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Review and synthesis

Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California

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ABSTRACT

Increasingly, objectives for forests with moderate- or mixed-severity fire regimes are to restore successionally diverse landscapes that are resistant and resilient to current and future stressors. Maintaining native species and characteristic processes requires this successional diversity, but methods to achieve it are poorly explained in the literature. In the Inland Pacific US, large, old, early seral trees were a key historical feature of many young and old forest successional patches, especially where fires frequently occurred. Large, old trees are naturally fire-tolerant, but today are often threatened by dense understory cohorts that create fuel ladders that alter likely post-fire successional pathways. Reducing these understories can contribute to resistance by creating conditions where canopy trees will survive disturbances and climatic stressors; these survivors are important seed sources, soil protectors, and critical habitat elements. Historical timber harvesting has skewed tree size and age class distributions, created hard edges, and altered native patch sizes. Manipulating these altered forests to promote development of larger patches of older, larger, and more widely-spaced trees with diverse understories will increase landscape resistance to severe fires, and enhance wildlife habitat for underrepresented conditions.

Closed-canopy, multi-layered patches that develop in hot, dry summer environments are vulnerable to droughts, and they increase landscape vulnerability to insect outbreaks and severe wildfires. These same patches provide habitat for species such as the northern spotted owl, which has benefited from increased habitat area. Regional and local planning will be critical for gauging risks, evaluating trade-offs, and restoring dynamics that can support these and other species. The goal will be to manage for heterogeneous landscapes that include variably-sized patches of (1) young, middle-aged, and old, closed-canopy forests growing in upper montane, northerly aspect, and valley bottom settings, (2) a similar diversity of open-canopy, fire-tolerant patches growing on ridgetops, southerly aspects, and lower montane settings, and (3) significant montane chaparral and grassland areas. Tools to achieve this goal

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include managed wildfire, prescribed burning, and variable density thinning at small to large scales. Specifics on "how much and where?" will vary according to physiographic, topographic and historical templates, and regulatory requirements, and be determined by means of a socio-ecological process.

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Contents

1. Introduction

1.1. Scope and extent of mixed severity fires

Mixed-severity fires are common in dry and moist mixed-conifer, ponderosa (*Pinus ponderosa*), and Jeffrey pine (*Pinus jeffreyi*) forests of the Inland Pacific Western US (Fig. 1), and in many other mixed conifer forests throughout the intermountain West, where summers are typically hot and dry (Collins and Stephens, 2010; Hessburg et al., 2007; Larson and Churchill, 2012; Odion et al., 2014; Perry et al., 2011: Fig. 3b, Schoennagel et al., 2004; Beaty and Taylor, 2001, 2008; Taylor and Skinner, 1998, 2003; Bekker and Taylor, 2008, 2010). As defined here, mixed-severity fires (hereafter, *MSFires*) roughly comprise the interquartile range of fire severities, where 20–70% of the dominant tree basal area or canopy cover *of a given patch of forest* is killed by any single instance of fire (Agee, 1993).

Areas of relatively homogenous fire mortality effects (often within a much larger fire event area) typically *define the size, shape, and extent of fire severity patches*, including mixed-severity, which historically were often shaped by prevailing topographic features. Fire severity patches commonly occurred in a continuum of sizes between $\sim 10^0$ and 10^3 ha; larger patches were also possible, but were historically rarer in number than those in this more common range of sizes (Moritz et al., 2011; Perry et al., 2011). However, these larger patches usually accounted for a large area burned by MSFires, and large fire area burned varied significantly by ecoregion (Moritz et al., 2011). Note that to be defined as MSFire, patches $>10^3$ ha did not burn with complete tree mortality, rather, individual trees and clumps of various sizes would have survived consistent with the definition. Indeed, the overall patchiness of a large landscape over space and time is the result of variation in disturbance severity (Pickett and White, 2013).

A mixed-severity fire regime forest is one, where over space, MSFires tend to naturally dominate, but not to the complete exclusion of occasional low- or high-severity fires over time. With high- and low-severity fires, >70% and <20% of the dominant tree basal area or canopy cover of a patch is killed by any single instance of fire, respectively (Agee, 1993). Mixed-severity fire regime forests (hereafter, *MSForests*) are poorly understood in comparison with those where either high- or low-severity fires dominate. One reason is that the mixed-severity class is a "catch-all bin" for what remains after the more clearly defined, end member, low- and high-severity classes are accounted for. Another is that while MSFires may commonly occur in a patch of forest, there is additional temporal variability in severity to be considered too. Over multi-centenary time frames, an individual patch of forest can characteristically experience MSFires, but occasionally experience low or high-severity events, over all or part of the area (e.g., Arno et al., 1995; Agee, 1993, 2003; Perry et al., 2011; Hopkins et al., 2014). Some use this notion of temporal variability in fire severity to refashion a more liberal definition of MSFire than used here, which essentially includes every forest type (Odion et al., 2014). However, we constrain our definition to describe MSForests as those where over space and time, MSFires tend to naturally dominate.

Postfire conditions after MSFires are some of the most structurally variable (Belote et al., 2015; Halofsky et al., 2011). Conditions within a patch can range from relatively high tree survival after primarily surface fires, with only modest amounts of individual tree and group torching (i.e., 20–50% of the dominant tree basal area or canopy cover is killed), to mixed surface and crownfires, where more trees are killed than survive (i.e., 51–70% of the dominant tree basal area or canopy cover is killed, Fig. 2). We also refer the reader to Perry et al. (2011) – *The ecology of mixed*

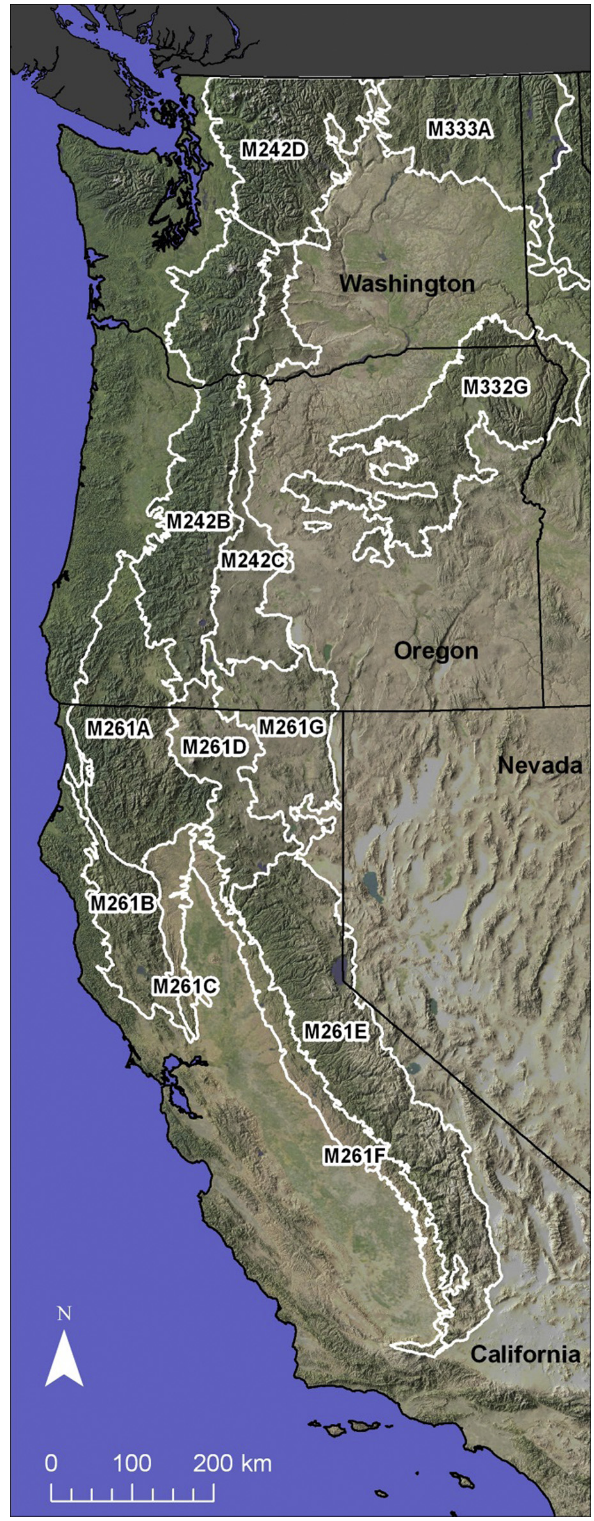

Fig. 1. Bailey Sections and Subsections in Oregon, Washington, and northern California, with forest vegetation types that often display mixed severity fire regimes (Bailey, 1995, 2009, http://svinetfc4.fs.fed.us/clearinghouse/other_resources/ecosubregions.html). Section M261A has been modified along the Southwest Oregon coast according to Frenkel (1993) to exclude an area of mostly high severity regimes. Most Sections also contain areas of either low-severity or high-severity regimes (or both). Section alphanumeric codes are: M242B (Western Cascades), M242C (Eastern Cascades), M261A (Klamath Mountains), M261B (Northern California Coast Ranges), M261C (Northern California Interior Coast Ranges), M261D (Southern Cascades Section), M261E (Sierra Nevada), M261F (Sierra Nevada Foothills), M261G (Modoc Plateau), M332G (Blue Mountains), M333A (Okanogan Highlands).

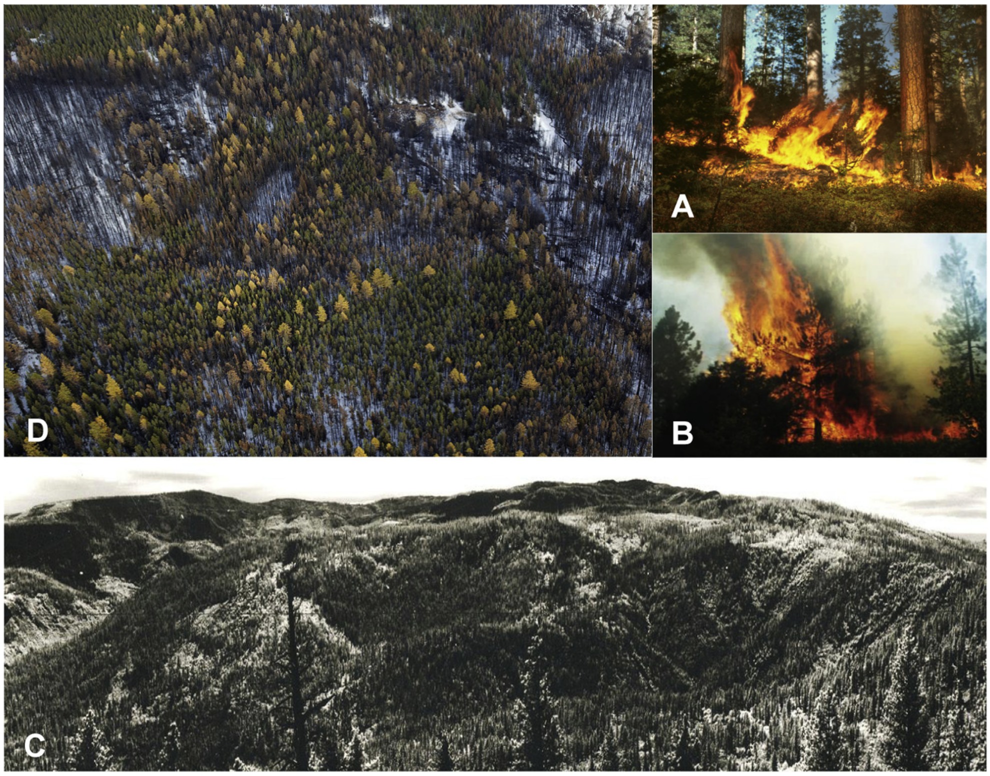

Fig. 2. Photographs of surface (A) and crownfire (B) behavior associated with mixed-severity fires, where 20–70% of the dominant basal area or canopy cover may be killed by the sum of all surface and crownfire effects. Panel (C) shows a typical mixed conifer forest in the eastern Washington Liberty-Beehive area that was historically frequented by mixed-severity fire. The historical photo was taken in 1934 by Albert Arnst; photo courtesy of the National Archives and Record Administration, Seattle, WA. The modern-era photo (D) was taken by John Marshall in 2012, after the Wenatchee Complex Fire. The photo in (D) was taken just months after the fire, on the occasion of the first snowfall, to highlight the mixed-severity effects.

Fig. 3. Repeat panoramic photographs of the Leecher Mountain area, Methow Valley, WA. In the 1930 black and white photo (above), dry mixed conifer forests are apparent. These forests were regularly burned by frequent lightning ignited fires, and by Native Americans, until the start of the 20th century. Notice how frequent fires maintained open canopy forest conditions and extensive areas of grassland cover in the top photo. Note how densely forested the same area has become in the 2011 bottom photo and that many grassland areas now support dense forest cover. Top photo courtesy of the National Archives and Record Administration, Seattle, WA, from the William Osborne Collection. Bottom photo courtesy of John Marshall Photography.

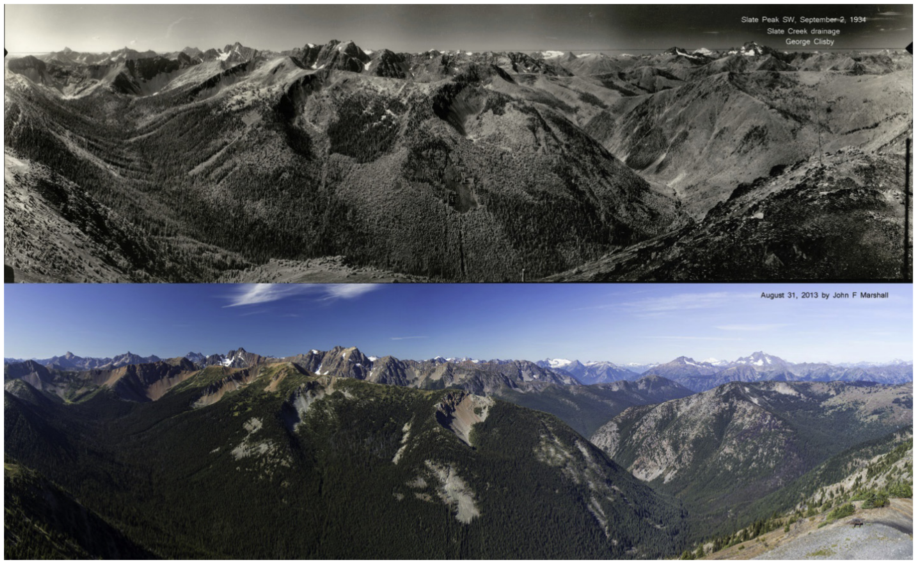

Fig. 4. Repeat panoramic photographs of the Slate Peak area, Slate Creek drainage, near Mazama, WA. In the 1934 black and white photo (above), cold site lodgepole pine, subalpine fir, and Engelmann spruce forests are apparent. These forests were burned by lightning ignited fires until well into the 20th century. Notice the exceptional variety in lifeform cover and of forest successional conditions (burned area, recovering forest, early seral grasslands and shrublands, patches of seedlings and saplings, young, intermediate and mature forests). While fires visited individual patches with relative infrequency, there was much evidence of recent fires nearby. Thus, infrequent, high severity fire does not connote a lack of fire on the landscape. Rather, it connotes low frequency in individual patches of forests, but the surrounding landscape may show much evidence of ongoing fires nearby. This is the change of consequence in forests of the high-severity regime. Notice also the recent landslides, avalanche chutes, and hillslope failures in recently burned areas of the 1934 photo, and their relative absence in the 2013 repeat photo. Top photo courtesy of the National Archives and Record Administration, Seattle, WA, from the William Osborne Collection. Bottom photo courtesy of John Marshall Photography.

severity fire regimes in Washington, Oregon, and Northern California, which is a companion paper to this one. In Perry et al. (2011), we expand on our discussion of the ecology and spatial geography of mixed severity fires, as a foundation for discussions about management of these same forests.

MSForests occupy a wide range of environments (Stine et al., 2014), and they historically exhibited a characteristic patch size distribution, revealing many small and fewer large patches, which resulted in high alpha, beta, and gamma diversity (Collins and Stephens, 2010; Hessburg et al., 2007, 2015; Moritz et al., 2011; Perry et al., 2011: Fig. 1a and b). That diversity derived primarily from fire interactions with vegetation structure, topography, and weather variations at the time a fire occurred. Over time these interactions contributed to a considerable pyrodiversity as well (*sensu* Martin and Sapsis, 1992). Pyrodiversity as defined here encompasses the broad spatial and temporal variability in fire frequency, severity, seasonality, distribution, and extent of fires naturally associated with all vegetation types. In fire-prone regions, pyrodiversity drives biotic, successional patch, and habitat diversity.

1.2. Recent changes in MSForests

Pyrodiversity in many MSForests has been simplified by the cumulative effects of past management, environmental changes arising from climatic warming (Abatzoglou and Kolden, 2013; Cansler and McKenzie, 2014), and increasingly larger and more severe wildfires. Because the structural and compositional diversity in MSForests is largely dependent on the prevailing disturbance regime, restoring pyrodiversity is central to restoring MSForests and perhaps most others in the western US (Hessburg et al., 2015). The variety in pre-management era spatial patterns of forest cover types, tree density, canopy cover, tree sizes and ages, and forest successional conditions was an emergent property of the pyrodiversity of each forest type. No two landscapes were alike. Variation in landscape patterns of physiognomic types too created unique fire regime interactions between types (Lauvaux et al., 2016; Odion et al., 2010). For example, in landscapes with mixed forest and grassland/shrubland conditions, grass-fire/shrub-fire cycles were often influential to adjacent forest fire frequency and severity.

Pyrodiverse conditions have been broadly simplified by the combined effects of a century of fire suppression, fire exclusion by livestock grazing and road building, selection cutting in dry forests, and clearcut logging in more productive moist forests. Shade-tolerant Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), white fir (*Abies concolor*), and subalpine fir (*Abies lasiocarpa*) now dominate in many areas formerly occupied by fire-tolerant and shade-intolerant ponderosa pine, Jeffrey pine, western white pine (*Pinus monticola*), sugar pine (*Pinus lambertiana*), and western larch (*Larix occidentalis*). This has simplified species diversity at patch and larger scales. South-facing aspects and ridgetops and lower montane settings were home to open-canopy forests, and fairly large areas of open woodlands, shrublands, and grasslands (Fig. 3). These were primarily maintained by frequent fires – low to mixed-severity in the forests and woodlands, and high-severity in the shrub- and grasslands. North-facing slopes and valley bottoms and upper montane settings were home to closed-canopy, multi-layered forests, but also shrublands and meadows, and these were primarily maintained by moderately frequent to infrequent mixed- and high-severity fires (Fig. 4). Mid-montane settings were a complex mixture of the two preceding examples and both open and closed canopy forests were present. Time-since-fire, topographic setting, and the severity of prior fires would typically dictate the severity of subsequent fires. Fire frequency in mid-montane environments varied from frequent to moderately infrequent (Fig. 5). These differences are no longer as starkly obvious as they once were, and changes have simplified successional

Fig. 5. Repeat panoramic photographs of the Stafford Creek drainage, Kittitas County, Washington. In the 1934 black and white photo (above), moist mixed conifer forests are apparent on north aspects and valley bottoms, dry mixed conifer forest persist on south aspects and ridges. These forests were burned by lightning ignited fires until well into the 20th century. Notice in the middle ground the complex structure of forest patches facing north (the angle of view is southwest) and the simpler open canopy structure of south facing and ridgetop forests in the foreground. On north facing aspects, mixed-severity fires tended to be most prevalent, but evidence for high-severity fires is also present in a number of patches. Much of this complexity has been lost in the current condition (2013, bottom photo) absent fire. Top photo courtesy of the National Archives and Record Administration, Seattle, WA, from the William Osborne Collection. Bottom photo courtesy of John Marshall Photography.

pattern diversity and pyrodiversity at patch and larger scales. Conserving and restoring these diverse fire regimes will be an enormous challenge (Spies et al., 2012), but it is the perhaps the most important challenge ahead.

Climate change is stressing forests worldwide as environmental conditions change at rates exceeding the adaptive capacity of some species and communities (Allen et al., 2010, 2015; Dai, 2011), and MSForests are no exception. Uncharacteristically large wildfires and insect outbreaks have become more common in most forest types, and will likely continue (Logan et al., 2003; Westerling et al., 2006; Miller et al., 2009; Littell et al., 2008, 2009; McKenzie et al., 2004; Pechony and Shindell, 2010; Rogers et al., 2011; Stavros et al., 2014). Significant impacts on MSForest hydrology are evident too. A warming climate will likely continue driving these trends (Chmura et al., 2011; Hay et al., 2011; Luce and Holden, 2009; Pederson et al., 2011).

Warming temperatures and low plant-available water are currently reflected in declining MSForest resilience[1] (Allen et al., 2010; Dale et al., 2001; Sturrock et al., 2011); tree mortality has increased and is associated with warming and drying (Bentz et al., 2010; Bigler et al., 2007; Breshears et al., 2005; Guarin and Taylor, 2005; Hicke et al., 2006; Lutz and Halpern, 2006; Raffa et al., 2008; Smith et al., 2015; van Mantgem et al., 2009; Williams et al., 2013). The combined effects of reduced snowpack, earlier springs, warming winter and summer temperatures, and hotter summer droughts have triggered chronic bark beetle (*Dendroctonus* spp., *Ips* spp., *Scolytus* spp.) outbreaks in nearly all forest types, at levels not seen in 125 years (Allen et al., 2015; Bentz et al., 2010; Raffa et al., 2008). High leaf area in dense stands can produce high water demand and ensuing drought stress (Lutz et al., 2010; Stephenson, 1998; Waring and Running, 2010) that can be further exacerbated by warming (Chmura et al., 2011). For example, at Blacks Mountain Experimental Forest, a MSForest in northeastern California, half of the trees >60 cm diameter at breast height (DBH) died in unthinned plots between 1934 and 1998, while few trees died during that time in thinned plots (Ritchie et al., 2008). Additional research relating tree density, forest type, and structure to water use and predicted future seasonal availability would help land managers better adapt future forests to climate change.

Throughout western North America, effects of climate change on MSForests are compounded by prior timber harvests and fire exclusion, which, via regeneration and release of shade-tolerant conifers, significantly increased tree density and abundance of young relative to older tree cohorts (Fettig et al., 2007; Halofsky et al., 2011; Hessburg and Agee, 2003; Hessburg et al., 2005; Loudermilk et al., 2013, 2014; Perry et al., 2011; Raffa et al., 2008). This shift in age structure manifests at patch to regional landscape scales (Larson and Churchill, 2012; Perry et al., 2004, 2011; Taylor, 2004; Brown and Wu, 2005; Hessburg et al., 2000a, 2005; Haugo et al., 2010, 2015; Naficy et al., 2010). For example, in eastern Oregon and Washington, a cover type transition from drought and fire-tolerant to intolerant species is accompanying the shift in age structure (Hagmann et al., 2013, 2014, Hessburg et al., 1999a, 1999b, 2000a; Merschel et al., 2014; Perry et al., 2004; Stine et al., 2014).

In many MSForest areas, the shift in age structure, density, and species composition stems from fire exclusion, and selection and clearcut harvesting that led to the loss of widely distributed remnant large and old trees, patches of old forest, and of naturally recovering early successional communities (Hessburg et al., 2000a, 2005; Hessburg and Agee, 2003; Beaty and Taylor, 2008; Swanson et al., 2010). In others, the transitions occurred under the influence of fire exclusion (and often livestock grazing), absent logging (Fig. 6). These simplified landscape patterns reduce biotic diversity and increase the risk of large, spreading disturbances that jeopardize remaining old forest patches (Binkley et al., 2007;

[1] We adopt the definition of resilience provided by the Resilience Alliance (http://www.resalliance.org/resilience) as "the capacity of a social–ecological system to absorb or withstand perturbations and other stressors such that the system remains within the same [succession and disturbance] regime, essentially maintaining its structure and functions. It describes the degree to which the system is capable of self-organization, learning and adaptation (Holling, 1973; Gunderson and Holling, 2002; Walker et al., 2004).

Fig. 6. Mixed conifer forest change in Lassen Volcanic National Park. Top left 1923 (Weislander), top right 1993 (Taylor), bottom left 2009 (Taylor), bottom right 2013 (Taylor). The photos of the stand were taken from a GLO survey marker. The forest was never logged. A functioning mixed-severity fire regime was present in these forests until 1905 when fire suppression was implemented. A wildfire burned through the park in 2012. Additional details on fire and forest change available in Taylor (2000).

Moritz et al., 2011; Perry et al., 2011; Raffa et al., 2008; Kitzberger et al., 2012, Fig. 6).

Clearcutting has been eliminated and post-fire logging is much reduced on federal lands (Moeur et al., 2005), but these practices continue apace on other ownerships. Legacies of unprecedented 19th and 20th-century anthropogenic disturbance (e.g., timber harvesting, mining) are evident at quite large spatial scales (e.g., see Loudermilk et al., 2013), raising serious questions about the long-term resilience of some current landscapes. In southwest Oregon, for example, soil food webs of MSForests originating after severe wildfires in the late 1800s have yet to recover (Perry et al., 2012). In this same region, forest areas that burned at high-severity and were planted after a fire in 1987, reburned again 15 years later at high-severity, raising the likelihood of a very large scale, disturbance mediated switch from forest to shrublands (Nagel and Taylor, 2005; Thompson and Spies, 2010; Collins and Roller, 2013). While patches of grass or shrubland are a key element of MSForest diversity, amount and configuration play an important role in the overall fire ecology of the surrounding landscape, and they frame questions concerning long-term resilience.

1.3. Management challenges in MSForests

Management direction has been widely discussed for forest types that were historically dominated by frequent surface fires, so-called low-severity forests (e.g., Allen et al., 2002; Agee and Skinner, 2005; Franklin and Agee, 2003; Franklin et al., 2013; Hessburg and Agee, 2003; Hessburg et al., 2005; Kaufmann et al., 2007; Noss et al., 2006; Stephens et al., 2009). However, MSForests pose unique challenges to managers because their successional pathways and disturbance patterns are so highly varied (Tepley et al., 2013; Larson et al., 2013). For example, an ongoing management problem in MSForests on public lands has been balancing the competing goals of introducing managed wild and prescribed fires to restore more characteristic successional patterns and disturbance processes, while protecting late-successional forest habitats from uncontrolled and damaging wildfires (Brown et al., 2004; Gaines et al., 2010a, 2010b; Noss et al., 2006; Spies et al., 2006, 2010; Zielinski, 2014). The wildfire threat to late-successional habitats is primarily associated with high surface fuel loads and ladder fuels provided by understory cohorts of pole to small-sized trees (Franklin et al., 2000; Keane, 2014; Roberts et al., 2011, 2015; Zielinski et al., 2013). Reducing these surface and ladder fuels can greatly reduce the likelihood of severe fire behavior (Agee and Skinner, 2005) without substantially altering the late-successional structure of older forests. Conceptual frameworks are needed at patch- to landscape-scales to guide managers seeking to lower the risk of large, high-severity dominated fires, reduce losses to drought and insects, and restore multi-scale habitat heterogeneity (Franklin and Johnson, 2012; Hessburg et al., 2015; North et al., 2014; Stephens et al., 2010; Spies et al., 2012).

Restoration needs of MSForests are the subject of debate. Hanson et al. (2009, 2010), Baker (2012), Williams and Baker (2012), Dellasala and Hanson (2015), Odion et al. (2014), and Baker (2015) have argued against fuels reduction or landscape restoration of any magnitude in Inland West pine and mixed-conifer forests. They provide evidence that current large patches of high-severity fire may be within the historical range of variability for these forests, and the risk of loss of dense multi-storied forest to high-severity fire is relatively low. Likewise, they suggest that widespread and ecologically important changes have not occurred in these forests in the last century, and that restoration activities on any significant scale are unjustified. Their inferences are based on conclusions drawn from vegetation reconstructions using General Land Office (GLO) or federal Forest Inventory and Analysis (FIA) data. While these data are useful for general descriptions and tabulations of historical vegetation conditions, they are unsuited to making spatially accurate inferences as to local historical vegetation conditions, or for inferring disturbance regimes from size distributions of trees (Fulé et al., 2014; Stevens et al., in press). While needs vary both regionally and locally, we strongly disagree with the contention that ecological restoration is unnecessary in MSForests of the Inland Pacific West, as do a host of authors throughout the Inland West: Barth et al. (2015), Collins et al. (2011a, 2015), Gaines et al. (2010a, 2010b), Spies et al. (2010), Hessburg and Agee (2003), Hessburg et al. (1999a, 1999b, 2000a, 2005, 2013, 2015), Taylor (2004), Stephens et al. (2009, 2010, 2015), Moghaddas et al. (2010), Scholl and Taylor (2010), Hagmann et al. (2013, 2014), Merschel et al. (2014), Perry et al. (2011), Harris and Taylor (2015), and Franklin and Johnson (2012). However, we recognize the importance of stand-replacing fire in appropriate forest types, and at appropriate spatial and temporal scales. The goal of ecological restoration is not to eliminate severe fire, but to have it resume a more characteristic role.

Following, we discuss the scientific basis for nine strategies aimed at reconciling potentially conflicting management goals in MSForests. We discuss the usefulness of each strategy as part of an ecological framework for management and conservation.

Strategy (1): Landscape-level approaches to restoring pyrodiversity.
Strategy (2): Protecting and restoring large and old, early-successional tree abundance.
Strategy (3): Expanding use of prescribed and wildfires to restructure forests.
Strategy (4): Using topography to tailor restorative treatments to the landscape.
Strategy (5): Rehabilitating plantations.
Strategy (6): Creating and maintaining successional heterogeneity.
Strategy (7): Integrating restoration with late-successional forest habitat needs.
Strategy (8): Mitigating threats from climate change, forest insects, and pathogens.
Strategy (9): Creating and maintaining early-successional forests.

2. Management strategies

2.1. Strategy 1: Landscape-level approaches to increasing pyrodiversity

2.1.1. Current pyrodiversity is atypically simple

Wildfire size and severity have increased in many MSForests (Cansler and McKenzie, 2014; Meigs et al., 2009) in recent decades, and landscape patterns of resulting successional conditions are undergoing simplification. Recent wildfire size has grown due to the combined influences of a changing climate and from past land management practices (Higuera et al., 2015). Furthermore, most fires (>95%) are quickly suppressed each year (Calkin et al., 2014, 2015), and those that escape initial attack generally burn under extreme weather conditions (Calkin et al., 2014, 2015) and become large. Fire severity is increasing by virtue of these same dynamics, and patches of high-severity fire tend to be uncharacteristically large and homogenous (Cansler and McKenzie, 2014). The combination of these two factors has led to a significant 'fire deficit' in forests and a 'fire surplus' in rangelands (Parks et al., 2015b).

2.1.2. A complex pyrodiversity as a bet-hedging strategy

Fostering a complex pyrodiversity is a useful bet-hedging strategy in any climate because it tends to encourage variation in fire size and severity. Historical variation in patch sizes of severity classes and spatial heterogeneity within severity patches was important because it fostered a multi-scale diversity of successional and lifeform conditions (Hessburg et al., 1999b; Larson et al., 2013; Swanson et al., 2010). Disturbances such as fire drive variation in successional and lifeform patterns across spatial scales, which in turn drives the extent and severity of future disturbances (Turner, 1989).

2.1.3. Need to restore a more characteristic pyrodiversity

Landscape-level approaches are needed that reduce live and dead fuel connectivity and limit large crown fires. Using appropriate combinations of prescribed and managed wildfire, and/or mechanical treatments (e.g., see Collins et al., 2014), management can be tailored to topography (see *Strategy 4*) and other recent fire event boundaries to alter the severity of future disturbances, both within and beyond the treatment boundaries. With modern-era prescribed fire and mechanical treatments, there is substantial evidence accumulating that fire hazard reduction and ecological restoration objectives can be accomplished with few unintended long-term consequences to soils and vegetation, small mammals, songbirds, bark beetles, and carbon sequestration (see Reinhardt et al., 2008; Stephens et al., 2012a). The data are more equivocal for species like the Pacific fisher (*Pekania pennant*) in California (Truex and Zielinski, 2013), yet recent modeling (Scheller et al., 2011) and empirical work (Zielinski et al., 2013) suggest that fishers too can tolerate the amount of restorative treatments (mechanical + prescribed fire) that may be needed to reduce fire spread rate (Syphard et al., 2011).

2.1.4. Lessons from fuel treatment simulation studies

Numerous studies have used fire spread models to examine the value of strategically placed fuel treatments across local and regional landscapes (Finney, 2001, 2004, 2007; Finney et al., 2007; Stratton, 2004). The basic premise is that an informed deployment of treated areas only covering part of the landscape can modify fire behavior on some portion of the untreated landscape. Various criteria have been used to inform the deployment of modeled or actual treated areas. One early method was to network discontinuous but spatially layered fuel treatments termed "*s*trategically *p*laced *l*andscape *a*rea *t*reatments" or SPLATs (Finney, 2001; Finney et al., 2007; Bahro et al., 2007; Schmidt et al., 2008) – an approach Loehle (2004) likened to the arrangement of bulkheads on a ship. Later modeling employed *d*efensible *f*uel *p*rofile *z*ones (DFPZ's, Moghaddas et al., 2010), prioritizing treated areas according to stocking density (Ager et al., 2010; Collins et al., 2011b), and protecting special value areas, such as threatened habitats for high priority species (Ager et al., 2007, 2012; Gaines et al., 2010a, 2010b; Kennedy et al., 2008; Lehmkuhl et al., 2007; Scheller et al., 2011; Syphard et al., 2011; Roloff et al., 2012), or urban/exurban development concentrations (Ager et al., 2010). Various studies have compared approaches (e.g. Schmidt et al., 2008) or modeled tradeoffs among strategies aimed at protecting potentially competing values (e.g. Kennedy et al., 2008; Ager et al., 2010).

Box 1 Key concepts from fire simulation studies.

(Schmidt et al., 2008; Collins et al., 2009, 2010, 2011a, 2011b; Finney et al., 2007; Ager et al., 2007; Moghaddas et al., 2010; Syphard et al., 2011):

- When treatments can be strategically placed (whether informed by knowledge of local fire patterns or through spatial optimization algorithms), reducing fuels on a portion of the landscape can substantially alter fire behavior on the larger landscape. For example, with as little as 15–25% of the landscape strategically treated, simulated fire size, flame length, and spread rate were reduced in treated vs. untreated scenarios (Ager et al., 2007, 2010; Collins et al., 2010; Moghaddas et al., 2010; Ritchie et al., 2007; Schmidt et al., 2008).
- In studies that simulated a range of treated area, increasing area treated improved protection of the whole landscape, but with a tendency for diminishing returns. For example, in Ager et al. (2007), a non-linear response to the amount of area treated to protect existing northern spotted owl (NSO, *Strix occidentalis caurina*) habitat was evident: treating 20% of the non-owl habitat area reduced the probability of habitat loss by 50%; doubling the treated area to 40% reduced the probability of habitat loss by 75%.
- When ⩾50% but less than the full area was available for treatment (due to reserved areas), randomly placed treatments necessitated a substantially greater treatment area than optimized treatments to achieve the same effect (e.g., 2–3× in Finney et al., 2007; Schmidt et al., 2008).
- When the proportion reserved from treatment due to land allocation (e.g., wilderness, roadless, riparian buffers) approached 45%, random and optimized treatment approaches did not differ; i.e., there was no effect of optimization (Finney et al., 2007; Schmidt et al., 2008).

Moghaddas et al. (2010) and Collins et al. incorporated actual landscape fuel treatment networks in their studies. The addition of DFPZ's coupled with spatial linkage to earlier fuel reduction treatments lowered the total area burned by 40%, compared to pre-treatment conditions, with the greatest reduction in area affected by moderate and high flame lengths. Conditional burn probability was reduced by 21–32% in California spotted owl (CASPO) habitat; however, when those closed-canopy, multi-layered forests did burn under post-treatment conditions, the modeled proportion of active vs. passive crown fire was 2–3 times greater than that in the DFPZ's. Schmidt et al. (2008), Prichard et al. (2010), and Prichard and Kennedy (2014) found similarly that simulated and actual wildfire area burned and area burned severely were reduced via the influence of earlier fuel reduction treatments (Kennedy and Johnson, 2014).

2.1.5. Navigating social and ecological trade-offs

In choosing among the options for type, intensity, size, and placement or pattern of fuel treatments, there are often social and ecological trade-offs associated with either reducing potential fire behavior or protecting other resources (see *Strategy 7*). Thus, in some instances it is not possible to locate fuel treatments so as to optimize effects on fire behavior (Collins et al., 2010; Moghaddas et al., 2010). Under these circumstances, treating a much larger area in non-optimal areas becomes the trade-off (Finney et al., 2007; Schmidt et al., 2008), unless prescribed burning alone or in combination with understory thinning can be allowed in otherwise protected areas. We note that prescribed burning alone in many reserve areas is not realistic due to the high initial hazard of active or passive crown fire.

Land allocation is often a significant factor when treating federal forest lands, particularly where fixed area reserves are used to allocate habitat for protected species (e.g., spotted owls – *Strix* spp., Pacific fisher, Spencer et al., 2011). Furthermore, regulations on forest management within and around nest stands and natal dens, along with those for riparian buffer zones, often affect the placement and pattern of fuel treatments to a high degree (Moghaddas et al., 2010), thwarting most efforts to spatially optimize them. This is a critical consideration when planning fuel treatments.

2.1.6. The need for ongoing fuel treatments

Since forests are living, growing systems, treatments will have a characteristic life expectancy (Collins et al., 2010, 2011a, 2011b; Hudak et al., 2011), and life expectancy will vary by treatment intensity and other environmental factors. A single treatment will not permanently "fix" the problem, even where treatment intensity is high (e.g., a large basal area reduction). As the time since treatment lengthens, tree and understory growth responses rebuild fuel load, fuel ladders, and surface and canopy fuel continuity (Agee and Skinner, 2005; Collins et al., 2010, 2013; Miller et al., 2009); without follow-up action, treatment ability to influence fire behavior declines (Vaillant et al., 2009; Stephens et al., 2012b). Where recurring frequent fires once maintained low fire hazard conditions, the continued suppression of fire today necessitates regular planned burning on par with the natural fire frequency. Thus, the design of landscape-level fuel treatments involves either spatially optimizing treatments where practicable, or treating a large fraction of the landscape initially, and then subsequently using managed wildfire or prescribed burning treatments to maintain treatment effectiveness (Finney et al., 2007). Both approaches are significant investments in land management that have yet to be realized over large landscapes.

2.1.7. Operational limitations on treatment placement

In addition to ecological considerations to treatment placement, there are practical considerations as well. For example, roadless, wilderness, and other administratively withdrawn areas, where new road construction is neither feasible nor desirable, can constrain treatment methods that may be considered. Remaining methods (e.g., prescribed burning) may not be feasibly applied due to surface and/or crown fuel conditions. Where thinning is a reasonable approach, slope and road access conditions will drive logging and yarding systems, each of which are significant cost considerations. Even where road access is possible, and slopes are gentle enough to allow for relatively low cost thinning and ground-based yarding systems, the available understory timber may be of insufficient value or quality to recover treatment costs. Moreover, mill infrastructure has declined dramatically over the last 30 yr making hauling costs often prohibitive. Considerations like these will place significant constraints on where treatments can be located and affordable. In many cases, restorative treatments may require subsidy. Decision support tools would be helpful to sorting out these operational considerations (Reynolds et al., 2014).

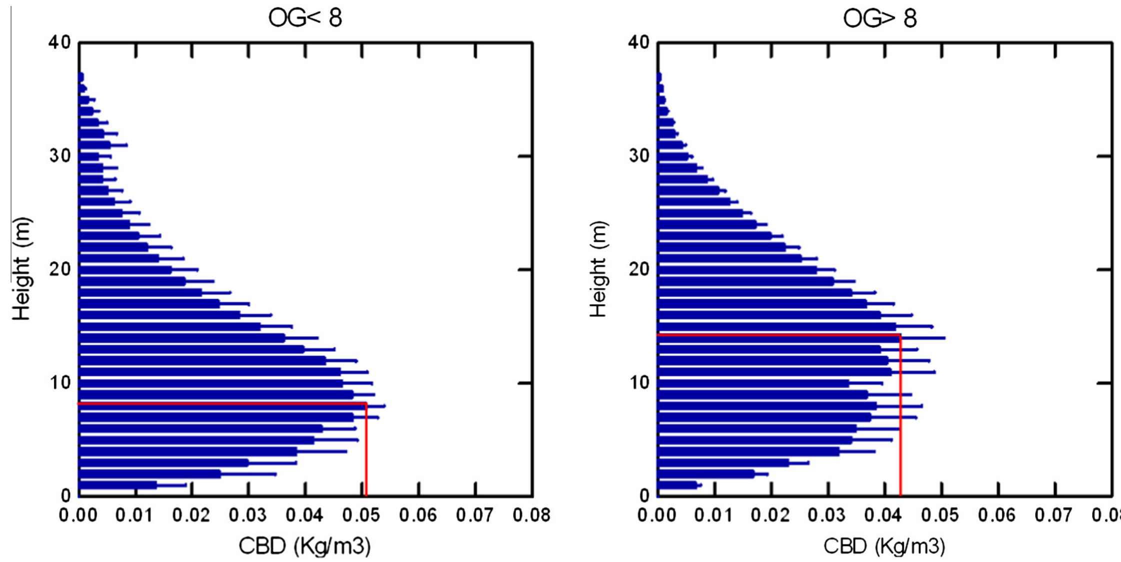

Fig. 7. Canopy bulk density (CBD) profiles as affected by the presence of living large (>80 cm) old growth trees or large stumps, where old growth trees were removed. Old growth (OG) is defined as older than 150 years. OG < 8/> 8 = fewer/more than 8 living large OG trees or stumps per 0.2 ha plot (8 large trees/0.2 ha plot = 40 large trees/ha). Stumps larger than 80 cm were assumed to have been living large OG trees that significantly influenced development of the existing canopy at the time of measurement. Bars = standard errors of the mean of 5 plots <8, and 9 plots >8). Figure is adapted from data presented in Perry et al. (2004). Red lines represent the height of the maximum CBD.

2.2. Strategy 2: Protecting and restoring large and old, early successional tree abundance

Large, old trees of early seral species were a core constituent of many historical MSForests, especially those that saw frequent low and MSFires. Mature, relatively open forests composed of *l*arge and *o*ld, *e*arly *s*eral *t*rees (hereafter, LOEST) with limited nearby fuel ladders are highly resistant to active crownfires, especially when coupled with low surface fuel loads (Agee and Skinner, 2005; Binkley et al., 2007; Thompson and Spies, 2010; Stephens et al., 2009; Perry et al., 2011; Taylor et al., 2014; Weaver, 1959, 1961). "Large" and "old" are relative terms and will vary depending on tree species, forest type, topographic and edaphic conditions, and disturbance history (Van Pelt, 2008). LOEST are missing from many MSForests because of past selection cutting, which often targeted these trees.

Retaining LOEST of fire-resistant species is one of four principles Agee and Skinner (2005) provide in support of fuel reduction treatments. LOEST directly influence potential fire effects; for example, Belote et al. (2015) showed that different tree species and sizes had very different 10-year post-fire survival probabilities in Northern Rockies mixed-conifer forests, with large diameter western larch being the most fire resistant. These tree-level differences scaled-up to influence fire severity and effects at plot and stand levels (Belote et al., 2015), and the observed MSFire effects were largely explained by variation in the pre-fire abundance of large western larch.

2.2.1. Added benefits of retaining large, old early seral trees

The effect of LOEST goes beyond individual tree resistance though, as large trees can influence stand structure by exerting control over understory fuels. Through shading, a canopy of large trees limits the size and amount of ingrowth, even when ingrowth is shade-tolerant. For example, a study in central Oregon found that the maximum canopy bulk density was 25% greater and 6-m lower in the canopy of 0.2-ha plots with <8 large trees (<40 large trees/ha) compared to plots with >8 large trees (>40 large trees/ha, Fig. 7, Perry et al., 2004). If the ingrowth is flammable, as was the case in this example, the need for periodic fuels reduction would be increased by harvesting the large overstory trees.

2.2.2. Key steps to maintaining or increasing LOEST

Maintaining an extensive cover of LOEST involves three basic steps: (1) identifying environmental conditions that clearly support low and mixed-severity fire regimes; (2) protecting existing LOEST from crown fires, logging, and other stresses (e.g., drought and bark beetles) that can lead to mortality; and (3) developing future cohorts of LOEST at fire-resistant densities. Fire history studies and landscape reconstructions from historical aerial photography are helpful to identifying the (characteristics of) sites that normally supported low- and mixed-severity fire regimes (e.g., Lydersen and North, 2012; Merschel et al., 2014). At a minimum, protecting LOEST from crown fire involves removing (where feasible) fuel ladders and heavy surface fuels from their immediate vicinity, and in the broader surrounding landscape. Developing future cohorts of LOEST can involve natural regeneration processes following wild or prescribed fire, replanting desirable early seral species where preferred seed trees are generally absent, and newer approaches that recognize the value of maintaining and creating tree clump and gap diversity during stand development (*sensu* Oliver and Larson, 1996, see also Churchill et al., 2013; Hessburg et al., 2015; Larson and Churchill, 2012; Lindenmayer and Franklin, 2002; North et al., 2009; Fahey and Puettmann, 2008; Knapp et al., 2012). A logical starting point for increasing the presence of LOEST would be to protect early seral trees over a minimum size or estimated age (Franklin and Johnson, 2012; Franklin et al., 2013). Landscape assessments are also useful to document areas of LOEST depletion (e.g., see Hann et al., 1997; Hessburg et al., 1999a, 2000a; SNEP, 1996; Raphael et al., 2001; Ritchie and Harcksen, 2005). Where lacking, a strategy could be built around the existing distributions of LOEST (Kaufmann et al., 2007; Johnson et al., 2013).

Box 2 What to do with older late-seral trees?

While an ample presence of LOEST provides a relatively resistant landscape matrix, retaining older late-seral trees (e.g., *Abies* spp.) trees >150 yr, especially in valley bottom and north aspect settings, protects extant genetic diversity, unique tree attributes (e.g., cavities and epicormic branches), and pathological decadence that accrues with age (Van Pelt, 2008, cf. Miesel et al., 2009). Older, shade-tolerant trees also provide critical habitat for certain species (Bull et al., 1992)[2]. However, in many MSForests, the abundance of both young and relatively mature late-seral trees has increased dramatically during the 20th-century, filling in formerly open-canopy forests. For example, in plots within the mixed-conifer zone of the Deschutes National Forest, the ratio of white fir (*A. concolor*) to ponderosa pine (*P. ponderosa*) was 0.15:1, 0.6:1, 6:1, and 9:1 in the 200–150, 150–100, 100–50, and 50–0 yr age classes, respectively, with some individual white fir <100 yr old attaining DBHs of up to 60 cm (from Perry et al., 2004). In some areas, widespread replacement of early- by late-seral trees has significantly altered disturbance patterns and successional pathways, and produced an overall younger forested landscape that is more synchronized for large-scale disturbance. For these reasons, removal of immature and relatively mature, large, late-seral trees is justified, although it is beyond the scope of this paper to address where this will be true.

2.3. Strategy 3: Expanding use of prescribed and wildfires to restructure forests

2.3.1. Mechanical treatments are not an option in some forests

Portions of many landscapes are administratively withdrawn as National Parks or Wilderness areas, or they are too steep and remote for mechanical treatments to be a practical means of achieving restoration objectives (North et al., 2012). This is true over fairly large areas of the western US (e.g., see Habeck, 1976; Parks et al., 2014, 2015a; Miller et al., 2012; North et al., 2012, 2015). Under these circumstances, the use of either prescribed or wildfire then becomes a primary means to alter stand structure and reduce surface and ladder fuels (North et al., 2012), especially where the need of restoration is clearly established (Naficy et al., 2016). Unless existing barriers to application are modified (e.g., see the excellent legal review by Engel, 2013), it is unlikely that prescribed fire will be used in the near term to treat sufficiently large wildland areas that are upwind from human populations centers (Quinn-Davidson and Varner, 2012; North et al., 2012).

2.3.2. Managed wildfire is a promising approach

Resulting severity patterns from numerous 20th-century wildfires in northwestern California suggest that using managed wildfire under appropriate burning conditions may be a promising means to restore some aspects of forest resilience (see Box 3, Miller et al., 2012). Evidence for this approach is broadly apparent in landscapes that have experienced some repeat fire in the past century, at patch (Fulé and Laughlin, 2007; Larson et al., 2013) and landscape (Parks et al., 2014, 2015a) scales, in the Gila/Aldo Leopold Wilderness of New Mexico, on the north rim of the northern Arizona Grand Canyon, in the Frank Church Wilderness of Idaho, in the Klamath Mountains of northern California, in the Selway-Bitterroot Wilderness of Idaho and Montana, and in Glacier National Park and the Great Bear, Scapegoat, and Bob Marshall Wilderness Complexes of northwest Montana. For example, during outbreaks of wildfire in northwestern California since the late 1980s (e.g., 1987, 1999, 2006, 2008, 2012), fires in the steep terrain of the Klamath Mountains became large due to their sheer number and the extremely rugged topography. Under these circumstances, suppression resources can be overwhelmed, and fires can burn for weeks to months under less than severe conditions. The resulting fires often produced topographically driven severity patterns, much like those described from fire history reconstructions (Taylor and Skinner, 1998, 2003; Weatherspoon and Skinner, 1995; Jimerson and Jones, 2003; Skinner et al., 2006; Miller et al., 2012). Parks et al. (2014, 2015a) similarly found that previous wildfires limited subsequent fire spread in all four of their study areas, but the effect eroded with time since fire. They also found that the ability of fire to regulate subsequent fire growth was substantially reduced during extreme fire weather conditions (see also Collins et al., 2009). It appears that purposefully planning for the use of managed wildfire under these less than severe burning conditions would help to achieve long term goals of ecological restoration and high-severity fire risk reduction (Miller et al., 2012; North et al., 2012).

Box 3 An enlarged role for managed surface and crown fires.

Much of the work of restoring landscapes will likely need to be done using managed wildfires over large areas, with more intensive silvicultural and prescribed burning activities in key areas that require spatial precision of outcomes (e.g., Miller et al., 2012; North et al., 2012, 2014). Cutting trees can emulate fire effects on tree density and layering, but it cannot reproduce the effects of fire on nutrient cycling, snag creation, and fuel reduction (Stephens et al., 2012a, 2012b; McIver et al., 2013). In the past, tree cutting often resulted in the removal of now scarce large-sized trees to cover costs of harvesting, and it reduced snag densities to meet logging safety requirements, compacted soils, and left residual fine fuels on site that could promote future fire spread. Many of these effects can be avoided today by focusing attention on thinning out understories, removing the smaller trees that make up the bulk of the ingrowth, and with application of modern harvesting practices, improved seasonal timing, and better and more lightweight equipment. But without follow-up burn treatments, there is little chance that tree cutting alone will mimic fire effects for all other essential ecosystem functions (Schwilk et al., 2009). In contrast, management ignited or managed wildfires burning under moderate fire weather conditions can often accomplish ecological objectives without tree cutting, as has been observed in wilderness and roadless areas, and other managed forests where mixed- and high-severity fires naturally dominate. We emphasize that for managed fire to be effective, it must be allowed to burn under moderate weather conditions. If fire only occurs under extreme weather conditions, as has happened in many recent wildfires, fire effects will tend to be severe and not achieve desired ecological outcomes.

[2] (Bull et al. (1992) provide management guidelines.)

2.4. Strategy 4: Using topography to tailor restorative treatments to the landscape

2.4.1. Topography strongly influences site productivity and fire severity patterns

A key consideration in development of restoration prescriptions is the topography of the landscapes in question. Topography strongly influences plant communities, site productivity, fire behavior, and fire severity patterns over large landscapes (Weatherspoon and Skinner, 1995; Jimerson and Jones, 2003; Lydersen and North, 2012; Hessburg et al., 2015). For example, a typical current pattern in California is for more severe fire effects to be manifested in mid- to upper-slope positions on south and west facing slopes, and less severe effects in lower slope positions, and on north- and east-facing slopes (Weatherspoon and Skinner, 1995; Taylor and Skinner, 1998; Skinner et al., 2006; Holden et al., 2009; North et al., 2009; Lydersen and North, 2012; Harris and Taylor, 2015). However, the strength of topographic effects varies by ecoregion, because of unique influences and interactions among geology, geomorphology, and prevailing wind and weather patterns (Habeck, 1976; Neilson, 1986, 1995; Pearson and Dawson, 2003; Collins and Skinner, 2014). Nonetheless, the effects of topography on severity patterns generally appear to be manifest in a gradient: the strongest effects are in steep, complex, rugged landscapes, and effects lessen as relief becomes gentler (Collins and Skinner, 2014).

2.4.2. Frequent burning of historical MSForests reduced the likelihood of severe fires

Many studies have shown that MSForests burned frequently, and these fires maintained conditions that were less likely to experience severe fire effects. Several recent studies have shown that the risk of high-severity fire can be substantially reduced following initial prescribed fire treatments (Baker, 1994; Stephens, 1998; Stephens et al., 2009; Vaillant et al., 2009; Fulé et al., 2012). However, this advantage is short-lived if not followed up within a few years with subsequent prescribed burns (Baker, 1994; Skinner, 2005; Schmidt et al., 2008). Understory trees and shrubs, killed in the first burn, soon after become surface fuel. If these areas are not reburned, fire hazard can again become high (Skinner, 2005). This is not the case for areas that have been mechanically thinned prior to burning; thinning removes the trees that would otherwise be killed by surface fire, especially where activity fuels are burned or otherwise treated (e.g., chipped) after thinning (Stephens et al., 2012b). Furthermore, the potential influence of topography on fire severity patterns can be reduced by decades of fire suppression and fuel accumulation (e.g., see Harris and Taylor, 2015). From a management perspective, this work indicates that in some locations: (1) numerous low- to mixed-severity burns would be needed to reduce the total amount and connectivity of surface fuels and thin forest canopies before wildfires could be used to regulate the successional mosaic; and (2) topography can be used as a guide to prioritize locations with a greater local risk of high-severity fire, and where fuels reduction would have a wider effect on potential fire severity across landscapes (Taylor and Skinner, 1998; Hessburg et al., 2015; Holden et al., 2009; North et al., 2009; Lydersen and North, 2012). Public sentiment concerning intentional addition of prescribed fire smoke makes this a formidable but worthwhile challenge (Engel, 2013).

2.5. Strategy 5: Rehabilitating plantations

2.5.1. Plantations may be a good source of future LOEST

In formerly clearcut forests (current plantations), and in selectively harvested areas that lack LOEST but have sufficient populations of well-adapted, early seral species, variable density thinning can accelerate the development of LOEST and restore patchiness at multiple scales (e.g., Churchill et al., 2013; Harrod et al., 1999; Knapp et al., 2012; Larson and Churchill, 2012; Ritchie, 2005). Retaining ponderosa and Jeffrey pines, western larch, Douglas-fir, and other early seral tree species such as incense cedar (*Calocedrus decurrens*), sugar pine, and western white pine, especially the larger and older cohorts, will help to restore diversity of early seral species, large tree structure, and resistance to severe wildfires. Subsequently these patches can be thinned from below and/or under-burned to further develop fire tolerance (Ritchie, 2005).

2.5.2. Plantation thinning can accelerate growth and development of fire resistance

While little may be done to reduce immediate fire hazard in young plantations (Stephens and Moghaddas, 2005; Weatherspoon and Skinner, 1995), thinning increases growth rates of remaining trees and accelerates the development of more fire-resistant boles and crowns, especially among the most fire-tolerant species (Ritchie, 2005). Reducing average tree diameter of younger stands (thinning from above) would be contrary to a goal of restoring a more fire- and drought-tolerant landscape.

Variable density thinning will not be the best approach to restoring LOEST on selectively harvested sites where early seral species have been eliminated. In these cases, regenerating new vigorous cohorts of early seral species will be necessary. Variable retention treatments (Franklin et al., 2007) can also be used, however, gaps must be sufficiently large to regenerate and establish dominance of the desired early seral species (Bigelow et al., 2011; York et al., 2004). Mixed- and high-severity fire, whether prescribed or managed wildfire, offer another approach to achieving openings large enough to regenerate early seral species. Whether mechanical or fire approaches are used, planting of the desired species will often be necessary to ensure successful establishment, as seed source is often limited in these situations. These treatments should be integrated with strategies that restore early-seral habitats (*Strategy 9*).

2.5.3. Plantation boundaries are often inconsistent with the topographic template

One significant impact of prior clearcut harvesting and plantation development is local and regional fragmentation of closed-canopy, late-successional and old forest conditions. Before plantations, patterns of ridges and valley-bottoms and north- and south-facing aspects were a natural physical template for patterns of structure and composition, tree size and age, density, and layering (Skinner et al., 2006; Hessburg et al., 2015; North et al., 2009; Stine et al., 2014). Closed-canopy and multi-layered tree conditions were characteristic of north aspects and valley bottoms, while south aspects and ridgetops supported more open canopies with grass, forb, and shrub understory conditions. Plantations and broad patterns of selection cutting had the effect of decoupling these patterns from their topographic template. Low thinning treatments described above can be applied beyond plantation margins to minimize hard edges and tailor more characteristic density, layering and composition conditions back to the topography.

2.5.4. Wildfire effects on current plantations broadly vary

The effects of wildfire on current plantations vary depending on: (1) methods of post-harvest fuels treatments and site preparation; (2) the species mix of understory vegetation; (3) plantation size; and (4) how well fuels have been managed in the surrounding forest (Weatherspoon and Skinner, 1995; Skinner and Weatherspoon, 1996). In the Klamath Mountains, for example, broadcast burning before tree planting gave plantations greater long-term protection than either pile burning or no fuels treatment (cf. Thompson et al., 2007). Broadcast burning removed slash and encouraged understory vegetation regrowth that was less likely to burn than that associated with piling and burning. Creating plantations in untreated fuelbeds offered the least protection against future severe fire behavior. When plantations were generally small (i.e., <20-ha), the method of vegetation and fuels management in the surrounding forest matrix was as important as that occurring within the plantations themselves (Weatherspoon and Skinner, 1995; Skinner and Weatherspoon, 1996). When plantations were >50-ha, as is typical where large, severely burned patches are replanted, the method of vegetation and fuels management within plantations was more important than that of the surrounding forest matrix (Skinner and Weatherspoon, 1996).

2.5.5. Plantation thinning and slash disposal should go hand-in-hand

Because thinning generates logging slash that can increase severe fire behavior (Agee and Skinner, 2005; Huff et al., 1995; Raymond and Peterson, 2005; Stephenson, 1998; Stephens et al., 2009), treating surface fuel accumulations is an essential part of reducing risk of severe fires. Pile and broadcast burning after thinning can reduce surface fuels to acceptable levels (Ritchie et al., 2007; Schmidt et al., 2008; Weatherspoon and Skinner, 1995). Whole tree harvesting has the advantage of leaving limited activity fuels behind (Stephens et al., 2009), but is not operationally feasible in many areas due to limited access (North et al., 2015) and low availability of biomass or wood chip markets. Moreover, the nutrient concentration of crowns is significantly higher than other above-ground tree components (Perry et al., 2008); thus their wholesale removal can degrade future site productivity.

Box 4 Role of hardwoods in MSForests.

Treating fuel ladders requires some caution in the mixed-conifer/broadleaf forests (Perry et al., 2011; Lake and Long, 2014). Because of their crown structure and foliage characteristics, mature hardwoods rarely act as fuel ladders, and in some cases may limit crown fire. Hence, cutting them can exacerbate fire risk, rather than reduce it. Moreover, hardwoods often function as important habitat (Flack, 1976; Zielinski, 2014) and rapid response soil stabilizers, and can provide Native American preferred foods and other subsistence resources (Anderson, 2005; Codding et al., in press; Lake and Long, 2014). A desirable level of understory broadleaf trees will depend on initial density, but excessive removal is unwarranted. The same caution applies to any forest type that contains relatively nonflammable deciduous hardwood species. In the case of relatively pure patches of aspen (*Populus tremuloides*), cottonwood (*Populus trichocarpa* & *Populus fremontii*), and birch (*Betula papyrifera* & *Betula occidentalis*), fire exclusion has dramatically reduced their abundance, patch sizes, and vigor (Hessburg et al., 1999a). For their influence on habitat for certain species, landscape biodiversity, and fire behavior (Kuhn et al., 2011; Shinneman et al., 2013), there are clear advantages to revitalizing existing clones and patches, or to restoring their abundance (Jones et al., 2005) near wet meadows and seeps, in areas of seasonally high water table, and in floodplain and riparian areas (Bartos and Campbell, 1998; Campbell and Bartos, 2001; Seager et al., 2013).

2.6. Strategy 6: Creating and maintaining successional heterogeneity

Successional pattern heterogeneity naturally derives from a characteristic pyrodiversity; it provides habitat for pre-forest, early-, mid-, late-successional and old forest associates, and influences the spread and intensification of disturbances and other processes (Keane et al., 2009; Perry, 1988; Raffa et al., 2008; Moritz et al., 2011; Perry et al., 2011). Restoring the patchy composition and structure that is a byproduct of a more characteristic pyrodiversity of MSFire landscapes requires re-creating or maintaining spatial heterogeneity at all appropriate scales (Franklin and Van Pelt, 2004; Franklin et al., 2002; Lydersen et al., 2013; Knapp et al., 2013; Skinner, 1995; Harrod et al., 1999; Hessburg et al., 2010, 2015; North et al., 2009; Perry et al., 2011)[3,4]. Simply summarizing the amount of area in the different successional classes to meet some desired proportions is not appropriate because that leaves out critical aspects related to patch sizes and configuration.

2.6.1. Spatial heterogeneity from MSFires is important at several scales

MSFires not only influence tree clump and gap sizes at relatively fine, within-patch scales, but also the patchiness of local and regional landscapes, by influencing landscape patchiness of physiognomic types, forest overstory and understory canopy cover, species composition, variability in patch size, tree age, density, and canopy layering. The subregional context of landscapes is also important. Large patches are often found in mesic environments with gentle to rolling topography, while smaller patches are found in highly-dissected terrains that exhibit a summer-dry, Mediterranean climate.

2.6.2. The importance of fine-scale heterogeneity

Within patch heterogeneity provides fine-scale habitat for pre-forest, early-, mid-, late-successional and old forest associates, and influences the flow of fine-scale processes, including fire (Allen et al., 2002; Binkley et al., 2007; Stephens et al., 2010). Larson and Churchill (2012) reviewed the literature on fine-scale, within-patch patterns and mechanisms of pattern formation for fire-frequent pine and mixed-conifer forests in western North America. They interpreted this information in the context of fine-scale pattern restoration and its importance to overall landscape restoration.

Next-generation fire models that are capable of very fine-scale (1–10 m) spatial representations of fuels, winds, and fire behavior reveal important effects of heterogeneous tree canopy patterns, canopy openings, and tree clumps on fire behavior (Parsons et al., 2011; Pimont et al., 2011). To date, studies at a patch scale indicate that within-patch tree spatial patterns influence complex feedbacks among fine fuels (understory vegetation and tree litter) and fire behavior, which in turn influence species composition, future vegetation growth, and fuel accumulation (Rebertus et al., 1989; Thaxton and Platt, 2006; Hiers et al., 2009; Mitchell et al., 2009).

[3] Historically, the stand concept would poorly define MSForests, which were better described as extensive forest mosaics distinguished by both local and regional pattern heterogeneity (Franklin and van Pelt 2004; Hessburg et al. 2015; Kaufmann et al., 2007). Contemporary patterns on public lands reveal primary influence by 5–20 ha stand-scale patterns; an artifact of intensive timber management and size-restricted treatment area.

[4] We do not suggest cutting LOEST to achieve patchiness, an action likely to increases future risk of severe fire (e.g., see Perry et al., 2004). Rather, spatial heterogeneity would be shaped by working in plantations or intermediate aged patches where shade-tolerant, late seral tree species have captured the site from early seral species.

2.6.3. Re-creating fine-scale heterogeneity

Irregularly spaced trees, large and small openings, and resulting variation in fuelbeds all limit the potential for crown fire initiation and spread, and reinforce similar post-fire vegetation patterns (Beaty and Taylor, 2007; Pimont et al., 2011; Stephens et al., 2008) by means of a fine-scale, naturally occurring version of strategically placed fuel treatments (Churchill et al., 2013, in press). Spatially varying within-patch structure and composition also hinders bark beetle mass attack by disrupting chemical signaling among prospective mates, and breaking up continuity of susceptible hosts, tree sizes, and ages (Fettig et al., 2007).

An increasing number of studies from western US MSForests are available to define historical tree clump and gap variability of various forest types (Churchill et al., 2013, 2014; Fry et al., 2014; Harrod et al., 1999; Hopkins et al., 2014; Kaufmann et al., 2007; Knapp et al., 2013; Larson et al., 2012; Larson and Churchill, 2012; Lydersen and North, 2012; Lydersen et al., 2013; Stephens et al., 2008; Taylor, 2010; Clyatt et al., 2016). Within patches, tree patterns are defined by uneven-aged and irregularly patchy mosaics of individual trees, tree clumps ranging from 2 to 20 or more trees, comparably sized tree gaps, and even larger openings. These mosaics persist for centuries in a highly energetic shifting system of tree clumps and gaps; gap-phase replacement is primarily driven by ongoing, patchy fire, insect, and disease mortality (Agee, 1993), and other stand dynamics processes. Patch sizes and within-patch heterogeneity vary over space and time, and by biophysical setting (Kaufmann et al., 2007). Occasionally moderate to high-severity disturbances or climate synchronization of reseeding and regeneration (North et al., 2005) reset these patch-level patterns (Arno et al., 1995; Hessburg et al., 2007). An example of persistent fine-scale dynamics in a MSForest landscape occurs in Yosemite National Park, in the central Sierra Nevada, where Scholl and Taylor (2010) found evidence for gap dynamics (<0.2-ha canopy openings) under the influence of a frequent fire regime, prior to onset of fire exclusion.

2.6.4. The importance of meso-scale heterogeneity

Numerous studies used interpretation of early, stereo, black and white aerial photography to better understand variation in landscape patch patterns and development of stand structure in relation to fire history (Habeck, 1976; Hessburg et al., 1999a, 1999b, 1999d, 2000a, 2007; Taylor and Skinner, 1998, 2003; Beaty and Taylor, 2001, 2008; Bekker and Taylor, 2001, 2010). Nearly all studies found a strong influence of topography on patch and stand structure patterns (discussed above).

2.6.5. Re-creating and protecting meso-scale heterogeneity

Like fine-scale patterns, meso-scale successional patterns within local (e.g., 10^4 to 10^5 ha) and regional landscapes (e.g., 10^5 to 10^6 ha) are a second foundation of resistant and resilient MSForests (Hessburg et al., 2005, 2007, 2015; Keane et al., 2009; Keane, 2012; McGarigal and Romme, 2012; Moritz et al., 2011, 2013; Perry et al., 2011; Stine et al., 2014; Wiens et al., 2012). Successional patterns arise from patterns of disturbances, environmental conditions, and other ecological processes (Habeck, 1976).

Varying patterns of physiognomic conditions (sparse woodland, pre-forest, forest, herbland, and shrubland) are also clearly apparent in most historical MSForests as a consequence of disturbance frequency and intensity, and climatic influences (Hessburg et al., 1999a, 1999b; Lenihan et al., 2003; Maxwell et al., 2014; Millar et al., 2007; Neilson, 1986, 1995). For example, historically it was common for some MSForest settings to remain for a time in alternate woodland, shrubland, or grassland states due to relatively high fire frequency, coupled with occasional high-severity fire (Beisner et al., 2003; Odion et al., 2010). But with a warming climate creating conditions for more high-severity fire (Westerling et al., 2003; Lenihan et al., 2006), and the occurrence of larger and more frequent high-severity burned areas (Miller et al., 2009, 2012; Harris and Taylor, 2015), there is now a greater potential for severely burned patches to be converted to these alternative stable states (Lauvaux et al., 2016; Long et al., 2014a; Harris and Taylor, 2015; Perry et al., 2011; Savage and Mast, 2005). Typically this occurs where successive fires occur over the same area, especially when the initial fire burns with high severity effects and causes a transition to a grass or shrub-dominated community (van Wagtendonk et al., 2012). Documented examples are widespread in the Siskiyou Mountains (Silver Fire of 1987 followed by the Biscuit Fire of 2003), the Klamath Mountains (King-Titus Fire of 1987 followed by the Panther Fire of 2008), the northern Sierra Nevada/southern Cascade Range (Storrie Fire of 2000 followed by the Chips Fire of 2012 – see Coppoletta et al. (in press)), and in the central Sierra Nevada (Stanislaus Complex of 1987 followed by the Rim Fire of 2013).

2.6.6. The importance of understanding the historical range of variability (HRV) in meso-scale successional patterns

Many authors have focused attention on better understanding historical variability in meso-scale successional patterns (hereafter, the HRV) and its central role in landscape restoration (e.g., Allen et al., 2002; Hessburg et al., 1999b, 1999d; Landres et al., 1999; Morgan et al., 1994; Swanson et al., 1994; Swetnam et al., 1999; Millar and Woolfenden, 1999; Keane et al., 2009; Moritz et al., 2013; Wiens et al., 2012). Most discussions point to a pivotal role of using HRV information to guide landscape restoration, *especially to understand how meso-scale successional patterns provided critical context for the variability of the local wildfire regime*. It has also become important for managers to use HRV information to learn how landscape local conditions have changed to the present day under the influence of local management regimes.

2.6.7. Historical fire regimes maintained forest cover and density at levels far below carrying capacity

Across many MSForest landscapes, historical fire regimes maintained overall forest density and biomass at levels far below carrying capacities (e.g., see Hessburg et al., 1999a, 2000a, 2005), thus providing a substantial "buffer" against a periodically warming climate. A surprising amount of area capable of producing forest cover was in fact in grass-, shrub-, or woodland conditions (see *Strategy 9*). One is left to wonder whether the natural resilience mechanism of the local forest fire regime was in large part driven by a large area of non-forest life forms interspersed among the forest patches, with their flashy surface fuelbeds, low energy release, short flame lengths, low fireline intensity, and rapid rate of spread when burned—a sort of benign to moderate fire delivery system.

Nearly all authors working with HRV estimates suggest that climatic and environmental changes and introductions of non-native species should temper to some degree the use of HRV information going forward. Instead of recreating a picture of the past, managers can use knowledge of past fire regimes and supporting successional conditions to build a more resilient landscape.

2.6.7.1. HRV spatial pattern conditions can be derived empirically or by simulation techniques. Both empirical and simulation approaches have been devised for predicting HRV spatial pattern conditions (Beukema et al., 2003; Hemstrom et al., 2004, 2007; Keane, 2012; Maxwell et al., 2014; McGarigal and Romme, 2012). Reviews by Cary et al. (2006) and Keane et al. (2004, 2006) highlight four dozen landscape succession and disturbance models from around the world. Keane et al. (2004) provide a key for selecting the most appropriate landscape succession model for management and research applications based on operational characteristics needed by users.

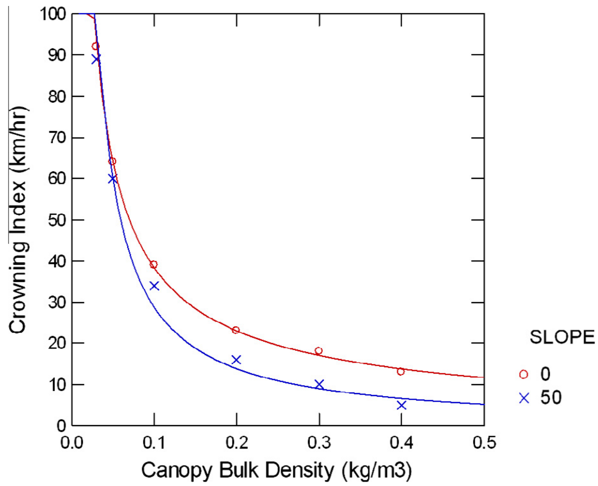

Fig. 8. NEXUS (Scott and Reinhardt, 2001) simulations of Crowning Index (wind speed required to initiate active crownfire) as affected by canopy bulk density and slope percent. Late summer fuel moisture conditions are shown. Canopy bulk densities represent a measured range on the Deschutes National Forest, central Oregon. Adapted from Perry et al. (2008).

Relatively few studies use empirically reconstructed HRV spatial pattern conditions. The available studies compare vegetation attributes from early and late 20th-century, stereo aerial photographs to determine the key changes in landscape patterns (Everett et al., 1994a, 1994b; Lehmkuhl et al., 1994). Huff et al. (1995) documented corresponding changes in fuel patterns, potential fire behavior, and smoke production. The work of Everett et al. (1994a, 1994b) was roughly tripled in a subsequent project, the 50 million ha Interior Columbia Basin Ecosystem Management Project (ICBEMP, Hann et al., 1997; Hessburg et al., 1999a, 1999c, 2000a; Raphael et al., 2001), which is home to extensive MSForests. Subsequently, Hessburg and others (Gärtner et al., 2008; Hessburg et al., 1999b, 1999d, 2004, 2013, 2014; Reynolds and Hessburg, 2005) developed operational decision support tools for estimating departures of contemporary landscape pattern conditions from both HRV and climate change analogue conditions for the mid-21st century, which they called the future range of variation (FRV, Hessburg et al., 2013, 2014, 2015). Spatially explicit prescriptions for restoring patterns of large landscapes are derived from these landscape evaluations. These decision support tools are in operational use today on three eastern Washington National Forests (Hessburg et al., 2013).

2.6.8. Landscape prescriptions are needed

Scientifically grounded landscape prescriptions are needed to create habitat and successional patterns at local and regional landscape scales that move landscapes toward conditions that confer climate and disturbance resilience, while creating functional, well-connected habitat networks for a broad array of native aquatic and terrestrial species (Hessburg et al., 2015). A landscape prescription should provide clearly articulated restoration objectives, target ranges for both total area (proportion of landscape) and patch size distributions of successional and habitat types, and specific guidance on how and where to adjust the spatial arrangement of patches (Hessburg et al., 2004, 2013; North et al., 2012; Perry et al., 2011). Local landscape prescriptions integrate and provide guidance on how to align different successional patches with the topographic template and how to protect and increase abundance of LOEST. In addition, terrestrial and aquatic habitat and road system restoration opportunities can be linked in local landscape prescriptions to take advantage of simultaneous problem-solving opportunities (Rieman et al., 2010). For example, local prescriptions can identify harmful road segments and fish passage barriers, opportunities to expand local fish strongholds and rebuild larger, more productive fish and wildlife habitat patches (*sensu* Rieman et al., 2000, 2010).

Box 5 On using historical reference conditions.

At least 4 caveats apply to using historical reference conditions as management guidelines:

- Mimicking historical conditions is not an end in itself, but is a means of accomplishing ecological objectives, and therefore appropriate only when it meets those objectives (Keane et al., 2009; Stephens et al., 2010; Wiens et al., 2012).
- The true value of historical information is in understanding how interacting fire and climate, and their variability through time and space, influenced ecological patterns of forest structure and successional conditions. This information can provide valuable direction for the complex process of ecological goal setting in management planning and implementation.
- Pervasive climate and land-use changes imply that past conditions may not fully reflect future climate–vegetation–disturbance–topography linkages (Hessburg et al., 2015; Millar, 2014; Moritz et al., 2013). Hence, one of the challenges may be deciding the degree to which past lessons are relevant to future management. Relevance will depend on goals, reasonable expectations of the future climate, and resources required to attaining those goals.
- Because regional landscapes are highly altered, restoration restricted to local landscapes is insufficient to address large-scale restoration needs (Hessburg et al., 2015; Stine et al., 2014).

2.6.9. Determining ranges of target tree densities

Risks from fire, drought, and insects are evaluated with metrics such as crown bulk density, crown base height, basal area, stand density index, and leaf area index. Developing strategies to achieve successional heterogeneity at multiple scales requires knowledge of how these measures relate to site conditions and ecological processes, including habitat values for wildlife communities.

Keane et al. (2005) evaluated relationships between crown bulk density and several measures of canopy gap fraction (≈1-canopy closure) across a range of sites in the western U.S. They found that for canopy gap fractions <0.3 (i.e., canopy cover >70%), crown bulk density ranged from 0.15 $kg\,m^{-3}$ to slightly <0.3 $kg\,m^{-3}$. For perspective, Fig. 8 shows NEXUS simulations of Crowning Index (critical wind speed to initiate active crown fire) as affected by crown bulk density. Crowning index decreases (i.e., vulnerability to crown fire increases) rapidly as crown bulk density increases to ~0.15 $kg\,m^{-3}$, and more slowly with further increases. Based on the findings of Keane et al. (2005) and the NEXUS simulations, crown bulk density values in stands with ⩾70% canopy closure will crown in winds of approximately 10–30 $km\,h^{-1}$, depending on slope steepness (cf. Moghaddas et al., 2010). In such cases, managers have the option to either accept a relatively high probability that closed-canopy forests are at risk of crown fire initiation and spread, or try to improve the odds by lowering the risk. There are two general situations that apply: (1) closed-canopy stands have few or no fuel ladders, in which case crowns are threatened by fire coming from adjacent stands (conditional crown fire); and (2) stands have sufficient fuel ladders (and/or surface fuels) to threaten

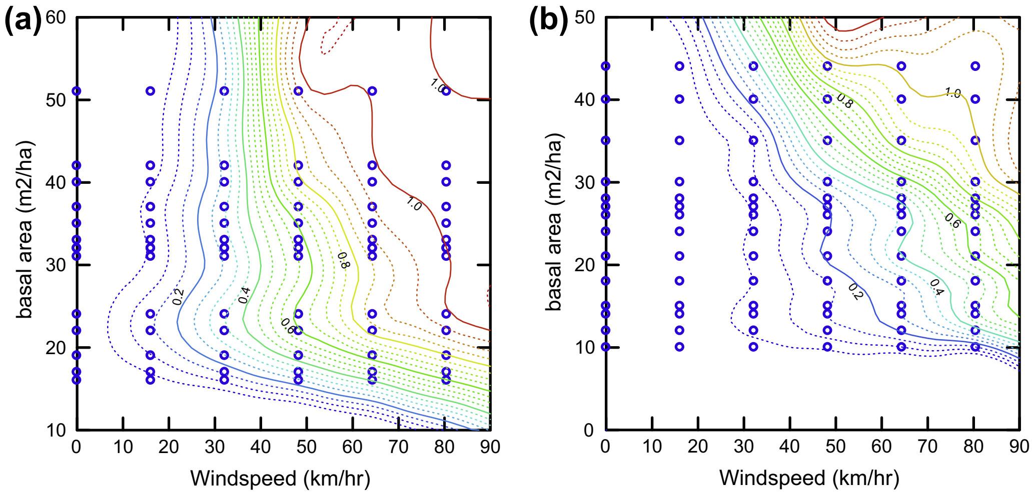

Fig. 9. Canopy fraction burned as a function of stand basal area (BA) and wind speed for two scenarios on the eastern slopes of the Oregon Cascades. This figure (a) is from 13 forest plots with measured crown bulk density (CBD), crown base height (CBH), and BA. This figure (b) is from the same plots but with all trees less than 20 cm DBH excluded from the CBD, CBH, and BA calculations (simulating a low thinning). Isolines represent crown fraction burned as calculated using NEXUS (Scott and Reinhardt, 2001), assuming late summer fuel moistures and using actual plot slope (which ranged from 0% to 13%). Circles represent simulated combinations of BA and wind speed. Data are from Perry et al. (2004). Notice in this example that canopy fraction burned at a given wind speed becomes strongly dependent on basal area. This will often but not always be the case. Exceptional examples will include true firs with relatively low CBH values, and stands with low BA but significant ladder fuels.

crowns by fire that moves upward from the ground within the stand, in which case risks come from both within and without. The former case is exemplified by closed-canopy forests with high levels of large conifer cover and low levels of understory shrub cover within the 2002 Biscuit Fire area in southwest Oregon and northern California, which had the lowest risk of crown fire (Thompson and Spies, 2009). The latter case occurs widely in current MSForests.

2.6.10. Determining ranges of target basal area

Managing for basal area ranges has the advantage of being a straightforward and common forestry metric, but basal area measures can be relatively difficult to crosswalk to other stand structural attributes influential to evaluating crownfire risk. For example, among structurally diverse stands throughout the western US, basal area was a much weaker predictor of crown bulk density than other measures discussed earlier, but among stands with similar structures, basal area correlated reasonably well with crown bulk density (Keane et al., 2005). In another study, 74% of the variation in crown bulk density among plots on the Deschutes National Forest was explained by basal area (Perry et al., 2004). Incorporating the number of *Abies* spp. stems per plot explained 88% of the variation, but including the number of pines had no effect. On those same plots, crown base height correlated poorly with basal area ($R^2 = 0.22$, $p = 0.088$); incorporating the numbers of *Abies* spp. or pine stems per plot did not improve the latter correlation. Clearly, variation in crown bulk density depends on species differences in crown architecture (Box 6).

In NEXUS simulations (Scott and Reinhardt, 2001) of the canopy fraction burned under different wind speeds, in the same plots as above (Perry et al., 2004), the interaction between wind speed and basal area was either highly significant ($p = 0.000$) or not depending on stand structure (Fig. 9). The models depicted in Fig. 9 show that a low thinning can significantly alter the canopy fraction burned and relations between wind speed and stand basal area. In the stands as measured in the field (Fig. 9a), wind speed was the primary driver of canopy fraction burned over a wide range of basal areas greater than 20 $m^2\ ha^{-1}$. With a simulated thinning of all stems smaller than 20 cm DBH (Fig. 9b), canopy fraction burned at a given wind speed becomes strongly dependent on basal area. The implication is that in stands with abundant small trees that can act as fuel ladders, basal area is a poor predictor of crownfire risk—the ladder fuels increase risk far out of proportion to their basal area. However, in stands with few or no fuel ladders, basal area provided sufficient information to predict an integrated risk of crownfire. This will not always be the case. Note that when prioritizing stands for treatment according to risk of crownfire, both topo-edaphic factors and the canopy fuels of the surrounding landscape can modify the effect of local stand structure. We return to the latter point in Section 2.9.2. Striking examples of topo-edaphic influence are in the Klamath Mountains and Southern Cascade Range of California where complex topography leads to inversion layers that trap smoke (Robock, 1988, 1991) and shade lower slope positions from solar heating, resulting in reduced fire severity in lower slope positions (Skinner et al., 2006; Taylor et al., 2013; Miller et al., 2012).

Box 6 Low thinning, leaving tree islands, and reducing crown base height and crown bulk density.

Thinning from below to reduce basal area can substantially increase crown base height because late-seral trees with larger live crown lengths will often be removed. Leaving patches of late-seral trees for ‘‘habitat islands” will retain some ladder fuels and areas of high crown bulk density that contribute to crown fire initiation and spread via individual tree or group torching. The likelihood of active crownfire might be lowered by buffering habitat islands with areas of relatively low crown bulk density and reduction of surface fuels, basically a microcosm of the landscape strategy discussed earlier. The degree to which such buffers might reduce the habitat value of the habitat islands or the likelihood of active crownfire is unknown.

On the Klamath Indian Reservation, Johnson et al. (2013) recommended allocating retained basal area into >50 and <50 cm DBH classes. In the dry pine and mixed-conifer forests within the Klamath, they recommended retaining 5–9 $m^2 ha^{-1}$ in trees <50 cm DBH, while trees >50 cm would be cut only in cases where a large late-successional tree had the potential to carry fire into the crown of a large early seral tree, and only then if the late-successional tree was younger than 150 years. Total retained basal area (all size classes) was ~22 $m^2 ha^{-1}$ in the dry mixed-conifer and ponderosa pine zones, and 27–32 $m^2 ha^{-1}$ in the moist mixed-conifer. These target basal area ranges were designed as a bet-hedging strategy where summer droughts are frequent, and where tree-killing bark beetles occur in relatively high endemic populations.

2.7. Strategy 7: Integrating restoration with late-successional forest habitat needs

Ideally, the goals of restoring pyro- and successional diversity of MSForests are integrated with other ecological objectives, some of which, by virtue of past management, may be negatively affected by ecosystem restoration activities. For example, forest densification, increased canopy cover and layering, and compositional shifts toward shade-tolerant tree species have benefited a number of species (e.g., northern goshawk (*Accipiter gentilis*), fisher, marten (*Martes caurina*), barred owl (*Strix varia*), CASPO (*Strix occidentalis occidentalis*), and the northern spotted owl (Brown et al., 2004; Gaines et al., 2010a, 2010b; Noss et al., 2006; Singleton, 2015; Spies et al., 2006, 2010; Zielinski, 2014; Keane, 2014). These novel habitats may be degraded by thinning and prescribed fire treatments that seek to create more open forests with LOEST.

2.7.1. Thinning effects on spotted owl prey species

Thinning effects can be variable depending on the species of interest, time since treatment, and landscape context (McIver et al., 2013). Gomez et al. (2005) found that northern flying squirrels (NFS, *Glaucomys sabrinus*), major prey of NSO and, in upper montane habitats, for the CASPO (Williams et al., 1992), were unaffected five years after thinning. However, in another study, Manning et al. (2012) found NFS were negatively impacted 15 years after thinning. A recent meta-analysis found negative impacts of forest management on NFS, but did not include any light thinning treatments (Holloway and Smith, 2011). These results suggest that there are definite landscape-level tradeoffs associated with creating more fire-resilient stand structures and habitat for species favoring denser forests. These tradeoffs must be weighed against possible habitat loss associated with extensive high-severity fires. Some of these risks can be ameliorated by leaving larger patches of untreated forest that are surrounded by thinned, open-canopy patches that can isolate the risk of crown fire (Ager et al., 2007).

2.7.2. Carefully considering trade-offs

Landscape-scale conservation of the structurally diverse, old forest habitats required by spotted owls and their prey requires careful consideration of the trade-offs between conserving existing habitat characteristics and promoting landscape conditions that are more resilient to the effects of large-scale, high intensity fire (Lehmkuhl et al., 2007). Another important consideration is the notion of accelerating "recruitment" of stands with more old forest characteristics (e.g., large trees), which are currently lacking in many landscapes. Gaines et al. (2010a, 2010b) evaluated the spatial overlap of modeled NSO suitable habitat with mapped priority fuel treatment areas in eastern Washington, to determine the magnitude and location of potential conflicts between fuels management and owl conservation. They found 34% overlap within dry mixed-conifer forests between high suitability NSO habitat and moderate-to-high priority fuels treatment areas, and also a high degree of overlap (35%) of low-suitability NSO habitat and moderate-high priority fuel treatment areas. They suggested that there was opportunity to accomplish fuel treatments and owl conservation by focusing treatments on dry mixed-conifer forest areas near areas of high quality NSO habitat. However, they did not address effects of barred owl (BDO, *Strix varia*) displacement of NSOs into drier mid-slope settings, which could make some drier habitats important for short-term NSO persistence (Singleton, 2013).

Roloff et al. (2005) modeled active and no-management effects on NSO in fire-prone landscapes in southwest Oregon. They found that management in owl foraging areas reduced habitat compared with no active management (only losses to wildfire). They attributed active management's lack of influence on fire behavior in part to the limited landscape area available to treat hazardous fuels, and to the fact that their treatments reduced owl habitat quality (from nesting to foraging), but did not reduce the risk of crown fire. Their model simulated vegetation dynamics (using FVS) and fire (using FlamMap).[5] In a second paper, Roloff et al. (2012) analyzed a different fuel management strategy for the same area. In that work they found that active management "was more favorable to spotted owl conservation…than no management." Using FlamMap, they assumed that if 50% of the owl territory had high crown fire potential, then all of the territory would be lost to a fire. In two other studies, Sovern et al. (2014, 2015) showed that NSO preferred to nest and roost in MSForests of eastern Washington State that included trees >50 cm and whose overstory canopy cover exceeded 70%. It is apparent that localized fire risk reduction and maintenance of NSO nesting and roosting habitats, as we currently understand and define them, are at cross purposes. However, it is likely that large-scale landscape fire risk reduction, when spatially optimized, can aid in maintaining NSO nesting, roosting, and foraging habitats in greater measure (Ager et al., 2007; Finney et al., 2007).

2.7.3. Competitive interactions between NSOs and BDOs

Another factor complicating NSO conservation in MSForests is competitive interaction with recently established BDO (Singleton, 2013; Sovern et al., 2014). BDOs appear to be displacing NSOs from suitable habitat, particularly in moist valley bottom settings that are preferred by BDOs (Singleton et al., 2010; Singleton, 2013; Yackulic et al., 2014; Wiens et al., 2014). BDOs are also increasing in number and distribution in the Sierra Nevada, and are an increasing risk factor for CASPOs (Keane, 2014). Competitive displacement between the two species is not fully understood, and proposed experimental removal studies will provide additional information (USFWS, 2013).

2.7.4. CASPO responses to landscape scale treatments

In areas of California where woodrats are important prey, NSO and CASPO foraging habitat is characterized as a heterogeneous mosaic of physiognomic types and successional stages interspersed with mature, closed-canopy forest (Zabel et al., 1992, 2003; Franklin et al., 2000; Tempel et al., 2014). Prey diversity and abundance is associated with heterogeneity in foraging areas (Roberts et al., 2011, 2015), while nesting stands are dominated by large, mature trees in a closed-canopy condition (Phillips et al., 2010). Tempel et al. (2015) reported that SPLAT

[5] Since the authors used FlamMap they likely were unable to account for topographic context – especially the effects of topography on inversion.

treatments have the potential to reduce fire risk to CASPOs under extreme fire weather conditions, but can have long-term negative effects on owls if fires do not occur. Under the wildfire-SPLAT treatment scenario, simulations showed that both owl habitat and demographic rates responded favorably to treatments for up to 30 years after wildfires. However, absent wildfires, treatments had persistent negative effects on habitat quality and demographic rates (Tempel et al., 2015). Similarly, Stephens et al. (2014) found a 43% reduction in the number of CASPO territories 2–3 yrs following implementation of a landscape fuels treatment strategy consisting of DFPZs and 0.2–0.8 ha patch-clearcuts in northern Sierra Nevada MSForests. This study was the first of its kind to monitor owl response to a landscape-scale treatment. While treatments reduced the risk of severe fire behavior, results suggest the overall strategy resulted in a reduced number of owl territories. Thus, consideration of the number and distribution of owl territories needed to sustain a viable population ought to be factored into MSForest restoration.

2.7.5. Other species responses to landscape scale treatments

Some wildlife species will likely benefit from restoration of MSForests, especially when LOESTs are retained or restored. For example, Gaines et al. (2010b) reported that thinning from below and prescribed burning were effective tools for restoring habitat structure for focal bird species (e.g., white-headed woodpecker), and other neotropical and migratory species showed either neutral or positive survival responses. In the same study, Lyons et al. (2008) reported that restorative treatments enhanced foraging habitat conditions for bark-gleaning birds. A key component of these restoration treatments was the retention of large trees and snags within treatment units. Finally, Lehmkuhl et al. (2013) suggested that management to restore resilience to disturbance in closed-canopy MSForests would likely increase forage for ungulates compared to a landscape impacted by fire exclusion and past grazing practices. Clearly, managers will need a way to evaluate how current MSForest landscapes have departed from historical successional pattern conditions to inform management of needed habitat diversity (Franklin et al., 2000; Gaines et al., 2010b; Tempel et al., 2014).

2.7.5.1. Treatment effects on Pacific fisher. Truex and Zielinski (2013) found that mechanical plus fire treatments had negative effects on predicted fisher resting habitat value and that late-, but not early-season prescribed fire also had a negative effect. A number of management activities were identified that could mitigate these effects. Garner (2013) and Zielinski et al. (2013) explored how tolerant fishers were to the combination of fuel treatments, commercial harvests, and prescribed burning in the southern Sierra Nevada. Both found that fishers would tolerate areas and frequencies of disturbance that were typical of current management in the mixed conifer zone. Garner (2013) reported that fisher home ranges tended to include larger proportions of treated than untreated areas, but when selecting microsites within their home ranges, fishers tended to avoid using sites within 200-m of a treated area. Zielinski et al. (2013) found fishers to consistently occupy areas where an average of 2.6% yr^{-1} of a home-range sized area had been treated or disturbed. This represented more treated area than is thought to be necessary to reduce fire spread rates in the southern Sierra Nevada (Syphard et al., 2011), but less than that needed to reduce spread rates in other geographic areas.

Box 7 Integrating owl (NSO and CASPO) conservation and fire regime restoration.

(1) The NSO nests, roosts, and forages in closed-canopy, multi-layered, medium- to large-diameter late-successional and old forest patches of natural or anthropogenic origin (Everett et al., 1997; Forsman et al., 1984; Forsman et al., 2011; Tempel et al., 2014, cf. Lee and Bond, 2015).
(2) In California mixed-conifer forests and woodlands, both NSO and CASPO nest and roost in structurally similar forest patches, but forage in landscapes with a heterogeneous mix of vegetation types and successional stages interspersed with the mature, closed-canopy forest (Zabel et al., 1992; Franklin et al., 2000; Roberts et al., 2015).
(3) Historically, the dry and mesic MSForest conditions that supported NSO, CASPO, and several of their prey species were less common and found in valley-bottom and northerly aspect settings, rather than in southerly aspects and ridgetops.
(4) Today, competitive interactions with recently established BDO populations are impacting NSO populations. BDOs are apparently displacing NSO from valley bottom habitats they prefer (Singleton, 2013; Yackulic et al., 2014).
(5) Other prey species like the bushy-tailed woodrat occupy early- or mid-seral patches within MSForest landscapes because preferred mast species are more common there.
(6) The goals of owl conservation and fire regime restoration involve tradeoffs that must be addressed at landscape scales, and likely depend on a careful strategy to increase within-patch and landscape successional pattern heterogeneity.
(7) Absent BDO displacement effects, such a strategy could entail conserving habitats for NSO and CASPO, and their prey species, in north aspects and valley bottom settings, and focusing thinning, fuel treatments and restoration of fire on drier environments, such as ridgetops and south-facing aspects. Given ongoing competitive pressures from interactions with BDOs, recent observed decline in the NSO populations (Forsman et al., 2011), and absent BDO removal, conservation of NSO habitats around occupied NSO sites may be important for short-term NSO population conservation in some places.
(8) It will be important to consider the number of owl territories that can likely be sustained under various restored forest conditions. Spatially-explicit owl population models can be used to estimate the number of owl pairs needed for highly probable persistence. Alternative restoration scenarios and their associated numbers of maintained owl pairs could be monitored under an adaptive management framework.

2.8. Strategy 8: Mitigating threats from drought, forest insects, and pathogens

When we re-create fine-scale spatial heterogeneity within patches, a more characteristic and truly dynamic complexity begins to emerge. This complexity is maintained by endemic insect and pathogen populations and fine-scale heterogeneity in fire behavior (Weaver, 1943). Indeed, resilient forests express a modest level of vulnerability to native insects, diseases, and fires. However, more grave concerns about recent severe droughts and bark beetle outbreaks in eastern Oregon, Washington, and California highlight the need for site-specific patch- and landscape-level management practices, including stocking level control.

2.8.1. Thinning can be useful in a variety of situations

A number of insect and disease concerns can be addressed by altering species composition, but cannot be prevented by density management alone (e.g., spruce beetle (*Dendroctonus rufipennis* Kirby), laminated root rot (*Phellinus weirii* (Murr) Gilbertson), Douglas-fir tussock moth (*Orgyia pseudotsugata* McDunnough), and western spruce budworm (*Choristoneura freemani* Freeman) (Hessburg et al., 1994). With others, thinning forests of host species well before an outbreak can often be a useful means of lowering the likelihood of mortality associated with mountain pine beetle (*Dendroctonus ponderosae* Hopkins), Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopkins), and western pine beetle (*Dendroctonus brevicomis* LeConte) infestations (Fettig et al., 2007; Mitchell et al., 1983). Thinning can also reduce the severity of some dwarf mistletoe (*Arceuthobium* spp.) infestations, especially in even-aged stands on relatively productive growing sites (e.g., see Barrett and Roth, 1985). Thinning is especially useful where the residual trees are well-adapted to the site, display high vigor, and where the top half of tree live crowns is generally free of mistletoe.

Thinning can be useful where widespread MSForest stagnation or growth suppression related tree mortality is likely. Native forest insects and pathogens are often the vehicle for such mortality. In managed forests, thinning combined with under-burning can take the place of frequent surface fires to reduce surface fuelbeds and stocking from below, while favoring larger diameter leave trees and fire-tolerant species compositions. Thinning typically increases the growth and vigor of remaining trees (Collins et al., 2014, McDowell et al., 2003, Ritchie et al., 2008, Hurteau and North, 2010) and also may be used to accelerate development of old-forest characteristics.

2.8.2. Foresters need a broader variety of stand density management tools

To match thinning needs to specific stands and environments, foresters need stocking level curves and other density management tools (e.g., see Cochran et al., 1994; Long and Shaw, 2005; Powell, 1999; Shaw, 2000), and these measures need to be cross-walked to wildlife, fuel, fire behavior, insect and disease hazard measures, and other resource and ecosystem service metrics such that multiple objectives can be planned, monitored, and realized.

Stand measures such as BA and mean DBH may be adequate when used to describe even-aged stands, but are inadequate for multi-aged stands. This is important for managing bark beetle susceptibility of future patches and forests, which will display clumped and gapped tree arrangements. Foresters need measures that accurately describe density and its relationship to the distribution of diameter classes. The stand density index (SDI, Reineke, 1933), as adjusted by Zeide (1983) and Shaw (2000), is capable of meeting this need for irregularly structured, multi-cohort stands. However, additional field research is needed to develop broad understanding of stand density thresholds for a wide variety of site conditions and species throughout interior Oregon, Washington, and California. This research would identify lower and upper limits of full site occupancy for even- and uneven-aged stands under both current and future climate conditions (e.g., see Powell, 1999).

2.8.3. Adapting stand density to future climatic changes

Projected changes in regional climate and related shifts in disturbance behavior present additional challenges to managing future stocking levels. Larson and Churchill (2012) show how tree patterns in pre-settlement era stands can be used to establish fine-scale reference conditions for restoring patterns of variably-sized tree clumps, gaps, and openings. Churchill et al. (2013) go on to introduce a method for taking these fine-scale tree patterns and adapting them to anticipated future climatic changes. They use regionally downscaled estimates of temperature, precipitation, and soil water-holding capacity to calculate annual actual evapotranspiration (AET) and annual climatic water deficit (Deficit), the difference between potential evapotranspiration (PET) and AET. AET and Deficit have been shown to be good predictors of species presence/absence and growth rates, forest structure, and fire effects (Littell et al., 2008, 2009; Lutz et al., 2010; Kane et al., 2015). Use of these reference conditions provides a way of adapting information from pre-settlement era conditions by factoring in future climate change projections. Methods like these will be especially relevant for adjusting species compositions and density/carrying capacity relations, which native bark beetles will be highly sensitive to. Similar studies are needed for a broad range of plant associations throughout the Inland Western US.

2.9. Strategy 9: Creating and maintaining early successional forests

All seral stages are vital to maintaining landscape patterns that support a wide variety of species and functional diversity. However, in recent decades, early successional conditions were noticeably undervalued (Swanson et al., 2010); perhaps because many suspected that with all of the 20th-century logging, they might never be in short supply. But they are in short supply in some areas, and current configurations of these conditions where they do occur bear little resemblance to historical conditions. For example, we now have large concentrated areas of early successional habitat created by intense fire, as well as large expanses that are completely void of it. What is often lacking is the fine- to meso-grain mosaic of early successional conditions dispersed across large landscapes.

Naturally recovering, structurally and compositionally diverse early successional ecosystems are biologically and functionally rich components of landscapes (Fontaine et al., 2009; Hutto, 1995; Kotliar et al., 2002; Smucker et al., 2005; Swanson et al., 2010), and they provide resources for many associated food webs (e.g., see Lehmkuhl, 2004; Lehmkuhl et al., 2006a, 2006b). Especially after severe fires, early successional forests are characterized by structural legacies (snags, logs, and remnant, mature live trees) with accompanying grassland, shrubland, woodland, or herbland dominance (Habeck, 1976; Hutto, 1995; Kotliar et al., 2002). In frequently reburned MSForests, many of the legacies created by fire are short-lived, which emphasizes the need for recurring fire at appropriate ranges of severity to continually recruit these features. In instances where MSForests have experienced recent uncharacteristic stand-replacing patch sizes, recruitment of these features will be considerably prolonged (see below).

As befitting their complexity, MSForests consist of a dynamic mosaic of structural conditions that allow light-demanding native shrubs, forbs, and grasses to survive and persist at individual patch levels for decades to centuries, and for many centuries at landscape scales, depending upon fire frequency and succession processes. Early successional vegetation can exist as relatively large patches, which may be susceptible (for a period of decades to centuries) to high-intensity reburns (Nagel and Taylor, 2005; Skinner and Taylor, 2006; Odion et al., 2010; Thompson and Spies, 2010; Coppoletta et al., in press), or more commonly as small- to mid-sized patches, depending on fire regimes, climate, soils, and topography (Franklin and van Pelt, 2004; Lutz et al., 2011). Sources of early successional patches in current landscapes are wildfires (managed and unmanaged) and variable-density thinned areas that contain large openings that are prescribed-burned relatively frequently (e.g., see Skinner et al., 2006; Weatherspoon and Skinner, 1995).

2.9.1. Broad historical extent of early successional conditions

Early work by Habeck (1976) in the Selway Bitterroot Wilderness of central Idaho called attention to the rapid loss of successional diversity and early successional pre-forest habitats after 50 years of fire suppression (see Fig. 4 in Habeck, 1976). Summaries from the Interior Columbia Basin meso-scale assessment (Hessburg et al., 1999a, 2000a) are another source of detailed data on the historically broad extent and variability of area in early successional herb (grass), shrub, sparse woodland, bare ground, and stand initiation (newly regenerated forest, O'Hara et al., 1996) patches. In that assessment, they identified area capable of supporting forest cover in early 20th-century historical photography by observing the same area in the late-20th-century aerial photography that supported at least 10% canopy cover of trees. After ⩾70 years of fire exclusion, trees had re-invaded many areas formerly occupied by fire-maintained grassland, shrubland, or woodland (Hessburg et al., 1999a, 2000a, 2000b). We provide an example of the areal extent and patch size distributions of early seral conditions using data from the 1.5 million ha North Cascades province (Table 1 and Fig. 10A and B).

When considering all forested PVTs,[6] ~81% of the province was capable of forest growth, and ~42% of the forest-capable area was in pre-forest or early seral conditions (Table 1). The presence of such a large area of early seral and flashy fuel conditions would have conveyed fire with relative ease throughout the landscape; however, owing to high fire frequency, the conveyed fires likely spread rapidly, and exhibited relatively short flame lengths and low fireline intensity. Patch size distributions of the early successional conditions depicted in Fig. 10B reflect an approximately natural log distribution, with few patches larger than 1000 ha and most patches ranging between several ha and 200–500 ha. Early successional conditions included a broad distribution of patch sizes, but note that the very large patches that are typical after contemporary wildfires are absent from these distributions.

2.9.2. Concerns with overabundant early successional forest conditions

The abundance, connectivity, and grain of early successional forest patches on the landscape have functional implications that go well beyond habitat alone. Depending on species composition and environment, early successional forest patches may initially resist reburning at high-severity, but after a decade or more of tree growth can become highly susceptible to fire, depending on site productivity and level of tree establishment (Andrews and Cowlin, 1940; Moritz et al., 2011; Thompson and Spies, 2010). In fact, modeling by Kitzberger et al. (2012) showed that when older forests were intermingled with these younger, more flammable forests, landscapes can become unstable. Thus, complex tradeoffs may exist between the amount and the spatial configuration of young forests and the degree of landscape-level fire resistance.

[6] Environments that are similar in their climate, landforms, and geomorphic processes display a similar distribution of vegetation in the absence of disturbance (Arno et al., 1985; Steele and Geier-Hayes, 1989).We term this unique vegetation class the potential vegetation type or PVT.

Box 8 On the question of postfire salvage.

Because we can influence but not control the rate at which wildfires and other disturbances create early successional patches, the issue of creating and maintaining a diversity of early successional patches will always have an adaptive component that reflects the occurrence of uncontrolled disturbances over both space and time. Where management objectives after MSFires are to prefer restoration via natural recovery processes, the obvious approach is to forego salvage and planting operations. To be effective though, such areas should retain the potential for the full suite of natural recovery processes, including natural reseeding by coniferous and other understory species that are naturally adapted to the site and fire regime.

Where this is not true (as in cases of high-severity reburn potential, very large and homogeneous high-severity burn patch sizes, and where desirable seed sources are well beyond probable dispersal distances), salvage and/or planting operations may be reasonable options, and could be planned and conducted so as to reduce disruption of early successional diversity (Lindenmayer et al., 2004; Noss et al., 2006; Long et al., 2014a). Where they are appropriate, salvage operations should focus on the primary fuels that are the reburn concern, i.e., the smaller understory shade-tolerant trees that comprised the ingrowth over the period of fire exclusion (Peterson et al., 2015). Salvaging large trees provides a large economic benefit but has no known ecological benefit, and significant ecological costs (Donato et al., 2013; Lindenmayer et al., 2004; Noss et al., 2006; Long et al., 2014a).

One recent study that focused on retaining the basal area of the largest trees was implemented in the severely burned 2003 Cone Fire area (Ritchie et al., 2007). In that study, the proportion of basal area retained was evaluated at 5 levels: 100%, 75%, 50%, 25%, and 0%, with three treatment replications at each level in a fully randomized assignment (Ritchie and Knapp, 2014). Regardless of retention level, ~80% of the retained standing bole biomass had transitioned to surface fuel by the eighth year of the study (Ritchie et al., 2013). After 10-yr, only 25% of the largest pines were standing, but 86% of the largest white fir and virtually all of the incense cedar remained standing (Ritchie and Knapp, 2014). Natural regeneration was scant and found mostly in units nearer the edge of the burn, and otherwise did not differ by intensity of treatment. Survival and growth of planted trees did not differ by intensity of salvage (Ritchie and Knapp, 2014, cf. Donato et al., 2006). These results suggest that large pines may provide a fairly short-lived snag resource. More research is sorely needed on snag longevity by tree species, size, and geographic area, and on snag abundance and arrangement requirements of wildlife species at patch to landscape scales.

3. Towards a comprehensive landscape strategy

We have proposed that a landscape management strategy for MSForests needs to simultaneously address a number of subcomponents. Developing such a strategy requires addressing both legacy management issues and ecosystem pattern and process

Table 1
Percentage of potential vegetation type (PVT) area in early successional (seral) herbland, shrubland, woodland, bare ground and forest stand initiation patches in early 20th-century dry ponderosa pine, and dry and moist mixed conifer mixed severity forest environments of the Northern Cascades province of the Interior Columbia River Basin (adapted from Hessburg et al., 1999a, 1999b, 1999c, 1999d, 2000a). [Note that the historical photography predates all but trace amounts of regeneration harvesting]. Woodland in this usage is defined as ⩽30% tree canopy cover (sparse coniferous or hardwood) growing on a forest PVT[a]; i.e., capable of producing forest cover (⩾10% tree canopy cover) in the late 20th century aerial photography. PVT labels are PIPO = ponderosa pine; WD PSME/ABGR = warm-dry Douglas-fir/grand fir; CM PSME/ABGR = cool-moist Douglas-fir/grand fir; WD ABLA2/PIEN = warm-dry subalpine fir/Engelmann spruce; and CM ABLA2/PIEN = cool-moist subalpine fir/Engelmann spruce (see Hessburg et al., 1999a, 2000a) for expanded definitions and typing methods.

Province	Potential vegetation type (PVT)								% of Province
	Early seral forest conditions					All other forest cond.	All other range cond.	Nonforest/non-range (i.e., human developments/croplands)	
Northern Cascades	PIPO	WD PSME/ ABGR	CM PSME/ ABGR	WD ABLA2/ PIEN	CM ABLA2/ PIEN	All other forest PVTs	All other range PVTs		
Herbland	6.01	1.56	1.73	0.52	1.18	0.63	45.61	3.41	7.58
Shrubland	0.67	0.89	0.42	6.13	1.27	1.22	29.56	0.09	5.03
Woodland	64.48	21.34	9.71	30.19	15.70	16.21	0.14	0.59	19.79
Bare ground	3.02	1.39	1.16	1.54	4.20	3.77	24.63	95.47	16.90
Stand initiation	9.23	8.14	8.82	6.38	14.63	7.00	0.00	0.05	6.78
Other forest structures	16.59	66.69	78.16	55.25	63.02	71.16	0.06	0.39	43.92
% PVT area	100	100	100	100	100	100	100	100	–
% of Province	1.83	10.16	24.12	2.97	14.92	26.73	13.39	5.87	100

[a] Environments that are similar in their climate, landforms, and geomorphic processes display a similar distribution of vegetation in the absence of disturbance (Arno et al., 1985; Steele and Geier-Hayes, 1989). We term this unique vegetation class the potential vegetation type or PVT.

needs going forward. In that light, strategies generally should specify:

(1) Where the landscape(s) in question fit on the MSFire gradient, as a general historical feature, from low- to high-severity dominated fire behavior. This will help guide the response to each of the following points.
(2) Areas of the landscape that would be untreated or lightly treated to protect key habitats or strongholds for listed species that favor dense forests, and other specific resource concerns.
(3) Areas where connectivity of habitat area is considered, but weighed against the potential to vector intense fire. Riparian areas may contribute to this connectivity, but upland connections need also be considered.
(4) The most effective locations for treatments in the remainder of the landscape, given knowledge of disturbance processes and their drivers. These treatments serve to both protect key features from uncharacteristically intense fire, and allow for greater fire use to achieve restoration goals (see 9 below).
(5) The intensity, frequency, and spatial distribution of treatments needed to create desired spatial pattern and disturbance regime conditions (e.g., see Hessburg et al., 2013, 2015).
(6) Explicit landscape prescriptions for achieving the spatial pattern rearrangement, recognizing that restoring patterns and the extent of LOEST will take several centuries.[7]
(7) Testing out the landscape prescriptions/scenarios using landscape models that project future conditions and allow users to see how well management actions over time and space achieve multiple goals (Spies et al., 2014).
(8) A portfolio of silvicultural prescriptions and a diverse toolkit for achieving the multivariate objectives.
(9) Wildfire management strategies that include the broad use of managed wildfire and prescribed burning to accomplish restoration objectives (Habeck, 1976).

[7] The reality of ongoing disturbance events will require adaptively managing these landscape prescriptions for the foreseeable future.

4. Overarching concepts

The recommendations we offer in foregoing sections are grounded in eight overarching concepts:

(1) *Avoid one-size-fits all approaches.* The mixed-severity fire bin is quite large, ecologically diverse, and not very useful as a category to guide management and policy. Management for a given set of ecological objectives should reflect the uniqueness of place, including what is known about historical patterns, what is predicted for future climates, and the stressors that exist or can be expected in the future.
(2) *Refine the mixed-severity fire bin.* We said in the Introduction that the mixed-severity class is a catch-all bin for fires that are neither low nor high severity. Because many fires are of a mixed-severity and much variety in successional conditions results from them, it may make sense to define subclasses within the mixed-severity class (e.g., 20–33%, 34–50%, 51–66%, >66% of the dominant tree basal area or canopy cover killed) to improve its utility for assigning successional outcomes associated with MSFires.
(3) *Don't be a prisoner of history.* Where the goal is to produce more resistant and resilient forests regionally and locally, while protecting or restoring critical habitats, historical landscape patterns may have to be adapted. Historical patterns provide valuable insight, and in some cases offer the best route to achieving desirable ecological goals. In other cases, however, the highly altered regional landscape of today may require unprecedented mitigations to conserve native species, adapt to climatic and non-native species changes, or restore fire regimes (Millar et al., 2007; Stephens et al., 2010; Scheller et al., 2011).
(4) *Buy time for climate adaptation and sensitive species.* Much of the MSForest landscape is highly altered and susceptible to intense fires, seasonal and longer droughts, and large-scale, protracted insect outbreaks. Where possible, strategically place burn treatments to break up the homogeneity of the broad regional landscape (Finney, 2001; Agee and Skinner, 2005; Moghaddas et al., 2010), to buy time for creating desirable conditions on the larger landscape, and enable managers to cover more area with initial burn treatments (Weatherspoon and Skinner, 1996; Agee et al., 2000). Where

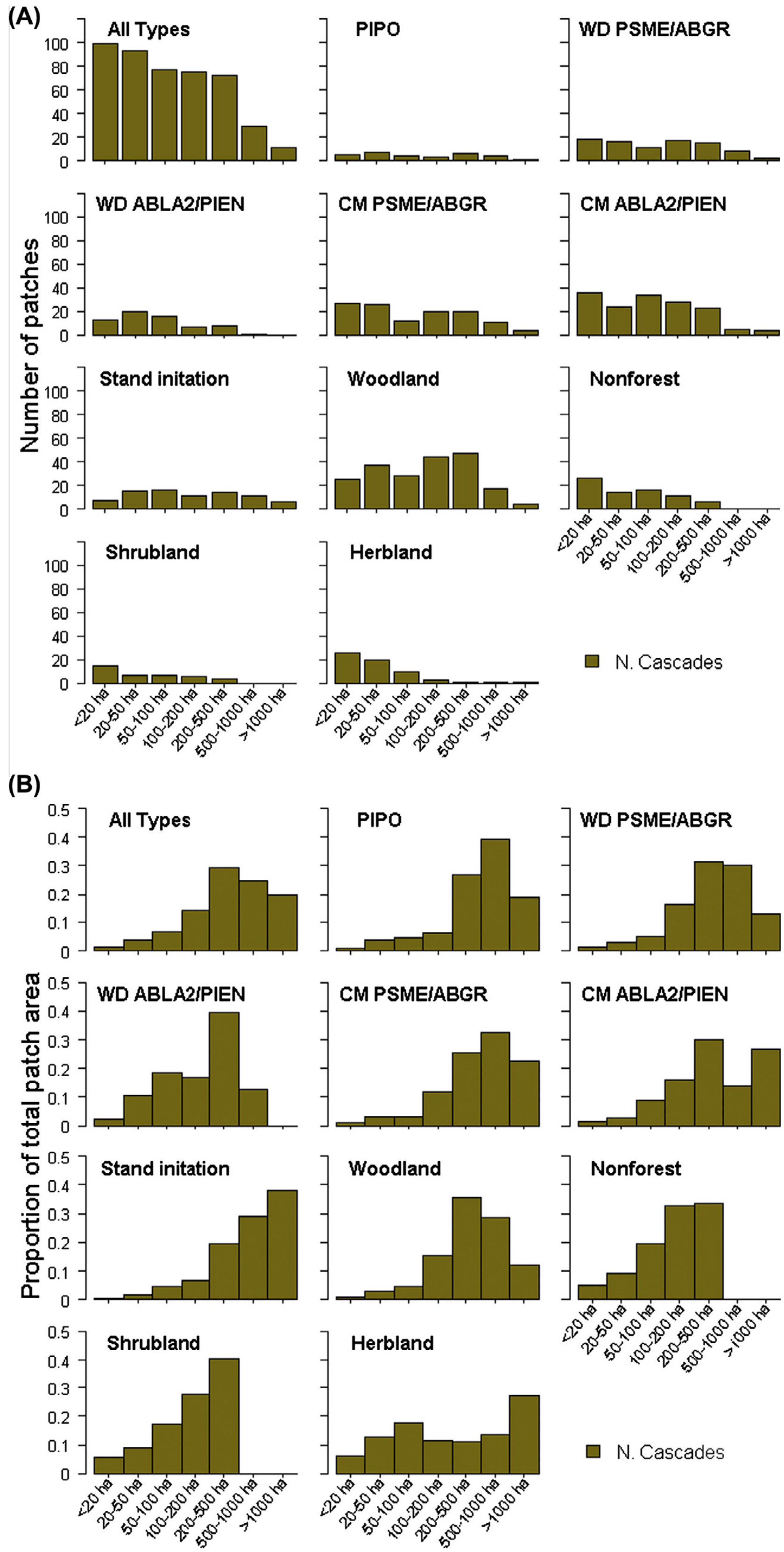

Fig. 10. (A) Patch size distributions of early 20th-century early successional conditions depicted in Table 1. Distributions from top left are: All types = pooled herbland, shrubland, woodland, bare ground (nonforest), and forest stand initiation structure patches of all forest PVT settings; PIPO, PSME/ABGR, CM PSME/ABGR, WD ABLA2/PIEN, CM PSME/ABGR = pooled herbland, shrubland, woodland, bare ground, and forest stand initiation structure patches of ponderosa pine, warm-dry Douglas-fir/grand fir (dry mixed conifer) cool-moist Douglas-fir/grand fir (moist mixed conifer), warm-dry subalpine fir/Engelmann spruce, and cool-moist subalpine fir/Engelmann spruce PVT settings, respectively; Stand initiation, Woodland, Non-forest, Shrubland, Herbland = pooled stand initiation, woodland, bare-ground, shrubland, and herbland patches of all forest PVT settings, respectively. Methods for modeling and assigning forest PVT settings are provided in Hessburg et al. (2000b). (B) Proportion of the total patch area in each early successional condition represented by patches of each size class.

appropriate (Naficy et al., 2016), strategically placed fire treatments could reduce fire severity in large, remote, or administratively withdrawn wild areas, as well (Schmidt et al., 2008). Wherever strategically placed treatments are implemented, they should be planned to accommodate the connectivity of dense, old forest habitats for the subset of key species associated with this habitat. Connecting valley bottom and north aspect topographies will be helpful in this light.

(5) *Maintain functional habitat networks for early-, mid-, late-successional and old forest specialists.* Where habitat needs are insufficient, actively develop replacement habitat or facilitate their development via natural processes.

(6) *Use the best practices available* to protect sensitive soils, streams, native fishes, and riparian corridors, listed terrestrial and other aquatic species and their habitats, and remaining LOESTs and old forests. With exception for hyporheic and floodplain environments, riparian zones in MSForests also experienced fire at similar frequencies to their adjacent upslope areas (Camp et al., 1997; Van de Water and North, 2010). Management can be designed to enable typical frequency and severity of this fundamental process (Beche et al., 2005).

(7) *Consider using managed wildfire wherever practicable.* Increasingly, natural ignitions can be used to increase spatial heterogeneity in forests with MSFire regimes (Rollins et al., 2001; Collins et al., 2009; Collins and Stephens, 2010). This is true in managed and wilderness forests, especially during periods of relatively benign fire weather. Because most remote areas are in congressionally withdrawn wilderness, National Park, or RARE 2 designation, it makes sense to allow naturally ignited fires to burn in these areas under carefully monitored conditions.[8]

(8) *Make significant progress with adaptive management.* Because climate change and wildfire uncertainties are large, research knowledge is always limiting, and surprises will occur. Designing new large and small scale experimental treatments has the potential to provide rich insights into future sources of landscape resistance and resilience. Historical references have been invaluable to providing insights about prior landscape processes and their interactions with forest conditions. But time marches forward, and ecosystem history is non-repeating. Much like the work of Churchill et al. (2013), our knowledge of historical conditions can be mindfully reshaped by our knowledge of how the future climate and land-use will create the MSForests of the future. We can either watch it happen, or we can accelerate learning (Bormann et al., 2007).

None of the concepts above preclude a commercial timber yield; however, they do assume ecological restoration as the central focus, with wood fiber yield as a by-product to support local communities, maintain restoration infrastructure capacity, and to subsidize some costs of restoration. Progress toward integrating ecological and human needs should be possible where collaboration builds trusted relationships and transparently shared goals, where efforts emphasize both social and ecological values, and where restoration emphasizes large landscapes that are resilient to disturbances and climate change (Costanza, 1991; Bengston, 1994; Hanna and Munasinghe, 1995; Long et al., 2014b).

[8] Smoke management will always be a concern, however, one of the more active areas of managed wildfire use in the western US is the Illilouette Creek Basin, 7 km south of Yosemite Valley (which receives millions of visitors each year). Portions of fires in this area are suppressed because of smoke management concerns but the entire basin of over 15,000 ha has burned at least once since 1974 with many areas burning multiple times. Similar management actions could occur in large remote areas throughout the western US.

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