (4) post-disturbance conditions are the primary opportunity to implement adaptation strategies (such as protecting live tree legacies and testing new regeneration methods), providing crucial learning opportunities. This approach will provide greater context and understanding of these systems for ecologists and resource managers, stimulate future development of adaptation strategies, and illustrate why public expectations for climate adaptation in these forests will differ from those for frequent-fire forests.

Key words: climate adaptation; high-severity fire regime; maritime forests; post-disturbance management; resilience; subalpine forests.

Received 12 December 2017; revised 2 February 2018; accepted 7 February 2018. Corresponding Editor: Debra P. C. Peters.

† **E-mail:** joshua.halofsky@dnr.wa.gov

Introduction

Managing forests for adaptation to climate change has become a pressing issue for natural resource scientists and managers. Although it is widely recognized that fostering resilience to natural disturbances is a key management objective (Turner 2010, Hessburg et al. 2015), the strategies for doing so are uncertain for forests historically characterized by high-severity, stand-replacing disturbance regimes (Stephens et al. 2013). High-severity disturbances can be important drivers of ecological change in a warming climate (Turner 2010, Seidl et al. 2016, Crausbay et al. 2017) because they reset successional pathways and allow new vegetation establishment, but are also an intrinsic feature of some forests.

Most climate change adaptation literature for forests has a common emphasis: reducing disturbance (i.e., drought, fire, and insect outbreak) severity through manipulation of stand and landscape structure. A prominent example is thinning and prescribed burning in dry forests with historically frequent low-severity fire regimes, to reduce the potential for severe fires that may exceed the capacity of those systems to recover (Agee and Skinner 2005, Peterson et al. 2005, North et al. 2012, Hessburg et al. 2015). This type of strategy so dominates the literature that few adaptation options have been described for other forest types, especially those in which disturbances are driven less by forest structure (fuels) and more by climate and weather, such as many cool or moist temperate forest types (but see Schoennagel et al. 2004, Millar et al. 2007, Halofsky et al. 2011, Raymond et al. 2014). How do common disturbance mitigation and management concepts apply in systems where severe disturbances are largely indifferent to manipulations of stand structure, are within normal system behavior, and from which biota naturally have the capacity to recover?

Public land agencies and private landowners are currently grappling with notions of ecological restoration and climate adaptation in forests with stand-replacing fire regimes, either explicitly or implicitly (WFRS 2017), and these issues have relevance in comparable social–ecological systems globally. Recent papers have made strides toward broadening the fire-adaptation dialogue beyond frequent-fire systems (Schoennagel et al. 2017), but recommendations specifically for stand-replacing fire regimes are still needed because of the imperative to anticipate and adapt to future climate change effects (Flannigan et al. 2009, Moritz et al. 2014, Barbero et al. 2015, Westerling 2016).

Our objective here is to stimulate a dialogue on climate change adaptation strategies specific to forests characterized primarily by stand-replacing disturbance regimes. We focus on fire, a key ecosystem-structuring agent across many temperate forests, and an agent that can (to varying degrees) be influenced by forest management. In forests where large stand-replacing fire patches are a primary component of natural disturbance regimes, we propose that:

1. Climate adaptation options prior to wildfire are generally fewer than in other forest types, in part because common approaches to mitigating fire severities will likely be riskier, ineffective in the long-term, or even counterproductive to many management objectives.
2. Managed wildfire use programs (managing natural fires under moderate conditions rather than suppressing all fires) can be beneficial in creating landscape diversity, burn mosaics, and early-successional habitat, especially in the short term; however, the long-term benefits to forest resilience are likely fewer, and the socio-ecological risks greater, than in other systems.
3. Reasoned fire exclusion (i.e., the suppression component of a managed wildfire use program) can be useful in maintaining specific ecosystem conditions and services in these systems, having fewer long-term negative consequences than in frequent-fire forest types.
4. Depending on management objectives, lower risk pre-disturbance adaptation strategies are mostly grounded in basic principles of conservation biology (e.g., increasing structural and compositional diversity in stands and across landscapes, while minimizing stressors such as fragmentation and invasive species).
5. Post-disturbance situations, arising from both wildfire and management, can be used as adaptation and learning opportunities, especially regarding resilience and adaptive capacity.

We develop these concepts below, first with two foundational principles, followed by three broad strategies.

Scope

To establish a common language, we define our focal forests as those in which large patches of high-mortality (>90% of overstory trees) wildfire have historically been a primary mechanism of forest establishment. Such forests typically have inter-disturbance intervals at the stand or local watershed scale measured in centuries (Romme 1982, Agee 1993, Schoennagel et al. 2004, Keane et al. 2008); however, some systems may have shorter cycles of stand-replacement fire (Keeley et al. 1999, Harvey and Holzman 2014). Although virtually all fires in any forest type are mixed severity at some scale (Dillon et al. 2011, Harvey et al. 2016*a*, Reilly et al. 2017), the key distinguishing feature is high-severity patch size (Agee 1993, Collins et al. 2017). We focus on forests in which very large patches (e.g., 10^3–10^5 ha) of stand-replacement fire are common and well within the natural range of variability. As such, focal forests include both nominal stand-replacement regimes (while acknowledging those patches are part of a coarse-scale mosaic) and the cooler/wetter end of nominal mixed-severity regimes that include a component of large high-severity patches at long intervals (Morrison and Swanson 1990, Weisberg 2004, Tepley et al. 2013).

These forests occur in many moist- to wet-climate areas throughout temperate latitudes. To further facilitate understanding of our scope, we use the example of relevant forest types of the western United States (Fig. 1). These include portions of the maritime coniferous forests of the U.S. Pacific Northwest (i.e., much of the Douglas-fir/western hemlock region west of the Cascade Range crest; see Agee 1993) and subalpine conifer forests of the Rocky Mountains (Romme 1982, Schoennagel et al. 2004), among several others.

Although the forests we address vary in composition and structure, their fire regimes are generally considered more climate- and ignition-limited than fuel-limited (Agee 1993, Schoennagel et al. 2004). High productivity, slow wood decay rates, and long disturbance intervals relative to stand development trajectories—singly or in combination—produce dense, closed-canopy, biomass (fuel)-rich stand structures. As such, fire incidence, area burned, and fire severity are largely determined by the relatively infrequent co-occurrence of exceptionally dry conditions and an ignition source, and less determined by stand structure, which is usually conducive to fire spread and fire-caused mortality (Fig. 1). Forest regeneration after stand-replacing fire is typically robust in these systems (Turner et al. 1997, Larson and Franklin 2005, Donato et al. 2009, Freund et al. 2014, Harvey and Holzman 2014), meaning that forest cover is maintained over the long term even when subject to large, severe fire patches.

As with many temperate forests, climate change is projected to affect these systems through higher temperatures and altered precipitation regimes, as well as through altered disturbance regimes (Peterson et al. 2014). Temperatures are projected to increase across all seasons in both the Rockies and Pacific Northwest, while precipitation projections may diverge by region (Walsh et al. 2014). The Pacific Northwest will likely experience amplification of its normal wet-winter/dry-summer seasonality (Mote et al. 2014), while seasonality in the Rockies may differ between northern and southern portions of the region (Walsh et al. 2014).

Fire frequency, length of the fire season, and area burned will likely increase with climate change in our focal regions (Rogers et al. 2011, Mote et al. 2014, Loehman et al. 2018). The Rocky Mountain region is characterized by more continental climate than the Pacific Northwest, particularly east of the Continental Divide. Thus, fires in subalpine conifer forests of the Rocky Mountains are most limited by ignitions, whereas fires in maritime forests of the Pacific Northwest are primarily limited by high fuel moisture (McKenzie and Littell 2017). Ignitions may increase in a warmer climate (Romps et al. 2014), and fuel moisture will likely decrease in a warmer climate (McKenzie and Littell 2017), increasing fire frequency in forests with high-severity disturbance regimes. Annual area burned is projected to increase by 300–500% across much of the Pacific Northwest and 400–600% across much of the Rocky Mountain region (Mote et al. 2014). However, it is unclear whether these proportional increases will fundamentally

Fig. 1. Examples of stand-replacing systems where wildfires tend to be more limited by ignition source and weather rather than fuel. (A) A naturally regenerated lodgepole pine (*Pinus contorta* var. *latifolia*) patch ~10 yr post-fire, (B) an old-growth western red cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) stand (photograph used with permission, Van Pelt 2008), and (C) a multi-aged and partially managed Douglas-fir (*Pseudotsuga menziesii*) and noble fir (*Abies procera*) forest consisting of both recently harvested patches (right foreground and far background), 115 yr-old patches originating from a 1902 wildfire (left foreground), and old-growth patches that escaped the 1902 fire (above the recent harvest on right).

alter ecosystem composition and structure, because they start from a very low number (i.e., the low annual area burned of the current and/or historic eras). Moreover, because these forests are already characterized by high-severity fire, fire severity is not expected to significantly increase. Interactions among disturbances (e.g., fire, drought, and insect outbreaks) may also be more important than shifts in any single disturbance regime (McKenzie et al. 2008).

What sets these forests apart from others in terms of climate impacts is that they are already inherently characterized by many of the future climate projections for drier forest types: very large patches of severe fire effects and challenging conditions for forest re-establishment. In other words, projections of large, severe burns are already the nature of the beast.

Principles

Principle 1

Forests with natural stand-replacing fire regimes are inherently resilient to severe wildfires; therefore, increasing their long-term resilience through commonly used treatments (e.g., fuel and fire management) is difficult.

Here, we define resilience as the capacity to experience severe wildfire without shifting to an alternative ecosystem state over the long term (adopted from Walker et al. 2004), that is, the ability to regenerate back to a structurally, compositionally, and functionally similar forest following large, severe events. Variations of Principle 1 have been articulated for many forests ranging from wet coniferous forests to subalpine ecosystems (Bessie and Johnson 1995, Turner et al. 2003, Schoennagel et al. 2004, Keane et al. 2008, Mitchell et al. 2009, Stephens et al. 2012, Timpane-Padgham et al. 2017), but we expand on it here.

In essence, these forest types are adapted toward resilience to stand-replacing, pulse disturbance events, otherwise they would not exist in their current form. A natural tendency for large, high-mortality events, followed by robust regeneration and eventual return to a mature state, has led to these forests being called "boom-and-bust" systems (White et al. 2002). With regenerative traits well-suited to severe disturbances (i.e., serotiny, light well-dispersed seeds, long-lived seeds in soil bank), shifts toward increasing fire frequency (within broad limits) would likely cause quantitative shifts in species abundance and structure but not qualitative shifts to novel communities or physiognomies (Keeley et al. 1999, Schoennagel et al. 2003).

Limitations of stand-scale fuel treatments.— Adopting common management strategies to reduce wildfire severity (e.g., stand-scale fuel reduction treatments; Agee and Skinner 2005) faces a basic challenge: Many forests with stand-replacing fire regimes are not only rarely fuel-limited, but generally cannot be made fuel-limited without fundamentally changing them and creating stand structures with little or no ecological precedent (Brown et al. 2004, Schoennagel et al. 2004). In addition to compromising their fundamental nature and the ecosystem services that these forests provide, a fuel-mitigation approach would also face the Sisyphean task of frequent re-treatment to keep up with the high productivity rates in many regions. Thus, other than focused hazard reductions around areas of human value (Stephens et al. 2012), there is low ecological utility or practicality of fuel treatments in forests with stand-replacing fire regimes (Schoennagel et al. 2004, Noss et al. 2006).

Limitations of managed wildfire.— A second commonly promoted adaptive tool is managed wildfire, involving monitoring natural fires in certain locations with low risk to human values under moderate weather conditions, while suppressing fires outside of those locations and conditions. In many forests, this is a way to use wildfires to do much of the work of fuel management and build adaptive resilience (North et al. 2012, Millar and Stephenson 2015, Schoennagel et al. 2017) at spatial scales that are often impractical to attain by other means. This approach is well-supported in low- to mixed-severity fire regimes (Collins and Stephens 2007). Managed fire can also play a role in stand-replacing regimes, because even large fires in these systems have patches of low, mixed, and high severity and create significant landscape diversity (Turner et al. 2003), which may promote adaptive capacity in responses to large disturbances (further discussed in subsequent sections). However, it is important to recognize a different set of considerations and expectations of this approach in stand-replacing regimes.

Three common assumptions about use of managed wildfire are that (1) managed fires burning under moderate conditions will mitigate the spread and/or severity of subsequent wildfires, by reducing fuel loading and continuity at landscape scales; (2) exposure to moderate-condition fires increases system resilience (regenerative response) to subsequent fire; and (3) the risk is acceptably low that a managed fire will morph into an extreme region-scale event that unduly threatens forest values and human communities, if weather were to shift critically. While still relevant, each of these criteria has less footing in stand-replacing regimes.

First, unlike forests with frequent-fire regimes, exposure to moderate-condition fires in forests that are rarely fuel-limited will have little effect on the spread or severity of extreme events that drive most stand-replacing regimes. Feedbacks from prior fires can limit the occurrence (Parks et al. 2016), spread (Parks et al. 2015), and severity (Parks et al. 2014, Harvey et al. 2016*a*) of subsequent fires across many systems, but in forests with prompt and abundant post-fire tree regeneration, fuel recovery is rapid (Nelson et al. 2016) and negative feedbacks is short-lived compared to typical fire-return intervals (Harvey et al. 2016*a*). In addition, typical fire size distributions (Moritz et al. 2011) suggest that, in systems with century to multi-century fire-return intervals, ~80–90% of burn area comes in very large events (e.g., 10^5–10^6 ha for many forest types) occurring under extreme weather conditions. These extreme fires essentially cannot be managed and are largely indifferent to landscape fuel structure resulting from prior fires or fire management (Turner et al. 2003). It is only the remaining 10–20% that can realistically be managed, with a recognition that managed wildfire would be used primarily to create landscape diversity and habitat rather than to mitigate effects of subsequent large fires that will catalyze the most system change.

Second, exposure to the smaller sized fires (regardless of fire intensity and severity) that can be managed will likely do little to increase system resilience to large and severe fires under climate change, because, as stated above, these systems are already well adapted to large and severe fires. Although exposure to smaller fires would provide learning opportunities and theoretically offer more frequent opportunities for climate-suited vegetation to establish, it is debatable whether those small events accounting for <20% of burn area would ultimately influence vegetation responses in the very large patches that will drive most changes. Thus, it may be that these systems are best positioned for a more fiery future, with or without managed wildfire use.

Third, the risk profile for managed fire use differs in stand-replacing regimes. Most large fires in these systems do not immediately start out as extreme events upon ignition; rather, fires are often initially benign, burning under moderate conditions for perhaps many weeks, but then become large, high-severity fires when extreme weather events occur (e.g., synoptic dry east winds in the Cascade Range). Well-known examples include the 1988 Yellowstone Fires (~400,000 ha; Turner et al. 2003), the 1933 Tillamook Burn in the Oregon Coast Range (~100,000 ha; Kemp 1967), and the 1902 Yacolt Complex in the western Washington Cascades (>400,000 ha; Holbrook 1960). Managed wildfires, by definition, provide the first precondition (benign fire under moderate weather). Should a shift to extreme weather occur, the fuel abundance that characterizes these systems, coupled with an already broadly dispersed fire front, would make a switch to full suppression difficult. Thus, weather shifts could carry greater, less manageable consequences than in other systems.

All of this is not to suggest that managed wildfire has no role in stand-replacing regimes—it does have a role, as we discuss in subsequent sections—but rather to set realistic expectations for what it can achieve in these systems, and with what risks.

Principle 2

Like all forests, susceptibility to reorganization of these systems is greatest following stand-replacing wildfire events, especially if irreversible thresholds have been crossed in underlying environmental conditions.

Mature individuals of long-lived tree species can tolerate unfavorable climate conditions for several centuries (Brubaker 1986, Noss 2001). In the absence of stand-replacing fire, climate-induced shifts in composition and distribution of existing forests will likely be muted or substantially lagged (Franklin et al. 1992). This concept of landscape inertia assumes extensive shifts in vegetation will likely occur following broad-scale, stand-replacing disturbances that bring into play the sensitive seedling/regeneration stage (Franklin et al. 1992, Donato et al. 2016, Crausbay et al. 2017).

Although temperate forests with stand-replacing fire regimes are resilient to major disturbances, even in the warmer climate of recent decades (Larson and Franklin 2005, Shatford et al. 2007, Donato et al. 2009, Harvey et al. 2016*b*, Turner et al. 2016), there is conceivably a limit if underlying conditions change sufficiently. If irreversible thresholds are crossed in temperature and/or moisture regimes such that tree regeneration is significantly altered, qualitative changes in forest composition and structure could occur (Millar and Stephenson 2015, Trumbore et al. 2015). This phenomenon is sometimes termed "resilience debt," because the diminished capacity for forest recovery is evident only after the disturbance event (Johnstone et al. 2016). Because these vegetation changes often occur abruptly (Pace et al. 2015, Lindenmayer et al. 2016, Crausbay et al. 2017), and management in these forest types has relatively little influence on the events that catalyze them (see *Principle 1*), the likelihood of system change depends primarily on the stochastic occurrence of major disturbances, and whether irreversible thresholds in climatic conditions have been crossed (external forcing).

Strategies

Three core strategies emerge from these principles (summarized in Table 1).

Strategy 1

As part of an overall wildfire management strategy in stand-replacing regimes, carefully reasoned fire exclusion (e.g., suppression) is a useful tool to maintain certain ecosystem conditions and services, with fewer long-term negative consequences than in other forests.

Given that large patches of severe fire will occur in these systems regardless of stand management (Principle 1), fire management generally focuses on influencing whether fires initially spread. Fire exclusion is currently practiced across many of these forests as a default risk reduction strategy for economic and non-economic reasons. Although suggesting fire suppression as a viable ecological strategy may seem anathema given the major emphasis in fire ecology literature from drier forests, it is important to compare its context in long-interval, stand-replacing fire regimes with that in frequent low-severity regimes (Table 2).

Fire suppression is linked to numerous dysfunctional conditions in dry, frequent-fire forests (e.g., fuel accumulation, composition changes, reduced resilience; see Agee and Skinner 2005, Hessburg et al. 2015), whereas excluding wildfire for several decades in forests where fire intervals are much longer produces relatively minor increases in fire-free periods. Periods of fire exclusion are more consistent with historical system dynamics in these forests than are other common fire management strategies (e.g., stand-scale fuel treatments; Fig. 2). Implementing this strategy does not typically move these

Table 1. Stand- and landscape-level examples of adaptation strategies for the different principles and strategies described in this paper.

Scale	Principle 1: Most forests with stand-replacing disturbance regimes are inherently resilient	Principle 2: Forest reorganization is greatest following stand-replacing disturbance	
Strategies stemming from each principle	Strategy 1: Exclude wildfire where appropriate	Strategy 2: Minimize stressors to create a safe operating space	Strategy 3: Use post-disturbance response to promote climate-adapted landscapes
Stand-scale examples	• Exclude wildfire in areas of high human risk or value such as timberlands, key habitats, around infrastructure, or where fire absence is more consistent with restoration goals	• Diversify homogeneous second-growth forest by decreasing density and promoting species and structural diversity • Control non-native invasive species	• Use disturbance events as natural experiments (e.g., experiment with novel mixes of species and/or genotypes), including natural recovery/adaptation pathways
Large-landscape examples	• Consider managed wildfire in areas of low human value or where escape risks are tolerable • Exclude wildfire in areas of high human risk or value such as timberlands, key habitat, around infrastructure, or where it is consistent with restoration goals	• Promote and maintain genetic, species, and structural diversity • Promote landscape connectivity • Protect unique habitats	• Develop and implement post-disturbance vegetation management plans • Use post-disturbance management to promote and maintain genetic, species, and structural diversity

Table 2. Likelihood of key consequences of fire suppression.

Consequence	Dry forests	Cold/moist forests
Inconsistency with natural disturbance dynamics	High	Low
Structural departure across most successional stages	High	Low
Reduced abundance of early-successional pre-forest conditions†	Moderate	Moderate
Artificial abundance of dense, late-successional conditions	High	Low
Loss of meadows caused by tree cover expansion	High	Moderate
Increased stand-replacing patch size, reduced patch diversity	High	Moderate
Patch size increases beyond major species' dispersal ability	High	Low
Loss of short-lived serotinous tree species‡	Low	High
Reduced role of low- and mixed-severity fire patches in creating structural and compositional diversity across landscapes	High	Moderate
Reduced opportunity for prior burn footprints to curtail spread or severity of subsequent wildfires	High	Low
Increased native insect outbreaks	High	Low
Reduced opportunity to "burn away" pathogens	High	Moderate
Reduced opportunities for establishment of climate-suitable plant species	High	Moderate
Fewer on-the-ground learning and adaptive management opportunities	High	High
Increased risk of large wildfires affecting human communities	High	Low
Worsened fire-related air quality issues	High	Low
Reduced long-term forest resilience (defined in Principle 1)	High	Low

† Dry forests are rated moderate because large patches of early-successional habitat are not abundant under natural fire regimes. Cold/moist forests are rated moderate because large severe fire patches are infrequent under natural fire regimes, and the largest events will likely occur irrespective of fire management strategy (see *Strategy 1*).

‡ Such species are less common in dry, frequent-fire forests.

systems outside the historical range of conditions, nor does it worsen future fire behavior, because fires that do most of the work in these landscapes burn under extreme conditions in which prior burn footprints and age/structure classes matter little to fire spread or severity (Turner et al. 2003).

This extreme version of a resistance approach (Walker et al. 2004) forestalls change and maintains the social–ecological values of infrequently disturbed ecosystems for as long as possible. It artificially increases system resistance in forests that have inherently low resistance to severe disturbance (not fuel limited) and buffers forests from human-caused ignition sources. This approach would be most suitable where the emphasis is on maintaining certain conditions, such as timber production, human structures, habitats for threatened species, unique features such as old-growth trees that are difficult to replace on a socially meaningful timescale, established carbon mitigation projects, and municipal watersheds (Table 3). In contrast, where allowing ecological disturbance processes to operate is a primary objective, wildfire could be managed where practical (Parsons et al. 1986, Romme and Despain 1989), to allow initiation of younger successional stages and establishment of climate-adapted vegetation communities (Table 3). Managed wildfire can also be appropriate where human risk is low, especially given costs of fire suppression and projections of increased area burned in a warmer climate.

Although more justifiable than in other systems, fire suppression in stand-replacing regimes is not completely without ecological consequences (Table 2). By limiting fires to those occurring under such extreme conditions that they cannot be suppressed, this strategy would diminish the creation of low- to moderate-severity fire patches, reduce landscape patch diversity, and lessen opportunities for climate-suited vegetation establishment; however, the effects of these changes are almost certainly less critical than in frequent-fire forests (Table 2). Consistent with a managed wildfire approach, the decision to suppress fires would consider ecological and social factors such as fire regime, community protection, ignition source (human or lightning), values at risk, and management objectives for a landscape (e.g., emphasis on conditions vs. disturbance processes). The decision space will often vary depending on ownership patterns (Table 3). For example, in a landscape

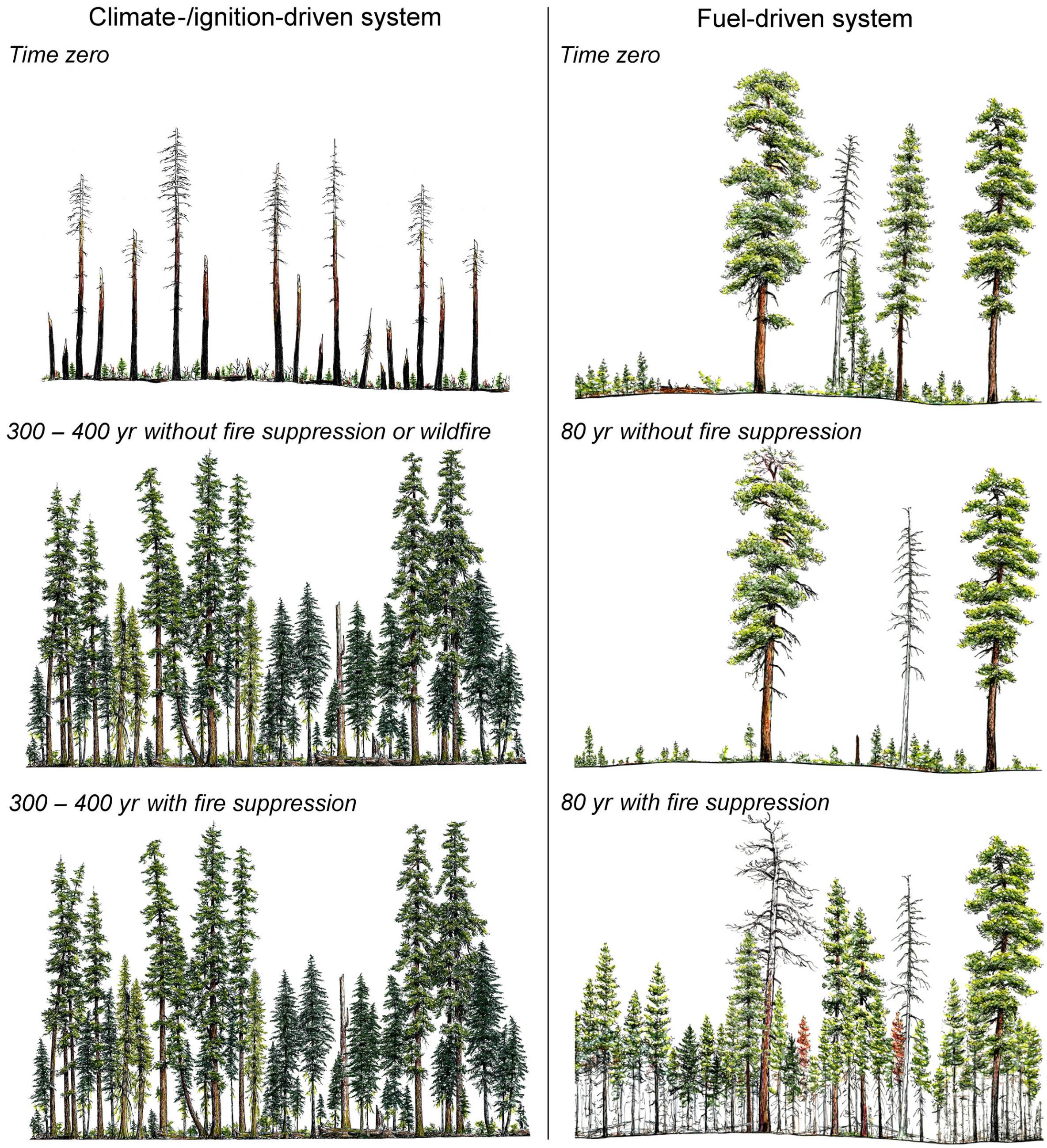

Fig. 2. A comparison of forest structure in climate- and fuel-driven systems following a natural disturbance (Time zero). In climate-driven systems, large patches of stand-replacing wildfire are more common and fire rotations are sufficiently long that forest structures and fuels are similar irrespective of the mechanism causing wildfire absence (e.g., temporal stochasticity or management). In contrast, fuel-driven systems tend to have more low- and mixed-severity wildfire with shorter fire rotations, resulting in fuel accumulation when wildfire is excluded. Illustrations depict a moist *Pseudotsuga-Tsuga-Thuja* forest of the western Cascade Range (left panels) and a dry *Pinus-Pseudotsuga* forest of the eastern Cascade range (right panels). Adapted with permission (Van Pelt 2007, 2008, Franklin et al. 2008). Figures not drawn to scale.

Table 3. Some management options for different forest ownerships/managers and land designations in forests with high-severity fire regimes under changing climate.

Ownership/management/land designation	Forest management options
Managed primarily for ecological process	• Allow lightning-ignited fires to burn where appropriate, recognizing potential risk of inability to "control" the fire • Suppress wildfire around high-value resources • Reduce existing ecosystem stressors to increase resilience to disturbance and climate change • Coordinate with adjacent landowners/managers on a fire management plan • Use natural regeneration after fires • Monitor stand trajectories after fires
State and federal lands managed for both economic and ecological values	• Suppress most wildfires • Reduce existing ecosystem stressors to increase resilience to climate change and disturbance • Increase species and structure diversity at the stand and/or landscape scale • Conduct fuel treatments or create fuel breaks in or around high-value resources • Coordinate with adjacent landowners/managers on a fire management plan • Develop a post-fire response plan • Use disturbance events as opportunities for experimentation and to influence stand successional trajectories
Non-industrial private landowners	• Suppress wildfires • Reduce existing ecosystem stressors • Increase species and structural diversity at the stand scale • Plant species and genotypes that will perform well under future conditions (i.e., hotter and drier conditions)
Industrial private landowners	• Suppress wildfires • Experiment with different mixes of species and genotypes to increase resilience to climate change • Create fuel breaks around high-value stands

composed of developed lands, private timberlands, or areas of high conservation value, near-complete fire suppression can be ecologically, economically, and socially defensible, although some values may decline (Table 2). Conversely, managed wildfires may be ecologically and economically justified where natural processes are prioritized, risk to human values is low, or the more consequential fire escape risks (Principle 1) are acceptable.

An example of tradeoffs associated with Strategy 1 relates to conservation-relevant successional stages. A common restoration objective in these forests centers on redevelopment of late-successional (mature and old-growth) forests, following decades of extensive timber harvest (Davis et al. 2015). Allowing the landscape to age via fire exclusion can therefore be considered restoration at broad scales. Fig. 3a projects trends in older forest conditions across 2.5 million ha of high-severity and historically infrequently disturbed forest in western Washington state under different climate and management assumptions. In addition to public lands, this landscape encompasses other lands of high conservation value, high- and low-intensity development, and both private industrial and non-industrial landowners. Without climate change (short-dashed line) and left to grow, older forest conditions would become more abundant on the landscape over time. However, with climate change, increases in older forests are muted because of higher rates of stand-replacing wildfire (solid line). Implementing a management approach of reducing burned area by 50% (Strategy 1; see long-dashed line) returns the forest trajectory to near that of the no-climate-change scenario. Thus, in the right context, a carefully considered fire suppression strategy can increase the likelihood of achieving certain restoration objectives, as well as economic and conservation goals.

On the other hand, this strategy would temporarily further constrain the abundance of naturally generated, early-successional (pre-forest) conditions, an important habitat thought to be well below historical levels in many regions (Swanson et al. 2011, Reilly and Spies 2015, Fig. 3b). This constraint is not to be discounted, because structurally complex early-successional

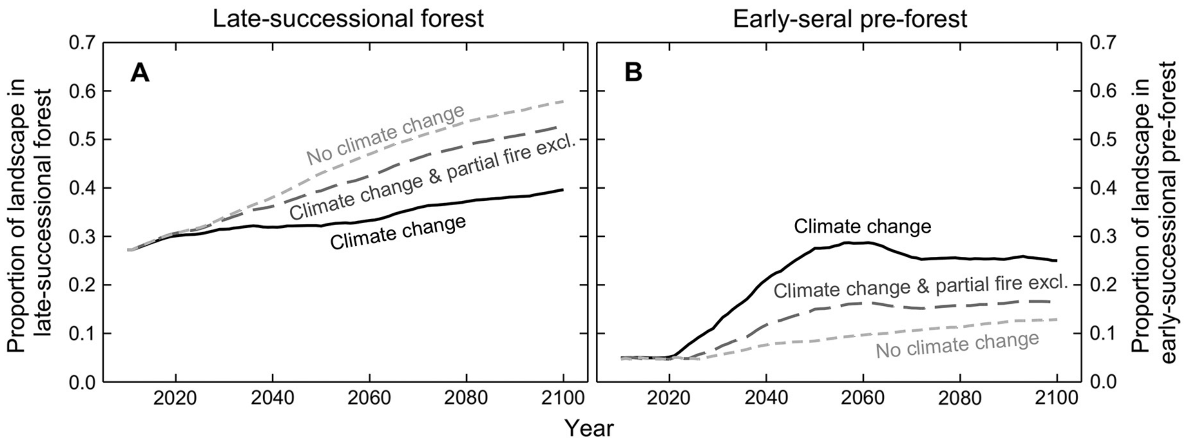

Fig. 3. Projected trends in late-successional and early-successional habitats in a high-severity and infrequently disturbed forest in western Washington state under different climate and management assumptions. Projections were made with climate-informed state-and-transition simulation models (see Halofsky et al. 2013 for detailed method description) using the software ST-Sim (Apex RMS, 2013). Climate change trend lines represent average annual area across three global climate models (HadGEM2_ES RCP 8.5, CSIRO_MK360 RCP 8.5, and NORESM1 RCP 8.5) and 300 Monte Carlo simulations. Partial fire exclusion assumes a 50% reduction in area burned caused by fire suppression. Late-successional forest is defined as trees with a quadratic mean diameter ≥58 cm, ≥40% canopy cover, and ≥2 canopy layers. Complex early-successional pre-forest encompasses the period between a disturbance and tree canopy closure.

pre-forest habitat is already glaringly absent in some regions. In most stand-replacing regimes, however, early-successional conditions are typically generated by large events occurring at century to multi-century intervals, under extreme conditions in which wildfire control would be ineffective (Romme 1982, Turner et al. 2003). The large patches of early-successional forest will eventually be delivered by large wildfires over the long term, even with fire suppression (Fig. 3b). The key for those large patches, as well as smaller patches created more frequently, is whether passive or active management pathways are taken post-disturbance (see *Strategy 3*), and whether essential structural features of high-quality, early-successional habitat (Swanson et al. 2011) are retained.

Strategy 2

Until fires inevitably occur, minimizing other stressors by applying basic principles of conservation biology can promote robust system response to severe disturbance, avoiding many negative impacts on ecosystem function and services.

The success or failure of fire exclusion depends not on whether major fires ever occur (they will sooner or later), but on whether system responses to disturbance are similar to those in the past. If thresholds in climatic conditions and fire frequency have not been crossed, and the system is not otherwise stressed, the forest can be expected to cycle through characteristic successional stages toward its pre-disturbance condition.

Irreversible thresholds are difficult to model or anticipate, but if thresholds are crossed by the time fire exclusion fails, then the system (species, processes, functionality) may change following a major event (Crausbay et al. 2017). The magnitude of these changes will depend, in part, on the length of fire-free periods; the longer the interval, the more resilience debt accumulates, and the larger step the system must take to re-equilibrate to the new climate (assuming climate departure will generally increase with time). Thus, a tangible drawback of Strategy 1 (emphasizing fire suppression) would be fewer small-scale opportunities for post-fire plant establishment under the current climate (Table 2), resulting in larger incremental changes when fires occur, even if system endpoints are ultimately similar. The social, economic, and ecological implications of these larger increments are poorly defined, but because these changes will likely occur whether the approach is managed wildfire or suppression, fostering adaptive resilience in these systems will entail minimizing other stressors that could push a system toward thresholds.

Forest management in these systems under climate change may appear like ecosystem-based management under static conditions (Noss 2001), but with increased emphasis on minimizing stressors that could strain resilience. Minimizing stressors helps maintain a safe operating space for ecosystems by staying within acceptable levels of tolerance of existing ecosystem functions, facilitating resilience to perturbation (Scheffer et al. 2015). Examples include promoting landscape connectivity, controlling non-native species, identifying and protecting climate refugia and other unique habitats, and maintaining diverse genetic resources among and within species (Halofsky et al. 2011).

Consistent with conservation biology principles, promoting species and structural diversity across multiple scales is an important pre-disturbance strategy. Many of the forests addressed here are dominated by conifers, often by only a handful of species, with less redundancy in autecological strategies compared to more diverse forests. Maintaining a full complement of species may be key to resilience, keeping a variety of regeneration strategies and climatic responses as part of the system. For example, retaining an abundance of broadleaf species in stands and across the landscape may provide several important ecosystem benefits. Deciduous hardwood species can reduce the flammability of forests and landscapes dominated by conifers (DeRose and Leffler 2014), add to resilience by sprouting in response to disturbance (Johnstone et al. 2004), use less water than evergreen conifers resulting in more streamflow (Swank et al. 1988), and enrich forest habitat for many biota (Swanson et al. 2011). Depending on management objectives and rotation length, diversifying the landscape can also sustain the economic viability of timber production in an uncertain future.

Managing for structural diversity starts with promoting full representation of the successional spectrum on a landscape for a given forest type (Halofsky et al. 2011, Franklin and Johnson 2012, Raymond et al. 2014, Lindenmayer et al. 2016), rather than focusing exclusively on old-growth or rotation-age production forests. Managing for a range of successional stages provides structural and compositional diversity at broad scales, which could help increase resilience to other climate-related stressors, such as insect and disease outbreaks (Raymond et al. 2014). Managed wildfire is one tool to promote this structural diversity across large landscapes, where risk to human values is low or fire escape risks can be tolerated. Indeed, one drawback of lessening the use of managed wildfire and emphasizing suppression is that it would simplify fire-induced landscape diversity to some extent, by lessening opportunities for low- to moderate-severity fire to operate (Table 2). Where managed wildfire use is not practical or desirable, such as some landscapes with both ecological and economic objectives, diversifying stands and the landscape will necessitate forest management tools other than wildfire, recognizing that wildfire provides ecological outcomes that cannot be entirely replicated through silviculture.

At the stand scale, decreasing forest densities and increasing species and structural diversity in second-growth forests that have developed from previous timber management could reduce drought stress (Sohn et al. 2016) and provide a broader range of habitats (and adaptive capacity) for some animal species (Hunter 1998). Retaining cohorts of large/old legacy trees in otherwise younger forests, as in variable retention harvest systems (Fedrowitz et al. 2014), can provide another resilience mechanism. These trees are the most fire resistant, providing seed sources for regeneration that span centuries. The role of density-reduction treatments will vary by forest type. For example, in highly productive coastal forests, thinning may be employed mostly to accelerate the development of late-successional conditions, whereas in less productive continental-interior forests, thinning may be employed largely to mitigate drought stress or other factors.

As with Strategy 1, the degree of compositional and structural diversity emphasized at different spatial scales will depend on management objectives and ownership patterns. It is unlikely that short-rotation forestry focused on revenue will diversify much of the land base beyond those conditions present under such a management regime. However, these lands will also have the most frequent opportunities to alter species composition and genetics in response to a changing climate. Lands primarily set aside for conservation (e.g., national parks) are also less likely to provide the full complement of structural stages until disturbed, because these areas have minimal active management. Forest landscapes that fall between these ends of the management spectrum have the greatest opportunities for promoting structural and compositional diversity. This diversity is more likely in landscapes where multiple landowners with different management objectives are managing adjacent lands, thereby creating different sets of forest structures.

Strategy 3

In these forest types, post-disturbance response strategies are among the greatest opportunities to create climate-adapted landscapes, as well as opportunities for learning.

Unlike other forest types in which resilience strategies can be readily implemented before and after disturbance events to facilitate gradual change (Millar and Stephenson 2015), post-disturbance situations provide a primary opportunity to both develop and implement adaptation strategies in forests with stand-replacing disturbance regimes. Similar to other forest types, the stand-initiation/regeneration stage provides opportunities for (1) recovery processes to adapt the system to changing climate conditions via establishment and survival of different species mixes, (2) management actions to promote desired ecological and/or economic outcomes under a changing climate, and (3) ecological learning that can be integrated into future post-fire responses.

For natural recovery pathways, as in other forest types, a simple first strategy is to take advantage of the spatial heterogeneity of fires (Turner et al. 2013). Even fires with the largest stand-replacement patches are part of a mixed-severity mosaic with unburned islands (Kolden et al. 2012), convoluted edges, and refugia (e.g., riparian areas) that act as propagule sources and thus nuclei for regeneration (Donato et al. 2009, Harvey et al. 2016*b*). Actions that simplify this mosaic, such as intentional burning of patches of surviving trees, directly reduce a key resilience mechanism for burned landscapes. Also, resilience encompasses more than just rapid re-establishment of forest cover. Even where tree regeneration is slow, retaining most ecosystem components will facilitate ecosystem function and adaptive capacity over long timescales.

For actively managed forests, having post-disturbance management plans in place prior to disturbances will help ensure timely and effective response (Peterson et al. 2009, Halofsky et al. 2011). Managers will need to determine how to respond following a stand-replacing disturbance in an era of changing climate, differentiating actions with respect to disturbance agent, environmental conditions before and after an event (e.g., drought), and whether a disturbance is a single or coupled event (e.g., re-burn, or fire following a wind event). Post-disturbance management plans will vary by how much intervention might be necessary to maintain ecosystem resilience, and whether the degree of intervention will be consistent with other social and management goals (Hobbs et al. 2010). For example, by depending on climate and available seed source, landscapes managed for process and left alone to recover will adapt to novel conditions (Principle 2) in ways that may or may not be consistent with socio-ecological objectives. If human intervention is allowed, the decision space becomes large, and potential post-disturbance strategies can be culled from the existing literature (O'Neill et al. 2017) or online libraries (e.g., http://adaptationpartners.org/library.php).

Because of the uncertainties associated with climate change and effectiveness of management actions under potentially novel conditions, we propose capitalizing on stand-replacing disturbances as natural experiments, where compatible with other management objectives. For example, a portion of the disturbed area could be allowed to regenerate naturally, a portion could be planted using traditional strategies and species composition, and a portion could be planted with novel mixes of species and/or genotypes at varied densities, based on anticipated shifts in site suitability. This approach provides learning opportunities within an adaptive management framework, and creates diversity across the landscape (e.g., increasing the currently rare pre-forest condition; Swanson et al. 2011), thereby promoting broad-scale heterogeneity in ecosystem properties and resilience to changing climatic conditions and disturbance regimes.

Managers need not wait for broad-scale stand-replacing natural disturbance events prior to implementing Strategy 3. If regeneration harvests are also viewed as stand-replacing disturbances, many agencies and private landowners will have more frequent opportunities to incorporate post-disturbance adaptation strategies in the near term. For example, on lands that need to meet both economic and ecological objectives, actions to increase genetic and species diversity of seedlings could be implemented sooner than on lands managed for mostly ecological values (especially old forests). Diversifying genotypes and species could occur within stands (alpha diversity) if little economic return is required, or between stands (beta diversity) to reduce harvest costs. As mentioned earlier, a purposeful effort to include deciduous hardwoods as well as conifers could further support ecosystem services and diversify management options.

For recently regenerated stands with longer rotations, some stands, or portions thereof, could be allowed to regenerate naturally, and other stands (or portions of stands) can be planted using traditional silvicultural prescriptions. Recently harvested stands on southerly aspects could perhaps be planted at a lower tree density with more frequent thinning through time. However, managers may also want to account for increasing mortality as a result of increased soil moisture deficit and other stressors.

The timing of a shift in management strategy will depend on management objectives and forest sensitivity to a changing climate. Higher elevation and continental-interior forests with stand-replacing disturbance regimes are likely more sensitive to changes in temperature and precipitation than temperate coastal forests, where the ocean may act as a buffer to change (Turner et al. 2013). Although there may be short-term economic costs to implementing post-disturbance adaptation strategies in managed forests, this perceived liability may become an asset as the future becomes less certain (Puettmann 2014). The timing of this shift will depend on geographic area, risk tolerance of landowners, and management objectives.

Conclusion

In some ways, forests with stand-replacing disturbance regimes may be easy to manage from a climate adaptation perspective, because major changes in policy are not required, but some changes in policy and forest management practices will likely be necessary to sustain ecosystem services. For example, fire suppression is already a default policy across many forest ownerships, albeit not necessarily with the more specifically targeted approach we describe. Planting following disturbances is a common activity on forest lands, but forest managers have the option of altering species, genotypes, and densities to maximize survival and growth under a changing climate. Some of these shifts in practices will require managers to address policy barriers (e.g., U.S. National Environmental Policy Act), organizational capacity (e.g., employee education and financial costs), and public acceptance (e.g., of managed wildfire) before implementation can occur.

There are few options for directing or mitigating the size and severity of disturbances that ultimately shape these systems without negatively affecting values currently associated with them (e.g., carbon sequestration, wildlife habitat, timber production). It is the nature of the beast. Given the disturbance regime, broad-scale transitions to novel forests may be abrupt over space and time. As such, management for climate adaptation is limited primarily to delaying major disturbance events where appropriate, creating a safe operating space for the system to respond to disturbances, and capitalizing on the post-disturbance period as an opportunity for adaptation when disturbances occur. Careful planning now will allow organizations to articulate a range of future possible responses in collaboration with stakeholders and the public, setting reasonable expectations for what management can and cannot do to foster resilience.

Acknowledgments

We thank A. Estep and A. Hayes for project support and D. Siemann and K. Moon for initiating the discussion topic. This manuscript benefitted greatly from reviews and comments from D. Churchill, J. DeBell, J. Freund, K. Hagmann, R. Kramer, S. McLeod, M. Turner, R. Van Pelt, S. Winter, and anonymous reviews. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author(s) and do not necessarily reflect the views of the funding or affiliate organization(s). Joshua S. Halofsky and Daniel C. Donato are co-first authors; order determined by Magic 8-ball.

Literature Cited

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C., USA.

Agee, J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83–96.

Apex RMS. 2013. ST-Sim state-and-transition simulation model software. http://www.apexrms.com/stsm

Barbero, R., J. T. Abatzoglou, N. K. Larkin, C. A. Kolden, and B. Stocks. 2015. Climate change presents increased potential for very large fires in the contiguous United States. International Journal of Wildland Fire 24:892–899.

Bessie, W. C., and E. A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology 76:747–762.

Brown, R. T., J. K. Agee, and J. F. Franklin. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18:903–912.

Brubaker, L. B. 1986. Responses of tree populations to climatic change. Plant Ecology 67:119–130.

Collins, B. M., and S. L. Stephens. 2007. Managing natural wildfires in Sierra Nevada wilderness areas. Frontiers in Ecology and the Environment 5:523–527.

Collins, B. M., J. T. Stevens, J. D. Miller, S. L. Stephens, P. M. Brown, and M. P. North. 2017. Alternative characterization of forest fire regimes: incorporating spatial patterns. Landscape Ecology 32:1543–1552.

Crausbay, S. D., P. E. Higuera, D. G. Sprugel, and L. B. Brubaker. 2017. Fire catalyzed rapid ecological change in lowland coniferous forests of the Pacific Northwest over the past 14,000 yr. Ecology 98:2356–2369.

Davis, R. J., J. L. Ohmann, R. E. Kennedy, W. B. Cohen, M. J. Gregory, Z. Yang, H. M. Roberts, N. Gray, and T. A. Spies. 2015. Northwest Forest Plan–the first 20 years (1994–2013): status and trends of late-successional and old-growth forests. PNW-GTR-911. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

DeRose, R. J., and A. J. Leffler. 2014. Simulation of quaking aspen potential fire behavior in Northern Utah, USA. Forests 5:3241–3256.

Dillon, G. K., Z. A. Holden, P. Morgan, M. A. Crimmins, E. K. Heyerdahl, and C. H. Luce. 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984–2006. Ecosphere 2:130.

Donato, D. C., J. B. Fontaine, J. L. Campbell, W. D. Robinson, J. B. Kauffman, and B. E. Law. 2009. Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath-Siskiyou mountains. Canadian Journal of Forest Research 39:823–838.

Donato, D. C., B. J. Harvey, and M. G. Turner. 2016. Regeneration of montane forests 24 years after the 1988 Yellowstone fires: A fire-catalyzed shift in lower treelines? Ecosphere 7:e01410.

Fedrowitz, K., et al. 2014. Can retention forestry help conserve biodiversity? A meta-analysis. Journal of Applied Ecology 51:1669–1679.

Flannigan, M. D., M. A. Krawchuk, W. J. de Groot, B. M. Wotton, and L. M. Gowman. 2009. Implications of changing climate for global wildland fire. International Journal of Wildland Fire 18:483–507.

Franklin, J. F., M. A. Hemstrom, R. Van Pelt, J. B. Buchanan, and S. Hull. 2008. The case for active management of dry forest types in eastern Washington: perpetuating and creating old forest structures and functions. Washington Department of Natural Resources, Olympia, Washington, USA.

Franklin, J. F., and K. N. Johnson. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry 110:429–439.

Franklin, J. F., et al. 1992. Effects of global climate change on forests in Northwestern North America. Pages 244–257 *in* R. L. Peters and T. E. Lovejoy, editors. Global warming and biological diversity. Yale University Press, New Haven, Connecticut, USA.

Freund, J. A., J. F. Franklin, A. J. Larson, and J. A. Lutz. 2014. Multi-decadal establishment for single-cohort Douglas-fir forests. Canadian Journal of Forest Research 44:1068–1078.

Halofsky, J. E., M. A. Hemstrom, D. R. Conklin, J. S. Halofsky, B. K. Kerns, and D. Bachelet. 2013. Assessing potential climate change effects on vegetation using a linked model approach. Ecological Modelling 266:131–143.

Halofsky, J. E., D. L. Peterson, and K. A. O'Halloran. 2011. Adapting to climate change at Olympic National Forest and Olympic National Park. PNW-GTR-844. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Harvey, B. J., D. C. Donato, and M. G. Turner. 2016*a*. Burn me twice, shame on who? Interactions between successive forest fires across a temperate mountain region. Ecology 97:2272–2282.

Harvey, B. J., D. C. Donato, and M. G. Turner. 2016*b*. High and dry: Post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. Global Ecology and Biogeography 25:655–669.

Harvey, B. J., and B. A. Holzman. 2014. Divergent successional pathways of stand development following fire in a California closed-cone pine forest. Journal of Vegetation Science 25:88–99.

Hessburg, P. F., et al. 2015. Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecology 30:1805–1835.

Hobbs, R. J., et al. 2010. Guiding concepts for park and wilderness stewardship in an era of global environmental change. Frontiers in Ecology and the Environment 8:483–490.

Holbrook, S. H. 1960. Burning an empire: the story of American forest fires. The Macmillan Company, New York, New York, USA.

Hunter, M. L. 1998. Wildlife, forests and forestry: principles of managing forests for biological diversity. Prentice Hall, Englewood Cliffs, New Jersey, USA.

Johnstone, J. F., F. S. Chapin III, J. Foote, S. Kemmett, K. Price, and L. Viereck. 2004. Decadal observations of tree regeneration following fire in boreal forests. Canadian Journal of Forest Research 34:267–273.

Johnstone, J. F., et al. 2016. Changing disturbance regimes, ecological memory, and forest resilience. Frontiers in Ecology and the Environment 14: 369–378.

Keane, R. E., J. K. Agee, P. Fulé, J. E. Keeley, C. Key, S. G. Kitchen, R. Miller, and L. A. Schulte. 2008. Ecological effects of large fires on US landscapes: Benefit or catastrophe? International Journal of Wildland Fire 17:696–712.

Keeley, J. E., G. Ne'eman, and C. J. Fotheringham. 1999. Immaturity risk in a fire-dependent pine. Journal of Mediterranean Ecology 1:41–48.

Kemp, J. L. 1967. Epitaph for the giants: the story of the Tillamook burn. Touchstone Press, Portland, Oregon, USA.

Kolden, C. A., J. A. Lutz, C. H. Key, J. T. Kane, and J. W. van Wagtendonk. 2012. Mapped versus actual burned area within wildfire perimeters: characterizing the unburned. Forest Ecology and Management 286:38–47.

Larson, A. J., and J. F. Franklin. 2005. Patterns of conifer tree regeneration following an autumn wildfire event in the western Oregon Cascade Range, USA. Forest Ecology and Management 218:25–36.

Lindenmayer, D. B., C. Messier, and C. Sato. 2016. Avoiding ecosystem collapse in managed forest ecosystems. Frontiers in Ecology and the Environment 14:561–568.

Loehman, R. A., et al. 2018. Effects of climate change on ecological disturbance in the Northern Rockies. Pages 115–141 *in* J. E. Halofsky and D. L. Peterson, editors. Climate change and rocky mountain ecosystems. Advances in global change research. Volume 63. Springer International Publishing, Cham, Switzerland.

McKenzie, D., and J. S. Littell. 2017. Climate change and the eco-hydrology of fire: Will area burned increase in a warming western USA? Ecological Applications 27:26–36.

McKenzie, D., D. L. Peterson, and J. J. Littell. 2008. Global warming and stress complexes in forests of western North America. Developments in Environmental Science 8:319–337.

Millar, C. I., and N. L. Stephenson. 2015. Temperate forest health in an era of emerging megadisturbance. Science 349:823–826.

Millar, C. I., N. L. Stephenson, and S. L. Stephens. 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications 17:2145–2151.

Mitchell, S. R., M. E. Harmon, and K. E. O'connell. 2009. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. Ecological Applications 19: 643–655.

Moritz, M. A., P. F. Hessburg, and N. A. Povak. 2011. Native fire regimes and landscape resilience. Pages 51–86 *in* D. McKenzie, C. Miller, and D. A. Falk, editors. The landscape ecology of fire. Springer, New York, New York, USA.

Moritz, M. A., et al. 2014. Learning to coexist with wildfire. Nature 515:58–66.

Morrison, P. H., and F. J. Swanson. 1990. Fire history and pattern in a Cascade Range Landscape. PNW-GTR-254. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Mote, P., A. K. Snover, S. Capalbo, S. D. Eigenbrode, P. Glick, J. Littell, R. Raymondi, and S. Reeder. 2014. Chapter 21: Northwest. Pages 487–513 *in* J. M. Melillo, T. C. Richmond, and G. W. Yohe, editors. Climate change impacts in the United States: the third national climate assessment. U.S. Global Change Research Program, Washington, D.C., USA.

Nelson, K. N., M. G. Turner, W. H. Romme, and D. B. Tinker. 2016. Landscape variation in tree regeneration and snag fall drive fuel loads in 24-year old post-fire lodgepole pine forests. Ecological Applications 26:2422–2436.

North, M., B. M. Collins, and S. Stephens. 2012. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. Journal of Forestry 110:392–401.

Noss, R. F. 2001. Beyond Kyoto: forest management in a time of rapid climate change. Conservation Biology 15:578–590.

Noss, R. F., J. F. Franklin, W. L. Baker, T. Schoennagel, and P. B. Moyle. 2006. Managing fire-prone forests in the western United States. Frontiers in Ecology and the Environment 4:481–487.

O'Neill, G., T. Wang, N. Ukrainetz, L. Charleson, L. McAuley, A. Yanchuk, and S. Zedel. 2017. A proposed climate-based seed transfer system for British

Columbia. Prov. BC, Victoria. BC Tech. Rep. 099. www. for. gov. bc. ca/hfd/pubs/Docs/Tr/Tr099. htm

Pace, M. L., S. R. Carpenter, and J. J. Cole. 2015. With and without warning: managing ecosystems in a changing world. Frontiers in Ecology and the Environment 13:460–467.

Parks, S. A., L. M. Holsinger, C. Miller, and C. R. Nelson. 2015. Wildland fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression. Ecological Applications 25:1478–1492.

Parks, S. A., C. Miller, L. M. Holsinger, L. S. Baggett, and B. J. Bird. 2016. Wildland fire limits subsequent fire occurrence. International Journal of Wildland Fire 25:182–190.

Parks, S. A., C. Miller, C. R. Nelson, and Z. A. Holden. 2014. Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. Ecosystems 17:29–42.

Parsons, D. J., D. M. Graber, J. K. Agee, and J. W. Van Wagtendonk. 1986. Natural fire management in national parks. Environmental Management 10: 21–24.

Peterson, D. L., J. K. Agee, G. H. Aplet, D. P. Dykstra, R. T. Graham, J. F. Lehmkuhl, D. S. Pilliod, D. F. Potts, R. F. Powers, and J. D. Stuart. 2009. Effects of timber harvest following wildfire in western North America. PNW-GTR-776. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Peterson, D. L., M. C. Johnson, J. K. Agee, T. B. Jain, D. McKenzie, and E. D. Reinhardt. 2005. Forest structure and fire hazard in dry forests of the western United States. PNW-GTR-628. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Peterson, D. L., J. M. Vose, and T. Patel-Weynand, editors. 2014. Climate change and United States forests. Advances in Global Change Research 57. Springer, Dordrecht, The Netherlands.

Puettmann, K. J. 2014. Restoring the adaptive capacity of forest ecosystems. Journal of Sustainable Forestry 33(sup1):S15–S27.

Raymond, C. L., D. L. Peterson, and R. M. Rochefort. 2014. Climate change vulnerability and adaptation in the North Cascades region, Washington. PNW-GTR-892. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Reilly, M. J., C. J. Dunn, G. W. Meigs, T. A. Spies, R. E. Kennedy, J. D. Bailey, and K. Briggs. 2017. Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). Ecosphere 8:e01695.

Reilly, M. T., and T. A. Spies. 2015. Regional variation in stand structure and development in forests of Oregon, Washington, and inland Northern California. Ecosphere 6:192.

Rogers, B. M., R. P. Neilson, R. Drapek, J. M. Lenihan, J. R. Wells, D. Bachelet, and B. E. Law. 2011. Impacts of climate change on fire regimes and carbon stocks of the US Pacific Northwest. Journal of Geophysical Research: Biogeosciences 116:G3.

Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecological Monographs 52:199–221.

Romme, W. H., and D. G. Despain. 1989. Historical perspective on the Yellowstone fires of 1988. BioScience 39:695–699.

Romps, D. M., J. T. Seeley, D. Vollaro, and J. Molinari. 2014. Projected increase in lightning strikes in the United States due to global warming. Science 346:851–854.

Scheffer, M., et al. 2015. Creating a safe operating space for iconic ecosystems. Science 347:1317–1319.

Schoennagel, T., M. G. Turner, and W. H. Romme. 2003. The influence of fire interval and serotiny on postfire lodgepole pine density in Yellowstone National Park. Ecology 84:2967–2978.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. AIBS Bulletin 54:661–676.

Schoennagel, T., et al. 2017. Adapt to more wildfire in western North American forests as climate changes. Proceedings of the National Academy of Sciences of the USA 114:4582–4590.

Seidl, R., T. A. Spies, D. L. Peterson, S. L. Stephens, and J. A. Hicke. 2016. Searching for resilience: addressing the impacts of changing disturbance regimes on forest ecosystem services. Journal of Applied Ecology 53:120–129.

Shatford, J. P. A., D. E. Hibbs, and K. J. Puettmann. 2007. Conifer regeneration after forest fire in the Klamath-Siskiyous: How much, how soon? Journal of Forestry 105:139–146.

Sohn, J. A., S. Saha, and J. Bauhus. 2016. Potential of forest thinning to mitigate drought stress: a meta-analysis. Forest Ecology and Management 380:261–273.

Stephens, S. L., J. K. Agee, P. Z. Fulé, M. P. North, W. H. Romme, T. W. Swetnam, and M. G. Turner. 2013. Managing forests and fire in changing climates. Science 342:41–42.

Stephens, S. L., J. D. McIver, R. E. Boerner, C. J. Fettig, J. B. Fontaine, B. R. Hartsough, P. L. Kennedy, and D. W. Schwilk. 2012. The effects of forest fuel-reduction treatments in the United States. BioScience 62:549–560.

Swank, W. T., L. W. Swift Jr., and J. E. Douglass. 1988. Streamflow changes associated with forest cutting, species conversions, and natural disturbances. Pages

297–312 *in* W. T. Swank and D. A. Crossley Jr., editors. Forest hydrology and ecology at Coweeta. Springer-Verlag, New York, New York, USA.

Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. B. Lindenmayer, and F. J. Swanson. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9:117–125.

Tepley, A. J., F. J. Swanson, and T. A. Spies. 2013. Fire-mediated pathways of stand development in Douglas-fir/western hemlock forests of the Pacific Northwest, USA. Ecology 94:1729–1743.

Timpane-Padgham, B. L., T. Beechie, and T. Klinger. 2017. A systematic review of ecological attributes that confer resilience to climate change in environmental restoration. PLoS ONE 12:e0173812.

Trumbore, S., P. Brando, and H. Hartmann. 2015. Forest health and global change. Science 349:814–818.

Turner, M. G. 2010. Disturbance and landscape dynamics in a changing world. Ecology 91:2833–2849.

Turner, M. G., D. C. Donato, and W. H. Romme. 2013. Consequences of spatial heterogeneity for ecosystem services in changing forest landscapes: priorities for future research. Landscape Ecology 28:1081–1097.

Turner, M. G., W. H. Romme, R. H. Gardner, and W. W. Hargrove. 1997. Effects of fire size and pattern on early succession in Yellowstone National Park. Ecological Monographs 67:411–433.

Turner, M. G., W. H. Romme, and D. B. Tinker. 2003. Surprises and lessons from the 1988 Yellowstone fires. Frontiers in Ecology and the Environment 1:351–358.

Turner, M. G., T. G. Whitby, D. B. Tinker, and W. H. Romme. 2016. Twenty-four years after the Yellowstone Fires: Are postfire lodgepole pine stands converging in structure and function? Ecology 97:1260–1273.

Van Pelt, R. 2007. Identifying mature and old forests in western Washington. Washington State Department of Natural Resources, Olympia, Washington, USA.

Van Pelt, R. 2008. Identifying old trees and forests in eastern Washington. Washington State Department of Natural Resources, Olympia, Washington, USA.

Walker, B., C. S. Holling, S. Carpenter, and A. Kinzig. 2004. Resilience, adaptability and transformability in social–ecological systems. Ecology and Society 9:5.

Walsh, J., et al. 2014. Chapter 2: Our changing climate. Pages 19–67 *in* J. M. Melillo, T. C. Richmond, and G. W. Yohe, editors. Climate change impacts in the United States: the third national climate assessment. U.S. Global Change Research Program, Washington, D.C., USA.

Weisberg, P. J. 2004. Importance of non-stand-replacing fire for development of forest structure in the Pacific Northwest, USA. Forest Science 50:245–258.

Westerling, A. L. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences 371:20150178.

WFRS (Westside Fire Regime Summit). 2017. Fire in the Pacific Northwest – past, present and future. http://www.nwfirescience.org/events/westside-fire-regime-summit

White, D. E., T. Atzet, and P. A. Martinez. 2002. Dead wood and fire relationships in Southwestern Oregon Western Hemlock Forests 1. Pages 49–487 *in* W. F. Laudenslayer, P. J. Shea, B. E. Valentine, C. P. Weatherspoon, and T. E. Lisle, editors. Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests. PSW-GTR-181. USDA Forest Service, Pacific Southwest Research Station, Reno, Nevada, USA.