

From: [Karl Halupka](#)
To: [Betty Grizzle](#)
Cc: [Gregg Kurz](#)
Subject: Climate effects on wildfire in Washington
Date: Tuesday, August 1, 2017 4:40:03 PM
Attachments: [Cansler and McKenzie 2014 - climate and fire in the North Cascades.pdf](#)
[Westerling 2016 - fire and timing of spring.pdf](#)
[Wimberly and Liu ForEcolMgmt 2014.pdf](#)
[Hessburg et al 2016 Mgmt of mixed severity fire regime forests in OR WA and NoCa.pdf](#)

Hi Betty,

Gregg asked me to pass along any recent literature I was aware of that was specific to Washington and dealt with issues surrounding climate change and wildfire.

I hope the attached references do the trick. Most of these are about the Pacific Northwest rather than just Washington. If I come across anything else that's on point, I'll pass it along.

Cheers,

k

Karl Halupka
Fish and Wildlife Biologist
U.S. Fish and Wildlife Service
Central Washington Field Office
215 Melody Lane, Suite 103
Wenatchee, WA 98801-8122
Phone: 509-665-3508 x 2001
Fax: 509-665-3509
www.fws.gov/wafwo/

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Author(s): C. Alina Cansler and Donald McKenzie

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Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA

C. ALINA CANSLER^{1,3} AND DONALD MCKENZIE²

¹*School of Environmental and Forest Sciences, University of Washington, Seattle, Washington 98195-2100 USA*

²*Pacific Wildland Fire Sciences Lab, USDA Forest Service, Seattle, Washington 98103 USA*

Abstract. Warmer and drier climate over the past few decades has brought larger fire sizes and increased annual area burned in forested ecosystems of western North America, and continued increases in annual area burned are expected due to climate change. As warming continues, fires may also increase in severity and produce larger contiguous patches of severely burned areas. We used remotely sensed burn-severity data from 125 fires in the northern Cascade Range of Washington, USA, to explore relationships between fire size, severity, and the spatial pattern of severity. We examined relationships between climate and the annual area burned and the size of wildfires over a 25-year period. We tested the hypothesis that increased fire size is commensurate with increased burn severity and increased spatial aggregation of severely burned areas. We also asked how local ecological controls might modulate these relationships by comparing results over the whole study area (the northern Cascade Range) to those from four ecological subsections within it. We found significant positive relationships between climate and fire size, and between fire size and the proportion of high severity and spatial-pattern metrics that quantify the spatial aggregation of high-severity areas within fires, but the strength and significance of these relationships varied among the four subsections. In areas with more contiguous subalpine forests and less complex topography, the proportion and spatial aggregation of severely burned areas were more strongly correlated with fire size. If fire sizes increase in a warming climate, changes in the extent, severity, and spatial pattern of fire regimes are likely to be more pronounced in higher-severity fire regimes with less complex topography and more continuous fuels.

Key words: bottom-up controls; burn severities; climate change; fire regimes; FRAGSTATS; North Cascades, Washington, USA; patch dynamics; relative differenced normalized burn ratio; RdNBR; remote sensing; top-down controls; topography.

INTRODUCTION

In western North America, the annual area burned by wildfires has substantially increased since the mid-1980s. Two main factors are believed to be driving the increase in area burned: (1) increased vertical and horizontal continuity of fuels due to previous fire suppression and other land-use practices such as logging and grazing (Hessburg et al. 2005), and (2) warmer and drier climate (Littell et al. 2010). Contemporary and historical records demonstrate that larger fires and greater area burned are positively correlated with warm and dry climate across a wide range of forest ecosystems (Heyerdahl et al. 2008a, b, Littell et al. 2009, Lutz et al. 2009, Miller et al. 2012a, Abatzoglou and Kolden 2013). Because climate change is expected to increase the length of summer drought, and the connectedness and flammability of fuels across the landscape, a significant increase in area burned is expected under climate change (Flannigan et al. 2009, 2013, Littell et al. 2010, Wotton et al. 2010).

These increases in area burned could have both desirable and adverse ecological impacts. In most ecosystems in western North America, plant and animal species evolved with fire and are adapted to it (Agee 1993, Sugihara et al. 2006). If fires burn in a way that restores pattern, structure, and function to that of pre-Euroamerican settlement, the increased area burned might help to maintain current ecosystem function and species diversity, and to increase ecosystems' resilience to a changing climate and subsequent disturbances. On the other hand, if increases in area burned bring changes in other fire-regime attributes, such as frequency, extent, spatial pattern, and severity of fires, the ecological effects of the fire will have changed and the ecosystem may be altered (Pickett and White 1985, Agee 1993, Sugihara et al. 2006).

Our understanding of large-fire growth and extreme fire behavior supports a connection between increased fire size and increased burn severity and spatial aggregation of severely burned areas. Here we define burn severity as the ecological effect of the fire on soils and plants, which includes the direct consumption of organic materials by the fire and delayed ecosystem responses, such as tree mortality, vegetation resprouting,

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³ E-mail: acansler@uw.edu

and colonization of new propagules (Cansler and McKenzie 2012). Because large fires tend to occur during periods of extreme fire behavior, we would expect increases in fire size to be correlated with increased severity and large, homogenous high-severity patches. For example, based on observations of severity and spatial pattern of the 1988 fires in Yellowstone National Park and simulation modeling, Turner and Romme (1994) hypothesized that extreme fire weather overrides the influence of local controls such as topography and fuels, causing extreme fire behavior and rapid fire growth.

The probability of weather and climate overriding local controls depends on both the strength of local controls and of the characteristic intensity of the fire regime (Agee 1997, Haire and McGarigal 2009, Thompson and Spies 2009). In high-severity fire regimes with weak local controls, such as *Pinus contorta* var. *latifolia* (lodgepole pine)-dominated forests of the Greater Yellowstone Ecosystem or boreal forests with relatively gentle topography, warm, dry climate and extreme weather produce synchronous fire behavior across the landscape (Turner and Romme 1994, Bessie and Johnson 1995). In forests with complex terrain, however, such as the Cascade Range, barriers to fire spread are present even under extreme climate and weather (Prichard and Kennedy 2012). In low-severity fire regimes, such as dry *Pinus ponderosa* (ponderosa pine) forests in which frequent fire or other processes have maintained fine-scale heterogeneity in fuels, fire-resistant traits of the dominant vegetation make synchronous high-severity fire less likely, even under extreme climate.

The goal of this study is to connect changes in climatic drivers of area burned and fire size with changes in the ecological effects of the fire in the northern Cascade Range of Washington State, USA. We focus on burn severity and burn-severity pattern, two fire-regime attributes that influence other ecosystem processes. Severity affects the composition and timing of postfire succession (Turner et al. 1997, 1999, Lentile et al. 2007); geomorphic process, such as the movement of water and sediment across the landscape (Swanson 1981); biogeochemical and belowground process, such as soil nutrient composition, physical properties, and microbial populations (Neary et al. 1999, Smithwick 2011); and wildlife habitat suitability (Roberts et al. 2008, Wightman et al. 2010, Dudley et al. 2012, Buchalski et al. 2013). The burn-severity pattern also influences postfire succession, and in some systems the severity and spatial pattern of previous disturbances can affect subsequent disturbance (Peterson 2002, McKenzie et al. 2011, Haire et al. 2013). In large high-severity patches, seed sources from wind- and animal-dispersed species may be limited, influencing the timing and composition of subsequent vegetation establishment and nutrient availability (Turner et al. 1994, Donato et al. 2009). In low-severity fire regimes, heterogeneous patterns of burn severity facilitate tree regeneration, allowing trees to survive long

enough to express fire-resistant traits such as thick bark (Agee 1993). Knowledge of how severity and its spatial patterns may change with increased fire extent suggests how other ecosystem processes may shift under climate change and what types of climate adaptation could be used to maintain ecosystem resilience into the future.

OBJECTIVES

There were three objectives to this study: (1) to examine the relationships between local climate and fire extent, (2) to test if area burned is correlated with severity and the spatial pattern of high-severity areas, and (3) to ascertain how local environmental and ecological variability mediate the relationship between fire size and burn severity or burn-severity pattern. To achieve the last, we conducted our analyses at two scales: across our whole study area, the northern Cascade Range of Washington, USA, and separately for four ecological subsections within the larger study area (Fig. 1). These subsections have different climate, topography, and vegetation, which we expected would produce different fire regimes. At each spatial extent, we tested the hypotheses that fire size is correlated with burn severity, and fire size is correlated with increased spatial aggregation of high severity. Metrics for the latter include attributes of patch-size distributions at varying scales.

METHODS

Study area

The study area encompasses 1 445 500 ha of federally managed land in the northern Cascade Range, from the western boundaries of North Cascades National Park and Glacier Peak Wilderness to the eastern boundary of the Okanogan-Wenatchee National Forest. The U.S.–Canadian border is the northern boundary and WA State Highway 2 is the southern boundary. Fifty percent (729 200 ha) of the study area is protected as wilderness. Vegetation within the study area changes gradually from moist forests on the west side of the crest of the Cascade Range to dry forests on the east side, reflecting gradients in climate, soil types, and elevation (Franklin and Dyrness 1988). In most of the study area, fire regimes are of high or mixed severity.

Ecological variation within the study area: ecological subsections

We used ecological subsections as the basis for delineating ecologically similar regions within our study area. Ecological subsections represent the smallest mapping unit identified under the USDA Forest Service's National Hierarchical Framework; larger ecoregions have been in use for over 30 years and refined through subsequent peer review (Cleland et al. 1997, ECOMAP 2007). Ecological subsections were demarcated based on climatic gradients, physiographic and geologic substrate, patterns in potential vegetation, soil groups, and hydrography. After visually evaluating

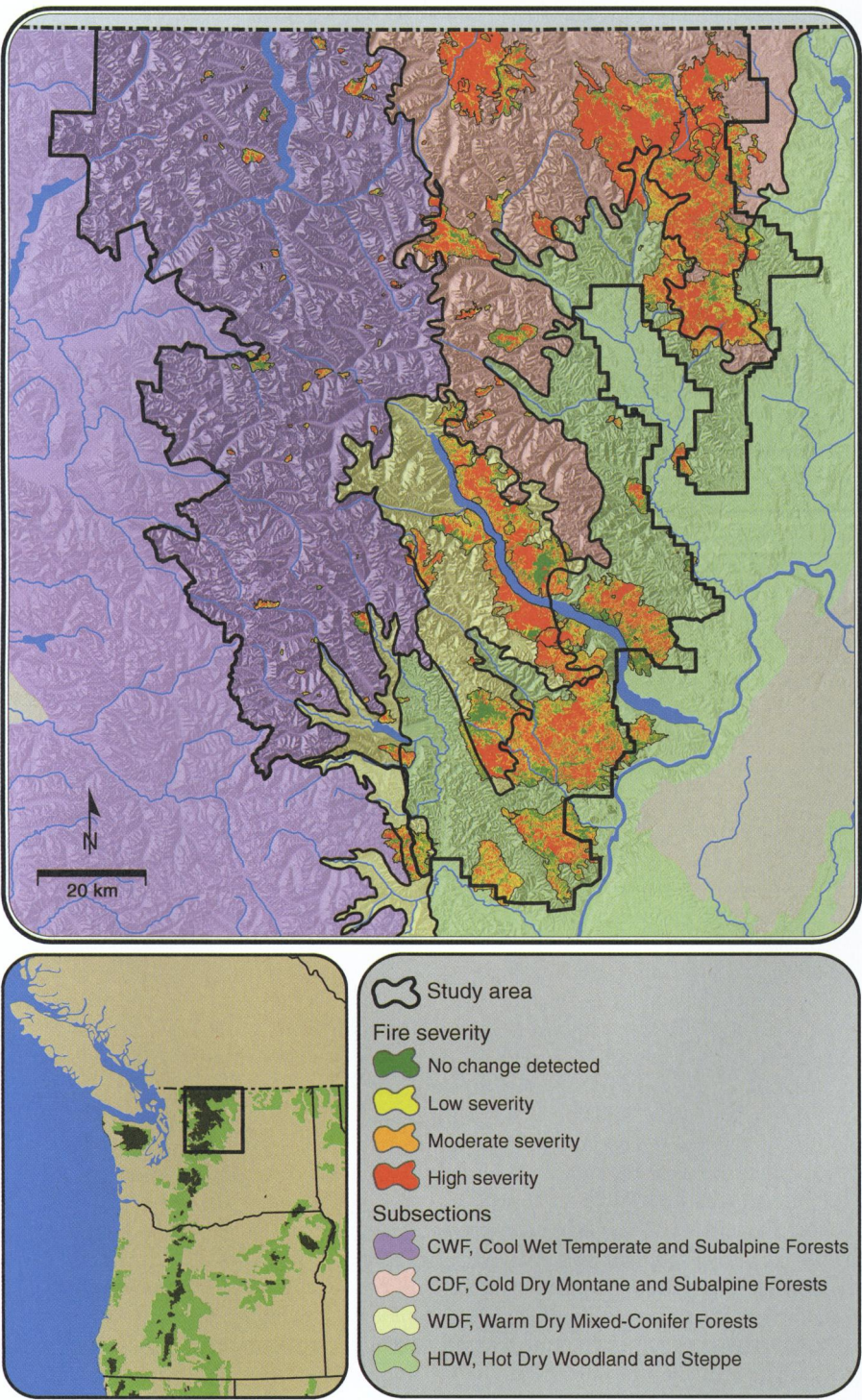


FIG. 1. Map of the burn severity of all 125 fires included in the study, overlaying the four ecological subsections used in this study. The insert reference map shows the location of the study area in the northern Cascades of northern Washington State, USA, adjacent to the U.S.–Canada border, with federal forest lands shown in light green, and lands in protected status (national parks, federally designated wilderness areas) shown in dark green.

TABLE 1. Climate, topographical variability, and vegetation types within each USDA Forest Service ecological subsection in the Northern Cascades, Washington, USA.

Variable	CWF	CDF	WDF	HDW
Climate†				
Annual maximum temperature (°C)	9 (0 to 15)	7 (3 to 14)	10 (3 to 17)	12 (7 to 17)
Annual minimum temperature (°C)	0 (−9 to 5)	−2 (−6 to 1)	0 (−6 to 4)	1 (−3 to 5)
July maximum temperature (°C)	20 (11 to 29)	20 (15 to 28)	22 (14 to 32)	26 (18 to 32)
January minimum temperature (°C)	−6 (−14 to 0)	−8 (−11 to −6)	−7 (−12 to −3)	−7 (−10 to −4)
Annual precipitation (cm)	208 (83 to 458)	107 (43 to 199)	111 (30 to 268)	64 (24 to 137)
Topographical variability				
Mean elevation (m)	1450	1785	1386	1039
Elevation range (m)	106 to 3213	624 to 2714	335 to 2875	256 to 2189
Vegetation or cover type‡ (ha and %)				
<i>T. heterophylla</i> / <i>P. menziesii</i> / <i>T. plicata</i> / <i>P. sitchensis</i>	206 617 (35%)	5343 (2%)	12 368 (7%)	3658 (1%)
Alpine/subalpine meadow or shrubland	78 661 (13%)	54 199 (15%)	17 637 (10%)	1392 (0%)
<i>T. mertensiana</i>	112 311 (19%)	3384 (1%)	3861 (2%)	35 (0%)
<i>P. albicaulis</i>	24 182 (4%)	62 219 (18%)	16 485 (10%)	583 (0%)
<i>P. engelmannii</i> / <i>A. lasiocarpa</i> / <i>P. contorta</i>	25 482 (4%)	97 146 (28%)	17 009 (10%)	4293 (1%)
<i>P. menziesii</i> / <i>A. grandis</i>	11 867 (%)	2500 (1%)	18 477 (11%)	23 050 (7%)
<i>P. ponderosa</i> / <i>P. menziesii</i>	10 874 (2%)	84 282 (24%)	49 193 (29%)	179 799 (55%)
Shrubland and grassland	52 920 (9%)	31 349 (9%)	18 665 (11%)	93 781 (29%)
Hardwood and riparian	12 875 (2%)	4668 (1%)	4940 (3%)	10 389 (3%)
Nonflammable/unlikely to burn (water, ice, barren, or sparse)	60 392 (10%)	7062 (2%)	11 395 (7%)	9916 (3%)
Total area (ha)	596 444	352 173	170 030	326 897

Note: Ecological subsection abbreviations are CWF, Cool Wet Temperate and Subalpine Forests; CDF, Cold Dry Montane and Subalpine Forests; WDF, Warm Dry Mixed-Conifer Forests; HDW, Hot Dry Woodland and Steppe.

† All climate variables are based on 1981–2010 averages from 4-km resolution PRISM gridded climate data (available at <http://prism.oregonstate.edu>; Daly et al. 2002a). Ranges reflect spatial variation in gridded climate data within the subsection; 425, 255, 115, and 229 grid cells were used in the CWF, CDF, WDF, and HDW, respectively.

‡ Classes are based on LANDFIRE existing vegetation layer (Rollins and Frame 2006; data available online at www.landfire.gov). Areas are given in hectares with percent cover within each subsection shown in parentheses. Species are *Abies grandis* (grand fir), *Abies lasiocarpa* (subalpine fir), *Picea engelmannii* (Engelmann spruce), *Picea sitchensis* (Sitka spruce), *Pinus albicaulis* (whitebark pine), *Pinus contorta* var. *latifolia* (lodgepole pine), *Pinus ponderosa* (ponderosa pine), *Pseudotsuga menziesii* var. *menziesii* (Douglas-fir), *Thuja plicata* (western redcedar), *Tsuga heterophylla* (western hemlock), *Tsuga mertensiana* (mountain hemlock).

geospatial layers of vegetation, fuels, and climate for similarities in relation to subsection boundaries, we developed four subsections that capture differences in vegetation, climate, and physiography within our study area (Fig. 1; original ecological subsections used as a basis for those in this study are in Appendix A: Fig. A1).

The Cool Wet Temperate and Subalpine Forests (CWF) have moderate summer temperatures and the highest annual precipitation and the most complex topography of all the subsections (Table 1; Appendix A: Figs. A2–A5). Low- and mid-elevations forests, dominated by *Tsuga heterophylla* (western hemlock), *Pseudotsuga menziesii* var. *menziesii* (Douglas-fir), and *Abies amabilis* (Pacific silver fir), have a low-frequency high-severity fire regime (Agee 1993). Montane forests near Ross Lake, near the center of the subsection, may burn more frequently (75–140 years) with mixed severity, and have highly variable fire-return intervals that respond to synoptic changes in climate over long time scales (Agee et al. 1990, Prichard et al. 2009). This subsection contains North Cascades National Park Complex, within which wildland fires have been allowed to burn for resource benefit since 1991. Nonflammable areas, such as lakes, glaciers, permanent snowfields, and barren rocky areas, and other areas where fire may be

virtually absent due to continuously high fuel moisture, are prevalent in the CWF (Agee et al. 1985).

The Cold Dry Montane and Subalpine Forests (CDF) include higher, cooler, and drier areas in the northeast part of the study area. This subsection also has less complex topography and large areas of continuous subalpine forests. Mid-elevation forests in the CDF feature *Abies lasiocarpa* (subalpine fir) and *Picea engelmannii* (Engelmann spruce), although *A. amabilis*, *Larix occidentalis* (western larch), and *Pinus contorta* (lodgepole pine) are also common (Fahnestock 1976), the latter two in the eastern portion. The *A. lasiocarpa*/*P. engelmannii* forests extend into subalpine areas. *Pinus albicaulis* (whitebark pine) forests with a significant component of *Larix lyallii* (subalpine larch) are common at timberline (Table 1). Fire regimes in the CDF are more similar to those in the northern Rocky Mountains than to subalpine areas in the Cascade Range, reflecting drier interior climate and the mixture of interior tree species (Fahnestock 1976, Franklin and Dyrness 1988).

The Warm Dry Mixed-Conifer Forests (WDF), located in the center of the study area east of the crest of the Cascades, include mid-elevation mixed-conifer forests (*Pinus ponderosa*, *P. menziesii* var. *glauca*, and *Abies grandis* [grand fir]), subalpine forests, and alpine

areas. This subsection has a climate similar to the CDF, but with slightly drier and warmer summers, and most of the area is lower in elevation, with more complex topography. Much of this subsection is in the drainage of Lake Chelan, a glacially formed lake, 1 mile wide by 55 miles long. Historically it had a predominantly mixed-severity fire regime (Hessburg et al. 2007).

The Hot Dry Woodland and Steppe (HDW) includes lower-elevation areas on the east side of the study area with low winter snowpack and high summer temperatures, with dry (*P. menziesii*/*P. ponderosa*) mixed-conifer forest. A low-severity fire regime was characteristic of the open grasslands and *P. ponderosa* parkland present at lower elevations (Agee 1993).

Burn-severity atlas development

Geospatial fire-occurrence data from federal land management agencies (National Park Service and U.S. Forest Service) were used to identify all fires > 10 ha that occurred in the study area from 1984 to 2008; 125 fires met the criterion (Appendix A: Table A4). Landsat satellite imagery (30-m resolution) and the Relative differenced Normalized Burn Ratio (RdNBR), a spectral burn-severity index based on the near- and mid-infrared portions of the electromagnetic spectrum (Miller and Thode 2007), were used to map burn severity (formula in Appendix A: Table A3). Higher values of RdNBR indicate higher burn severity. RdNBR and similar indices have been validated extensively with field data in temperate conifer forests (Miller et al. 2009a, Soverel et al. 2010, Cansler and McKenzie 2012). To compute the index, we used cloud-free Landsat image pairs with matching phenology, usually from the year before and the year after the fire, unless quality concerns dictated a longer interval. Some clouds were present in seven fire images, but made up <1% of the area burned in the study area or any subsection and were masked out of all analyses of severity and severity pattern. The image pairs were processed according to existing MTBS (Monitoring Trends in Burn Severity) protocols by the USGS National Center for Earth Resources Observation and Science (EROS) (Eidenshink et al. 2007).

Fire and climate

Climate variables were analyzed at the scale of the study area and for each ecological subsection. We acquired 4-km resolution (1600-ha) monthly maximum mean temperature, mean dew point temperature, and precipitation data from PRISM (Daly et al. 2002; PRISM data *available online*).⁴ The mean raster value for each observation of each climate variable was calculated within each analysis area. Gridded climate data can be influenced by the changes in the quality and total number of climate stations that are used as a basis

to interpolate climate. Because additional weather monitoring stations were established during the study period, we also developed regressions at the scale of the study area using monthly climate data from two weather monitoring stations centrally located within the study area that had consistent high-quality records. These data include mean temperature (May–September) and total monthly precipitation data from the Stehekin COOP station (48°21' N, 120°43' W, 387 m above sea level) through the United States Historical Climatology Network (USHCN; data *available online*).⁵ We acquired peak spring snow water equivalent (SWE) data (1 May; based on the median date of peak snow accumulation over the period of record) from an automated snow telemetry station (SNOTEL), the Lyman Lake SNOTEL station, 48°12' N, 120°55' W, 1823 m a.s.l. (data *available online*).⁶ PRISM data and climate station temperature and precipitation data spanned the entire study period (1984–2008; $n = 25$); analysis of SWE data is limited to 1985–2008 ($n = 24$), due to missing observations in 1984. Both the PRISM and climate station precipitation data were summarized by season.

In total, we used 13 climate variables from the PRISM data set and eight climate variables from the climate stations. Seven variables are related to the intensity of drought during the summer (summer precipitation, July, August, and September temperature, and July, August, and September dew point [PRISM only]), and five are related to moisture availability early in the fire season (spring SWE [climate station only], winter precipitation [PRISM only], spring precipitation, May, June temperature, and May, June dew point [PRISM only]). The majority of fires > 10 ha started in July and August (~65%), and thus the summer precipitation July and August temperature probably reflected the climate and weather during the peak of the fire season in the study area.

We used linear regression to test whether climate variables were significant predictors of annual area burned and annual mean fire size. All climate data were normalized and all area measurements were $\log_{10}$ -transformed for analysis, to compensate for the right-skewed area data. Regressions were built as conditional models, and only used data from years in which fires > 10 ha occurred. Predictor variables or two-way interactions that were independently significant ($\alpha = 0.1$) were included. We then used backward elimination ($\alpha = 0.05$) to select the minimum adequate model (the model that explained the most variance with the fewest variables). Likelihood ratio tests were used to choose among linear regression models. In cases where multiple models were similarly plausible, we present more than one model.

⁵ <http://cdiac.ornl.gov/epubs/ndp/ushcn/access.html>

⁶ <http://www.wcc.nrcs.usda.gov/nwcc/site?sitenum=606&state=WA>

⁴ <http://www.prism.oregonstate.edu>

Fire size and burn severity

We measured the severity of each fire using both the continuous and categorized RdNBR values. The continuous RdNBR images were converted into categorical burn-severity images, with four classes: no change detected, and low, moderate, and high severity (Miller and Thode 2007, Cansler and McKenzie 2012); see Appendix A: Table A1. Burn-severity class thresholds were developed based on regressions with field-based burn-severity data from 639 field plots located across four fires (Cansler and McKenzie 2012). Overall classification accuracy was 62% and classification was 40% better than a random classification, within the range found by similar studies in California and Canada (Miller and Thode 2007, Soverel et al. 2010). Classification accuracies of the high-severity class, emphasized in this study, were better or equal to the overall classification. Derived classification thresholds were used to create four-class categorical burn-severity images for all fires.

To measure the severity using continuous (unclassified) RdNBR values, we approximated a numerical integration of the area under the cumulative severity distribution and calculated the “severity metric,” SM (Lutz et al. 2011), for each fire as one minus the normalized area under the curve (see Appendix A: Table A3 for equation). Fires that have a greater proportion of their pixels in high RdNBR values will have less area under the cumulative distribution curve and thus a higher SM, indicating that they burned with higher severity. To compare fires, RdNBR values were limited to those between -450 and 1425 (encompassing 99% of the pixel values over all fires). For fires with unimodal RdNBR distributions, such as those in this study, the SM is highly correlated with the mean. We used the SM instead of the mean because doing so allows for comparisons of burn severity between ecosystems with very different burn-severity distributions, such as the unimodal distributions found in many temperate forests, and the bimodal distributions sometimes observed in boreal forest. Using classified and continuous severity data allowed the satellite index to be related to observed fire effects on the ground while also revealing any biases that are artifacts of categorizing the original continuous data into four classes.

We used simple linear regression to test whether fire size is a significant predictor ($\alpha = 0.05$) of the SM and proportional area in each of the four fire-severity classes. For the latter, we treated each severity class separately, although theoretically for data categorized into n categories, only $n - 1$ of the tests are truly independent, as the last category is based on the combined proportions of the other $n - 1$ categories. Fire sizes were $\log_{10}$ -transformed to account for the right skew of the data. The SM (range 0, 1) and the proportions were arcsine square-root transformed for analysis (Kutner et al. 2005). For all models in which the response variable was transformed (for this analysis and for the analysis of spatial pattern metrics to be described), we evaluated

model fit based on normal probability plots, residual plots, and partial residual plots, and identified and evaluated appropriate transformations using the Box-Cox method (Kutner et al. 2005). Regressions were developed at the scale of study area and of each ecological subsection. In cases where a fire burned in more than one subsection, the fire was assigned to the subsection in which it burned the most area.

Fire size and burn-severity pattern

We restricted our analysis of within-fire burn-severity pattern to only the high-severity class. As discussed previously, the high-severity class is of concern in relation to climate-driven shifts in fire regimes, and can be classified with the greatest accuracy (Cansler and McKenzie 2012). We chose seven spatial-pattern metrics to quantify four aspects of spatial pattern: patch size, size inequality of patches, patch interior, and spatial aggregation at the grain of the pixel (30 m) (equations in Appendix A: Table A3). Most spatial pattern metrics were calculated with FRAGSTATS (McGarigal et al. 2002).

Patch size (two metrics).—We used two metrics of patch size: mean patch size and area-weighted mean patch size. The former gives equal weight to each patch, regardless of its size; the patches themselves are the population that the mean is quantifying. The latter weights each patch by its size, reflecting the patch size in which a randomly chosen pixel would most likely be found. In landscapes with a strongly right-skewed frequency distribution of patch sizes, such as the burn-severity patches in this study, there will be large differences between these two measures of patch size.

Size inequality (one metric).—We quantified the extent to which larger patches dominated the landscape with the Gini coefficient, which provides a relative measure of the size inequality of the patch distribution. Unlike mean patch size, the Gini coefficient is calculated relative to the minimum and maximum patch sizes observed, and therefore it is a robust method for comparing fires of different sizes, which intrinsically have different maximum patch sizes.

Patch interior (two metrics).—Core area is the area within patches greater than a specified distance from the edge of the patch. More core area implies large patches that are simple in shape, whereas less core area indicates small patches, patches with convoluted edges, or patches with interior areas of different severity. Regressions with fire size were based on the proportion of the core area that was more than 90 m from the edge of a patch. We have higher confidence that at a two-pixel depth within a patch, the patch actually did burn with high severity. Because the distance used to define core area could potentially influence regression results, we also conducted a post hoc sensitivity analysis of the effect of the distance used to define core area. We tested regressions predicting core area based on fire size from 30 m to 300 m (in 30-m intervals).

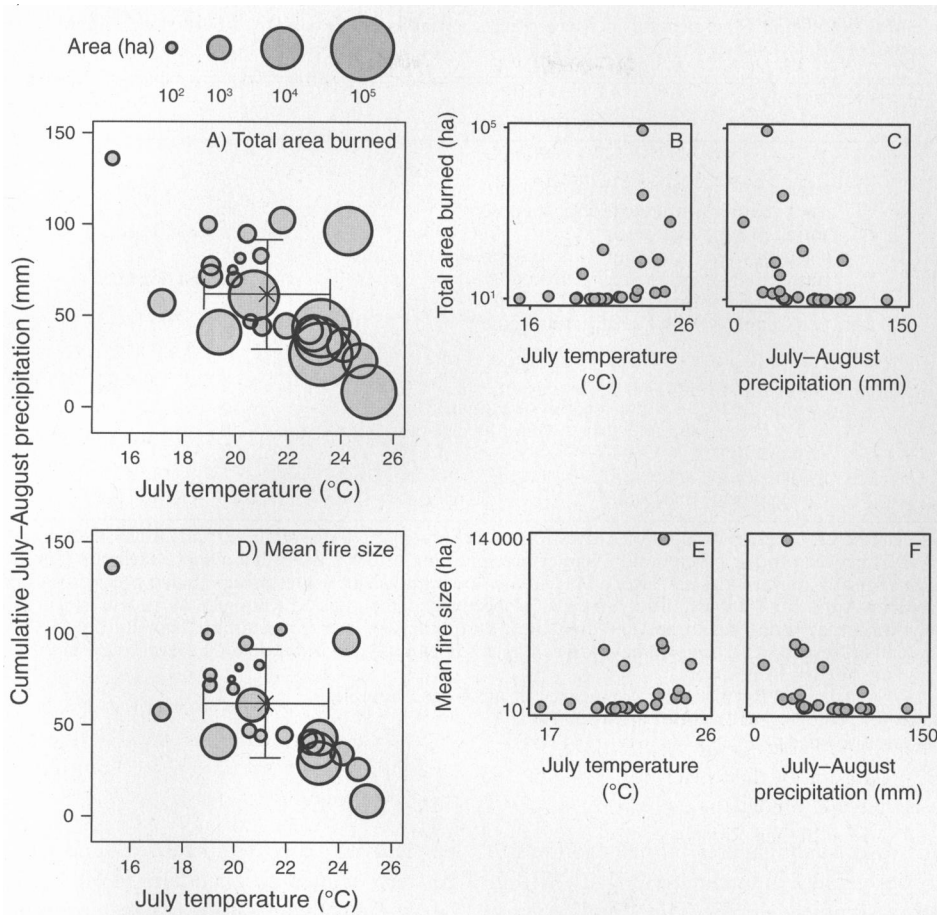


FIG. 2. Scatterplots of July temperature and the total precipitation during July and August, with plot symbols scaled by (A) the total annual area burned (ha) and (D) the annual mean fire size (ha): error bars with X show 25-year mean $\pm$ SD for July temperature and cumulative July–August precipitation. Smaller panels on the right are climate variables plotted against untransformed responses of (B, C) annual total area burned and (E, F) annual mean fire size.

Spatial aggregation at the scale of the pixel (two metrics).—Many statistics that assess spatial pattern vary systematically with landscape size (Neel et al. 2004). We used two metrics to measure the aggregation of the high-severity class at the grain of individual pixels that are not correlated with landscape extent: the proportion of like adjacencies (PLADJ) and the Clumpiness Index (CLUMPY). The PLADJ is the number of pixel edges that share an edge with a pixel of the same class, divided by the total number of possible adjacencies. CLUMPY measures the deviation of the PLADJ from that expected under a spatially random distribution.

Statistical analysis.—Simple linear regression was used to test whether fire size was a significant predictor ($\alpha = 0.05$) of within-fire burn-severity pattern. Area measurements (fire size, mean patch size, and area-weighted mean patch size) were log₁₀-transformed for analysis. Mean patch size, area-weighted mean patch size, and Gini coefficients were analyzed for the 120 fires that had high-severity area present. Metrics that assessed pattern at the scale of the pixel were analyzed

using data from 116 fires; nine fires (all but one of which were <60 ha) were excluded because they either had no high-severity area present or had a total high-severity area < 0.5 ha (five pixels), rendering those spatial-pattern metrics incalculable. For the Gini coefficient, we fit a nonlinear model because it was theoretically more appropriate, as values for the Gini coefficient should asymptote near its maximum value of 1 and residual plots and visual evaluation indicated that linear models fit the data poorly. To model the proportion of high-severity core area and the PLADJ, an arcsine square-root transformation was applied to the response variable (Kutner et al. 2005). All statistical tests in this study were conducted in the statistical programming language R (R Development Core Team 2010).

RESULTS

Fire and climate

Climate is an important driver of the annual area burned and fire size in the northern Cascade Range. The responses of area burned and fire size were nonlinear,

TABLE 2. Model results for multiple regressions between climate variables and annual area burned, and annual mean fire size for the study area and ecological subsections.

Response variable, by analysis area†	Predictors	F	df	P	R ² ‡
Area burned					
Study area§	temp.july** + temp.aug* + temp.june : swe*	10.15	3, 20	<0.001	0.54
Study area¶	tmax.july*** + winter.precip : tmax.sept**	13.56	2, 22	<0.001	0.51
CWF¶	spring.precip : sum.precip*	6.22	1, 17	0.023	0.22
CDF¶ (model 1)	spring.precip + tmax.july + spring.precip : tmax.aug	6.76	3, 10	0.009	0.57
CDF¶ (model 2)	tmax.july* + spring.precip : tmax.aug*	6.84	2, 11	0.012	0.47
WDF¶	winter.precip : tmax.sept*	8.87	1, 8	0.018	0.47
HDW¶	tmax.july* + winter.precip : tmax.june**	7.97	2, 14	0.005	0.47
Mean fire size					
Study area§	temp.july* + sum.precip*	8.91	2, 22	0.001	0.40
Study area¶	sum.precip*** + winter.precip : tmax.june**	13.16	2, 22	<0.001	0.50
CWF¶	spring.precip : sum.precip* + winter.precip : tmax.sept*	7.29	2, 16	0.006	0.41
CDF¶	sum.precip***	21.67	1, 12	0.001	0.61
WDF¶	winter.precip : tmax.sept*	7.71	1, 8	0.024	0.43
HDW¶	winter.precip : tmax.june*	8.25	1, 15	0.012	0.31

Notes: Climate predictor variables for multiple regressions were chosen based on significant univariate predictors and significant interactions ($\alpha = 0.1$) between spring precipitation, winter precipitation, or SWE and other climate variables (Appendix B: Tables B1–B6). Acronyms for climate variables are: swe, 1 May snow water equivalent; winter.precip, spring.precip, and sum.precip, total precipitation October–April, May–June, and July–August, respectively; temp.aug and temp.july, mean temperature for August and July, respectively; tmax.may, tmax.june, tmax.july, tmax.aug, and tmax.sep, average maximum temperature for May, June, July, August, and September, respectively; tmean.may, tmean.june, tmean.july, and tmean.aug, average dew point temperature for May, June, July, and August, respectively.

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$ for significance of individual variables.

† Abbreviations for ecological subsections are as in Table 1.

‡ Adjusted R^2 are provided.

§ COOP and SNOTEL data.

¶ PRISM data.

with drastic increases in annual area burned and fire size in response to warmer and drier climate (Fig. 2). Within the study area, summer climate (July and August temperatures and summer precipitation) explained the most variance and had more significant relationships with area burned and fire size in univariate models. July and August temperatures were highly correlated with summer precipitation, however, probably explaining why one was often dropped from multi-predictor models. We found significant positive relationships between temperature and negative relationships between SWE (snow water equivalent) or precipitation and both the area burned and fire size (Table 2; Appendix B: Tables B1–B6). Antecedent climate (spring SWE, winter and spring precipitation) was equally significant in many models, but slopes for antecedent climate variables were less steep and less variance was explained compared to summer season variable models. Summer temperature,

particularly July temperature, and the interaction between winter precipitation (SWE or rainfall) and late-summer temperature (August or September) were significant predictors in the models predicting area burned for the study area and individual subsections (Table 2). Summer (study area, CWF, CDF) or winter (study area, CWF, HDW, WDF) precipitation were consistently significant predictors of mean fire size (Table 2).

Fire size and burn severity

Burn severity reflected differences in fire regimes within the study area. Over the whole study area, 316 567 ha, or 22% of the study area, burned during the study period, and 41% of that area burned with high severity (Table 3). High and moderate severity constituted most of the area burned, in all subsections except the CWF (Table 3). The HDW had more moderate

TABLE 3. Burn severity by ecological subsection.

Analysis area	No. fires	Total area burned (ha)†	Area burned, by severity (ha)‡			
			High	Moderate	Low	Unchanged
Study area	125	316 567 (22%)	129 238 (41%)	84 181 (27%)	46 041 (15%)	55 720 (18%)
CWF	51	9541 (2%)	2329 (25%)	2569 (27%)	1696 (18%)	2896 (31%)
CDF	32	155 211 (44%)	80 972 (53%)	34 625 (22%)	18 725 (12%)	19 593 (13%)
WDF	12	46 078 (27%)	16 050 (35%)	13 552 (29%)	7125 (15%)	9353 (20%)
HDW	30	105 737 (32%)	29 887 (28%)	33 435 (32%)	18 495 (17%)	23 878 (23%)

† Percentage of subsection or study area is shown in parentheses.

‡ Percentage of total area burned at a given severity is shown in parentheses.

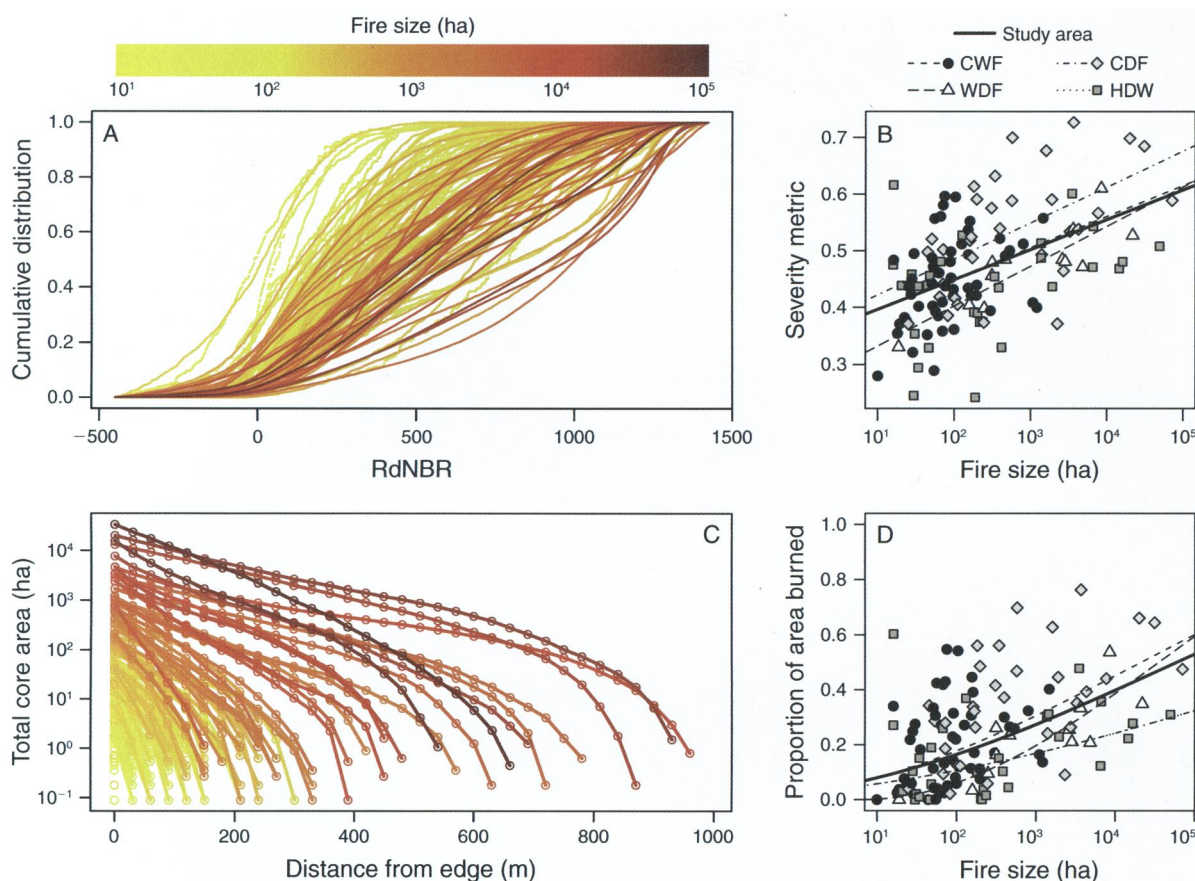


FIG. 3. (A) Cumulative distribution of the RdNBR (Relative differenced Normalized Burn Ratio) for all 125 fires > 10 ha that burned between 1984 and 2008. RdNBR is a spectral burn-severity index based on the near- and mid-infrared portions of the electromagnetic spectrum. Each distribution is colored according to the size of the fire. Fires that burned with higher severity have a slower increase in their cumulative distribution, and less area under the cumulative distribution curve. This is quantified by the severity metric (SM). (B) A scatterplot and linear regressions of fire size and the SM, showing a significant increase in severity with fire size for the study area and all ecological subsections except HDW (see Fig. 1 for locations: CWF, Cool Wet Temperate and Subalpine Forests; CDF, Cold Dry Montane and Subalpine Forests; WDF, Warm Dry Mixed-Conifer Forests; HDW, Hot Dry Woodland and Steppe). (C) Total high-severity core area in each fire, when core area is defined by a specific distance from the edge of the patch (shown here in 1 pixel/30 m increments). Lines represent individual fires, and are colored according to the size of the fire. (D) Scatterplots and significant linear regression equations for the proportion of area burned at high severity as a function of fire size, for the study area and each subsection. Only statistically significant regressions ($P < 0.05$) are shown.

severity than high severity, whereas the CDF and WDF burned predominantly with high severity. The CDF had largest total area burned (155 211 ha, or 44% of the subsection) and the greatest percentage of high severity (53% of the area burned in the subsection) of the four subsections, but had relatively less area burned at moderate severity (Table 3).

Burn severity increased with fire size. The cumulative distributions of the RdNBR for each of the 125 fires (Fig. 3A) demonstrate that the larger fires (red colors) have a greater proportion of their pixels in high RdNBR values. Some are even concave-up (i.e., no inflection point), whereas smaller fires (yellow colors) have sigmoidal cumulative distributions, indicating a higher proportion of the pixels in lower RdNBR values. The SM, which quantifies these curves, significantly increases with fire size at the extent of the study area ($R^2 = 0.23$, P

< 0.001) and in all subsections except the HDW (Table 4, Fig. 3C). The proportion of the area burned at high severity increased as fire size increased at the scale of the study area ($R^2 = 0.22$, $P < 0.001$) and in all subsections (Fig. 3D), and the proportion of area where no change was detected decreased as fire size increased across the study area ($R^2 = 0.19$, $P < 0.001$) and in all subsections except the HDW (Table 4).

Fire size and burn-severity pattern

Patch size.—At the extent of the study area, high-severity mean patch size and area-weighted mean patch size both increased significantly with fire size (Table 5). The relationship between fire size and area-weighted mean patch size followed a power-law distribution (Fig. 4). Within the subsections, high-severity mean patch size increased significantly with fire size only in the CDF

TABLE 4. Regression models predicting the severity metric (SM) and proportion of area burned at a given severity as a function of fire size, for the whole study area and for each subsection.

Burn severity, by analysis area	Slope (SE)	<i>P</i>	Adjusted <i>R</i> ²
Severity metric (RdNBR)			
Study area	0.11 (0.02)	<0.001	0.23
CWF	0.11 (0.04)	0.009	0.11
CDF	0.13 (0.04)	0.001	0.28
WDF	0.14 (0.03)	<0.001	0.70
HDW	0.06 (0.03)	0.063	0.09
High			
Study area	0.27 (0.04)	<0.001	0.22
CWF	0.30 (0.12)	0.012	0.10
CDF	0.30 (0.08)	0.001	0.28
WDF	0.42 (0.08)	<0.001	0.70
HDW	0.18 (0.08)	0.036	0.12
Moderate			
Study area	0.02 (0.02)	0.368	0.00
CWF	0.11 (0.06)	0.073	0.04
CDF	-0.07 (0.03)	0.019	0.14
WDF	0.10 (0.05)	0.091	0.18
HDW	0.06 (0.06)	0.383	-0.01
Low			
Study area	-0.05 (0.02)	0.009	0.05
CWF	-0.02 (0.04)	0.662	-0.02
CDF	-0.09 (0.03)	0.008	0.19
WDF	-0.17 (0.04)	0.002	0.60
HDW	-0.02 (0.04)	0.589	-0.02
No change detected			
Study area	-0.19 (0.03)	<0.001	0.19
CWF	-0.28 (0.09)	0.003	0.15
CDF	-0.17 (0.06)	0.007	0.19
WDF	-0.23 (0.05)	0.002	0.61
HDW	-0.12 (0.08)	0.152	0.04

Notes: The SM was calculated following Lutz et al. (2011) using continuous RdNBR (Relative differenced Normalized Burn Ratio) images clipped to the boundary of each fire, and modeled as $y = b_0 + b_1(\log_{10}(TA))$; higher values of the SM indicate increased burn severity. For the percentage of area at a given severity class, models are also in the form: $y = b_0 + b_1(\log_{10}(TA))$, where TA is total area of the fire. The SM and proportions were arcsine square-root transformed before analysis (Kutner et al. 2005).

(Table 5, Fig. 4A). High-severity area-weighted mean patch size increased significantly with fire size in all subsections, but more variance was explained in the CDF and WDF ($R^2 = 0.75$, $P < 0.001$ and 0.73 , $P < 0.001$, respectively), than in the CWF or HDW ($R^2 = 0.38$, $P < 0.001$ and $R^2 = 0.49$, $P < 0.001$, respectively). Regression for area-weighted mean patch size explained much more of the variance than the regression between fire size and mean patch size (Table 5), meaning that the patch size in which a randomly selected pixel is likely to be located increased more with fire size than did the size of a randomly selected patch.

Patch-size distributions reflected the size of fires in the subsection and the proportion of area burned at high severity (Appendix B: Fig. B1, Table B7). In the CDF, which had the largest fires and greatest proportion of high severity, a large percentage of the area burned

(37%) was within high-severity patches larger than 10^3 ha. In the CWF, which had the lowest total area burned and smaller fires, most of the area that burned at high severity was in patches of 10 – 10^3 ha. When calculated across high-severity patch distributions aggregated over all fires in a subsection, area-weighted mean patch sizes were highest in the CDF (7335 ha), followed by the WDF (807 ha), the HDW (110 ha), and lowest in the CWF (15 ha) (Appendix B: Table B7).

Size inequality.—The size inequality of high-severity patches, as measured by the Gini coefficient, increased with fire sizes both at the scale of the study area and within each subsection (Fig. 4C). Nonlinear models explained 56% of the variance at the scale of the study area, and 49%, 71%, 50%, and 45% of the variance in the CWF, CDF, WDF, and HDW, respectively (based on pseudo- R^2 ; Appendix B: Tables B8–B12). The parameters of the nonlinear models were highly significant at the scale of the study area and in the CDF, but were not always significant at $\alpha = 0.05$ in other subsections (especially CWF). Nevertheless, residual plots and visual examination of scatterplots indicated that nonlinear models were appropriate, whereas linear models poorly fit the data, despite having $R^2 > 0.44$ and $P < 0.001$ in each analysis area (not shown). When calculated across high-severity patch distributions aggregated over all fires within a given size class, the Gini coefficients were higher in the larger fire size classes, and highest in the CDF (0.97), followed by the WDF (0.93), the HDW (0.91), and lowest in the CWF (0.88) (Appendix B: Table B7).

Patch interior.—The proportion of high-severity core area increased significantly with fire size across the study area ($R^2 = 0.38$, $P < 0.001$) and within all subsections (Table 5, Fig. 4D). The increase was greater and more variance was explained in the CDF and WDF than the CWF and HDW (Table 5). In our post hoc sensitivity analysis testing definitions of core area ranging from 30 m to 300 m, regressions at the scale of the study area explained 27% to 47% of the variance and were consistently significant at the $P < 0.001$ level (Appendix B: Figs. B2 and B3).

Larger fires had more total core area across all definitions of core area (0–990 m in from patch edge) and had high-severity core area present at much greater distances from patch edges than did smaller fires (Fig. 3C). The relationship between core area and fire size was not absolute. The amount of high-severity core area in some of the largest fires dropped off at larger edge depths (>500 m), but three fires (the third, fifth, and eighth largest fires, of sizes 31 340, 20 070, and 8533 ha, respectively) had high-severity area >800 m in from patch edges.

Spatial aggregation at the scale of the pixel.—Over the entire study area, the PLADJ increased with fire size (Table 5, Fig. 4E), but no significant relationship with fire size was apparent when spatial aggregation was measured with CLUMPY ($P = 0.057$). There was a significant increase in the PLADJ in all subsections

TABLE 5. Regression models predicting spatial-pattern metrics for high severity as a function of fire size at the extent of each ecological subsection.

Spatial-pattern metric by analysis area	Slope (SE)	F	df	P	R ²
Mean patch size					
Study area	0.19 (0.06)	11.78	1, 114	<0.001	0.09
CWF	0.25 (0.15)	2.73	1, 45	0.105	0.04
CDF	0.27 (0.1)	7.78	1, 30	0.009	0.18
WDF	0.21 (0.17)	1.43	1, 9	0.263	0.04
HDW	0.14 (0.11)	1.72	1, 24	0.202	0.03
Area-weighted mean patch size					
Study area	0.99 (0.07)	179.80	1, 114	<0.001	0.61
CWF	0.89 (0.17)	28.86	1, 45	<0.001	0.38
CDF	1.16 (0.12)	94.97	1, 30	<0.001	0.75
WDF	1.07 (0.2)	28.39	1, 9	<0.001	0.73
HDW	0.86 (0.17)	25.43	1, 24	<0.001	0.49
Gini coefficient					
Study area	0.19 (0.02)	96.25	1, 114	<0.001	0.45
CWF	0.31 (0.05)	36.56	1, 45	<0.001	0.44
CDF	0.15 (0.03)	33.04	1, 30	<0.001	0.51
WDF	0.11 (0.03)	14.50	1, 9	0.004	0.57
HDW	0.21 (0.05)	21.66	1, 24	<0.001	0.45
Proportion of total area burned >90 m from patch edge (core area)					
Study area	0.25 (0.03)	70.67	1, 114	<0.001	0.38
CWF	0.22 (0.07)	10.06	1, 45	0.003	0.16
CDF	0.35 (0.06)	29.02	1, 30	<0.001	0.47
WDF	0.27 (0.10)	7.86	1, 9	0.021	0.41
HDW	0.15 (0.04)	11.97	1, 24	0.002	0.30
Proportion of like-adjacencies					
Study area	0.22 (0.04)	25.98	1, 114	<0.001	0.18
CWF	0.27 (0.1)	7.73	1, 45	0.008	0.13
CDF	0.25 (0.07)	13.81	1, 30	<0.001	0.29
WDF	0.21 (0.11)	3.72	1, 9	0.086	0.21
HDW	0.23 (0.11)	4.37	1, 24	0.047	0.12
Clumpiness index					
Study area	0.03 (0.02)	3.71	1, 114	0.057	0.02
CWF	0.01 (0.03)	0.14	1, 45	0.713	-0.02
CDF	0.05 (0.03)	3.82	1, 30	0.060	0.08
WDF	0.03 (0.05)	0.46	1, 9	0.513	-0.06
HDW	0.06 (0.05)	1.40	1, 24	0.248	0.02

Notes: Spatial-pattern metrics were modeled with a simple linear regression following the form: $y = b_0 + b_1(\text{TA})$, where TA is the total area of the fire. Proportions were arcsine square-root transformed before analysis (Kutner et al. 2005). Area measurements (total area of the fire, mean patch size, and area-weighted mean patch size) were $\log_{10}$ -transformed before analysis. Results for nonlinear models for Gini coefficient are present in the text and Appendix B: Tables B8–B12.

except for the WDF, but none of the regressions between CLUMPY and fire size were significant (Table 5, Fig. 4F). Therefore the significant relationships between fire size and the PLADJ were due to the increased proportion of area burned at high severity in larger fires.

DISCUSSION

Fire and climate

The larger climate patterns experienced in the northern Cascade Range facilitate interpretation of the climate variables found to be significant predictors. Whereas in many regions in the western USA the fire season consistently starts by July, conditions in the CWF may stay or become cool and wet even in the middle of the summer. Therefore the flammability of

fuels across the landscape is limited not only by spring climate, but also by summer temperature and precipitation. This differs from the HDW, where fuels almost always become dry enough during the summer to burn. In these areas, winter and spring climate, particularly winter precipitation and the temperatures at the beginning and end of the fire season, influence the overall length of the fire season and production of fine fuels, and therefore are more clearly associated with annual area burned and fire size. These results suggest that the importance of climate variables in controlling area burned and fire size is contingent. Adequate conditions for burning during the summer first need to be met before variation in the length of the fire season becomes important.

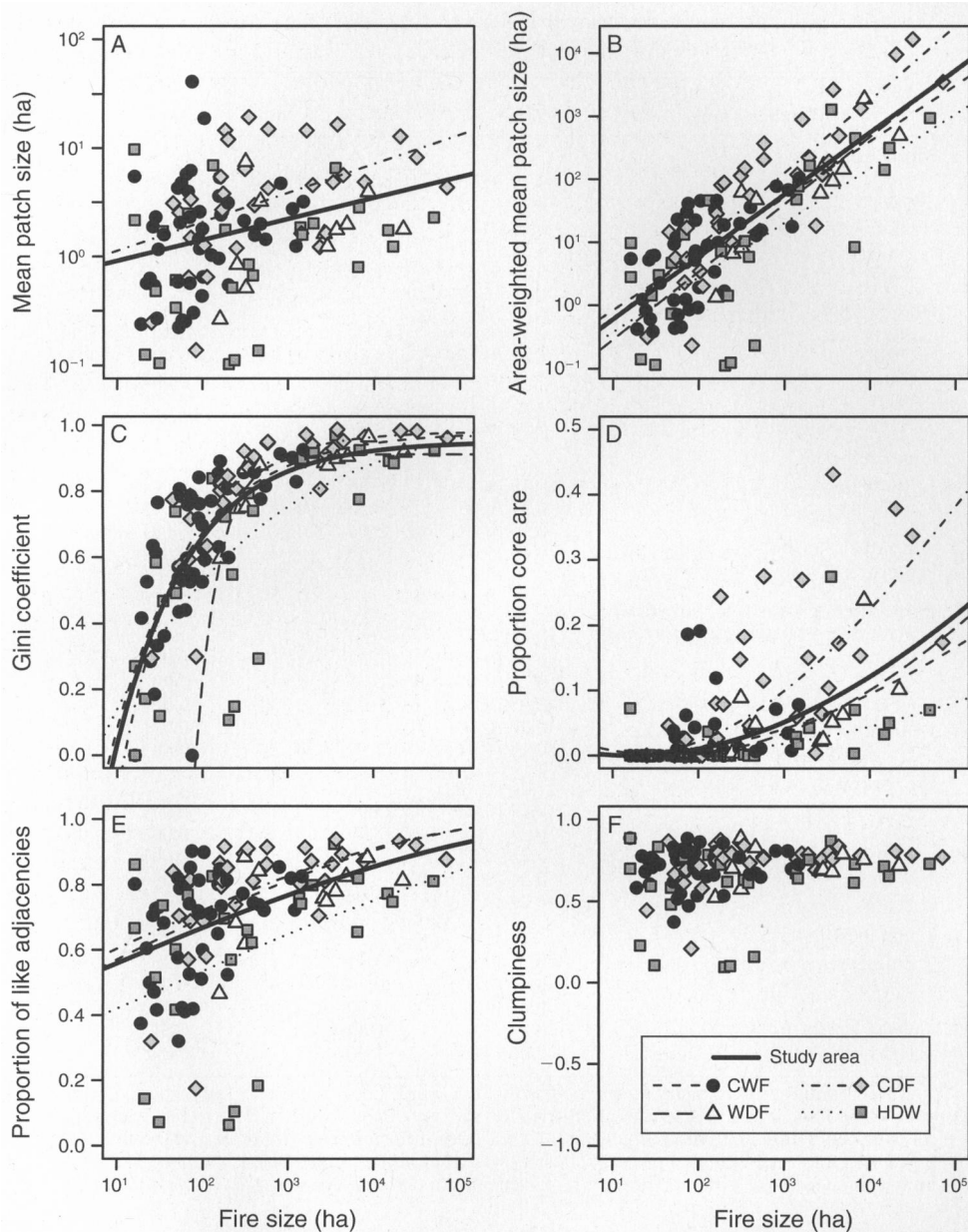


FIG. 4. Scatterplots and regression equations for fire size and (A) mean high-severity patch size; (B) area-weighted mean high-severity patch size; (C) Gini coefficient for high-severity patches; (D) the proportion of the total area burned made up of high-severity core area (where core area is defined as being >90 m from a patch edge); (E) the proportion of like adjacencies (PLADJ); and (F) the high-severity Clumpiness index (CLUMPY). Only statistically significant regressions ($\alpha = 0.05$) are shown. Area-weighted mean patch size weights each patch by its size, reflecting the patch size in which a randomly chosen pixel would most likely be found; like mean patch size, it provides an absolute measure of patch size. The Gini coefficient measures the size inequality of the patch distribution relative to minimum and maximum patch sizes observed, and measures patch size relative to fire size, allowing comparisons among fires of different sizes, with different maximum possible patch sizes. PLADJ and CLUMPY measure the aggregation of the high-severity class at the grain of individual pixels. PLADJ is the number of pixel edges that share an edge with a pixel of the same class, divided by the total number of possible adjacencies. CLUMPY measures the deviation of the PLADJ from that expected under a spatially random distribution, given the proportional cover of the high-severity class.

Although contemporary relationships between climate and area burned have been used to project future changes (Littell et al. 2010, Wotton et al. 2010), these relationships may not be stationary under future climate. In the Pacific Northwest, climate models project

increases in annual mean temperatures and possible increases in annual precipitation, but decreases in summer precipitation and relatively larger increases in summer temperatures (Mote and Salathé 2010). If summer climate in the northern Cascade Range changes

so that thresholds for burning are always reached during the summer, a shift in the specific climate factors that constrain area burned may occur. Climate variables such as spring SWE that are related to the start of the fire season may become more important drivers of annual area burned and fire size. The strength of the relationships between climate and the annual area burned and fire size in the future will also be mediated by the extent to which previous disturbances decrease the probability or severity of subsequent disturbance (Peterson 2002, Moritz et al. 2011). Uncertainty arises from lack of knowledge of whether fire regimes in which previous fires currently inhibit the spread of subsequent fires will continue to exhibit this trait under future climate (McKenzie and Littell 2011).

Translating observed increases in area burned to landscape effects

In the northern Cascade Range, larger fires created a more homogeneous landscape pattern by increasing patch sizes while reducing the proportion of the landscape in low and moderate severity. Burn severity and high-severity patch size, patch size inequalities, patch interior, and spatial aggregation of high-severity pixels were positively correlated with fire size. Our results correspond with significant relationships between fire size or area burned and burn severity and high-severity patch size in the Sierra Nevada (Lutz et al. 2009, Miller et al. 2009b, 2012a, b), and between area burned and burn severity in the Rocky Mountains (Dillon et al. 2011).

Despite the increasing evidence that increases in fire size and area burned correspond with increases in severity and larger, more homogenous high-severity patches, large fires should not simply be seen as ecological catastrophes (Keane et al. 2008). In our study area, large fires created more *total* area in smaller patch sizes than smaller fires did, even though larger patches made up a greater *proportion* of the area within large fires. Fires > 10 000 ha probably maintained the heterogeneity of 151 072 ha that were outside large (>100 ha) patches of high severity. This amounts to over 47.7% of the total area burned in the study area, and is twice as much area as the area outside of large patches burned by fires < 10 000 ha (71 531 ha). Therefore, within a least part of the area burned by large fires, landscape heterogeneity has increased, whereas in other portions, large homogenous patches were created.

We found little support for the idea that differences in fire suppression policies were influencing area burned or burn-severity patterns. Unlike many other large wilderness complexes, fires are actively suppressed in the majority of wilderness areas within our study area. Wildland fire use is permitted within the boundaries of North Cascades National Park Complex, which makes up ~18% of our total study area, almost entirely within the CWF. Unlike other wilderness areas with wildland fire use policies in places, where previous fires have been

shown to limit the spread or severity of subsequent fires (Collins et al. 2009, Lutz et al. 2009, Teske et al. 2012, Parks et al. 2014; but see Haire et al. 2013), fires in the CWF seem to be limited in size by the cool wet climate and limited fuel connectivity, as seen by their small size and spatial isolation (Fig. 1). We did not examine differences between wilderness and non-wilderness in other parts of our study area, as any differences most likely would be due to ecological differences: most of the wilderness is higher in elevation, with different forest types and different fire regimes, which are already captured within the different subsections.

Differences in fire regimes between the subsections directly reflect their differences in vegetation, topography, and climate. The strongest relationship between fire size and the spatial aggregation of high severity was in the CDF, reflecting connected forests of fire-intolerant tree species, few barriers to fire spread, and weather during the large fires. In this subsection, extreme fire weather and dry climate did seem to override local controls (*sensu* Turner and Romme 1994, Bessie and Johnson 1995), but that occurred within a context of a biophysical setting with weak “bottom-up” controls. In the CWF, the smaller fire sizes, smaller patch sizes, less core area, more equal patch-size distributions, and greater spatial complexity, relative to other subsections, reflected the constraints of complex topography, higher fuel moistures, and nonflammable areas on fire behavior and spread. In the HDW, the topography is relatively simple, but the fine-grain mosaics of vegetation and stand structure in low- and mid-elevation forest acted as strong local controls that keep weather from having an overriding influence on fire behavior (Agee 1997).

Our results provide the basis for the conceptual model presented in Fig. 5, in which the relationship between fire size and spatial pattern of severity is mediated by two main factors: (1) the severity of the fire regime, and (2) the strength of local “bottom-up” controls. Local controls include topographical barriers to fire spread, the variability or “complexity” of topography, and the spatial complexity of vegetation and fuels. In active low-severity fire regimes, fuel limitations and adaptive traits of the vegetation may limit the occurrence of high burn severity, even when fires are large. In mixed- and high-severity fire regimes, fuels are not limiting and the full range of severity can be expressed (Agee 1997). The spatial complexity and configuration of fuels is influenced by feedbacks from previous disturbances, temporal variation in climate, and topography. In areas with complex topography or greater spatial variability in vegetation and fuels, climate and weather are less likely to override those local “bottom-up” controls (Kennedy and McKenzie 2010, McKenzie and Kennedy 2012), which constrain fire behavior and the creation of large, homogeneous high-severity patches. In these areas, instead of “overriding” or removing the influence of fuels and topography, climate may increase the spatial

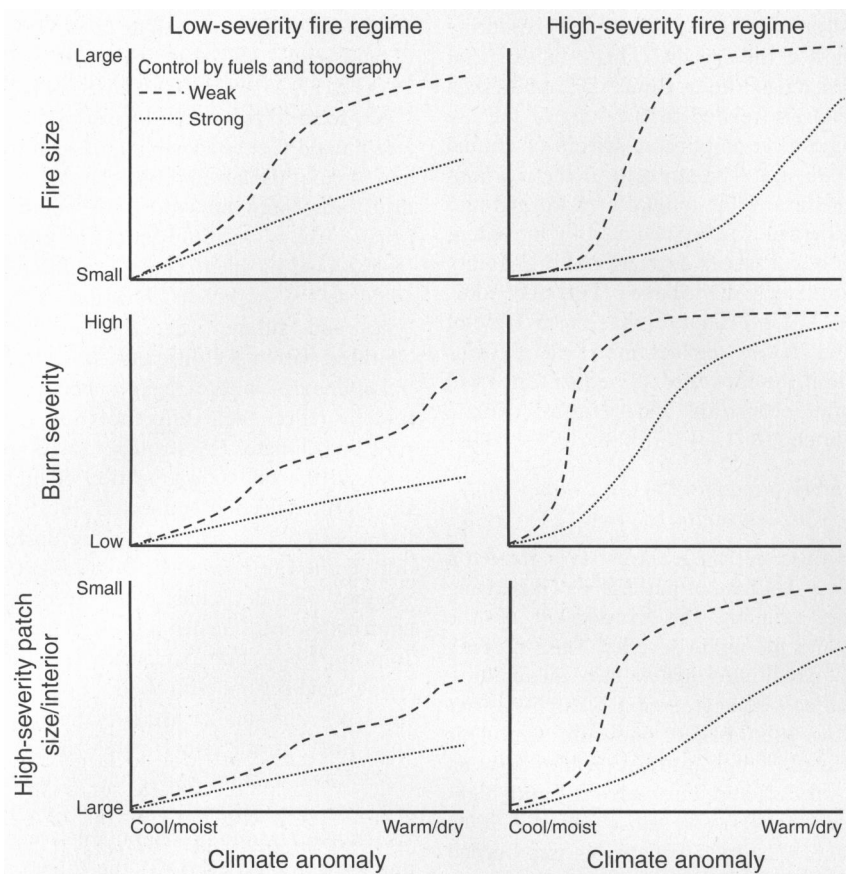


FIG. 5. Conceptual diagram of theoretical constraints imposed by (1) the severity of the fire regime and (2) the strength of local endogenous controls (i.e., fuels and topography) on the relationship between climate and fire-regime attributes (fire size, burn severity, and the size of high-severity patches). Lines represent the aggregate properties of multiple fires over time, as opposed to individual events. We hypothesize that in systems with weak endogenous controls, nonlinear changes in fire-regime attributes are more likely. In high-severity fire regimes, this is reflected in sudden and marked increases in fire size and severity, and the creation of extremely large high-severity patches. In lower-severity fire regimes, this may be reflected in simply a faster increase in fire size, severity, and high-severity patch size in areas with weak local controls, or as shown here for fire severity and high-severity patch size, a step-like increase.

grain at which those local controls influence burn-severity pattern.

An important question put forth by our conceptual model (Fig. 5) is how “top-down” climatic controls, the severity of fire regimes, and variation in drivers at local and meso scales interact to create linear or nonlinear changes in the emergent properties of fire regimes. Statistically significant relationships between fire size and burn severity are found in some regions—interior Alaska (Duffy et al. 2007), the northern Rocky Mountains (Keane et al. 2008, Dillon et al. 2011), Southern Rockies (Dillon et al. 2011), the Sierra Nevada, and southern Cascade Range (Lutz et al. 2009, Miller et al. 2012a)—but not others (Pacific Northwest, Inland Northwest, Colorado Plateau, and Mogollon Rim; Dillon et al. 2011). Although these results may be due, in part, to differing methodologies among studies, as we will discuss, across the larger-scale variation in the western USA they agree with our model. Stronger relationships between area burned and severity

are present in higher-severity fire regimes, such as more continuous mixed-conifer and subalpine forests in the northern Rocky Mountains and the Sierra Nevada. Regions where significant relationships have not been detected, such as the Inland Northwest or southwestern USA, have lower-severity fire regimes and more heterogeneous fuels, indicative of *Pinus ponderosa* and mixed-conifer forests.

As with climate–fire size relationships, the relationship between fire size and severity and the spatial pattern of burn severity may not remain stationary under a future climate. If the relationship between fire size and severity is nonlinear, then thresholds associated with climate may be reached after which the emergent properties of fire regimes shift suddenly. Other factors may either ameliorate or exacerbate the homogenizing influence of increased severity and increased high-severity patch size. The cumulative impacts of increases in the fire size, severity, and high-severity patch size and patch interior on the long-term landscape pattern will

depend on feedbacks from future processes, including availability of seed sources, spatial variability of vegetation recovery within large high-severity patches, and future disturbances. For example, the 90-m distance that we used to define core area corresponds to the distances within which >90% of wind-dispersed seeds of many temperate conifer species fall (McCaughey et al. 1986). Early-successional vegetation may persist for a long time within large patches with a substantial core area due to limitation to seed dispersal from the edge of patches (Donato et al. 2009, Haire and McGarigal 2010), although high winds, steep relief, and dispersal by bird and mammal species (Vander Wall 2008, Lorenz et al. 2011) make it likely that some seed dispersal occurs far from seed sources. Persistence of early-successional habitats on the landscape will benefit a diverse assemblage of early-successional obligate species (Swanson et al. 2011), but corresponding decreases in late-successional habitat and complicated intermixes of habitat types would have negative impacts on species that depend on late-successional habitat (McKenzie et al. 2004). With the creation of large high-severity patches and limitations on regeneration caused by shifts in climate, these large patches may change to novel species assemblages or from forest to non-forest (Brubaker 1986, Savage and Mast 2005, Bowman et al. 2013). On the other hand, heterogeneity in topography, soils, and water availability may diversify species composition and successional rates, even within large high-severity patches (Savage et al. 2013). The relative importance of variation in disturbance severity compared to spatial variance in successional processes on landscape pattern is unknown for most ecosystems, and research on how these two processes interact will be key to understanding the cumulative effects of climate change on landscape pattern.

Fire extent and severity, and prevalence of large, homogeneous high-severity patches, will also be affected by future disturbances. More disturbance could move vegetation community and structure to alternative stable states (Westerling et al. 2011, Bowman et al. 2013) or alternatively could restore resilience to landscape structure and function (Larson et al. 2013), depending on how much the severity or frequency of fires departs from those of current and historical fire regimes, and the taxa involved. Subsequent fires may reinforce homogeneous patterns created by past disturbances, or create new, more heterogeneous, patterns. In some mixed-severity or low-severity fire regimes, where fire suppression has increased the horizontal and vertical continuity of fuels (Hessburg et al. 2005), large patches of high severity may facilitate postfire establishment of flammable shrub species (e.g., *Arctostaphylos*, *Ceanothus*) that are likely to re-burn with high severity (Perry et al. 2011) and maintain a new stable state. Conversely, previous fires may limit the spread of future fires (Peterson 2002, Collins et al. 2009, Teske et al. 2012). For example, in the CDF, previous burns and fuel treatments that

include prescribed fire effectively decrease the severity of subsequent fires, and therefore contribute to a heterogeneous burn-severity pattern, even under extreme weather (Lyons-Tinsley and Peterson 2012, Prichard and Kennedy 2012). Little area re-burned in our study area (approximately 2% of the total area burned was burned twice or more during the 25-year study period; Cansler 2011), but in other regions with active wildland fire-use programs, re-burns were less severe (Miller et al. 2012b, Parks et al. 2014) and exhibited more complex spatial patterns (Collins and Stephens 2010, Miller et al. 2012b, Teske 2012).

Spatial scale influences analysis of burn-severity pattern

Remotely sensed burn-severity data, such as those used in this study, are now readily available for much of North America, facilitating analyses at spatial scales from individual fires to regions. Because ecological processes vary across scales (Levin 1992), and contagious disturbances such as fire are constrained by characteristic scales of landscape spatial structure (McKenzie and Kennedy 2011), the associated scales of ecological inferences are bounded. For example, fire-size distributions can be quantified statistically only if the geographic domain is large enough to capture extreme events, but not so large that it encompasses widely different bioclimatic zones (Moritz et al. 2011). Consequently, the scale of the data used can either facilitate or confound inferences. If the scales of design and inference are matched, we suggest that the former will ensue. In analyses such as this one, important design considerations include the minimum fire size cutoff, the ecological variation within vs. among the study areas, and the particular metrics used to quantify the burn-severity pattern. Each of these reflects a different dimension of scale compatibility.

Issues of scale regarding the minimum fire size and the extent of the study area may compromise the ability to detect significant relationships between fire size and burn severity or high-severity patch size. For example, the domain assessed by Dillon et al. (2011), the Inland Northwest ecoregions, includes our study area but is an order of magnitude larger. We found a significant increase in burn severity, whereas they did not. Differences are not due to our use of fire size instead of annual area burned (as they did) as a predictor. If we regress the proportion of annual area burned at high severity as a function of annual area burned, our results remain significant ($P < 0.001$) and have even greater explanatory power ($R^2 = 0.39$). (See Miller et al. [2012b] for an example of how the scale at which area burned data is aggregated—by fire or by year—may influence results.) Therefore, we suggest that the difference between our study and that of Dillon et al. (2011) is because they used a higher minimum fire size cutoff (400 ha, following that of the MTBS program), thereby excluding area burned by smaller, less severe fires, or because their geographic domain included areas where

there were significant positive relationships between burn severity and area burned (i.e., the northern Cascade Range), and areas where the relationship was negative or neutral, or both. If we predict percentage of high severity as a function of area burned using only data from fires > 400 ha ($n = 14$ years) the regression is only marginally significant at $\alpha = 0.05$ and explains less variance ($R^2 = 0.24$, $P = 0.044$). Fires < 400 ha are still ecologically important in the complex terrain of the West, and attributes of fire regimes can vary over relatively small spatial scales (10–100 km), due to the flammability and fire tolerance of the vegetation and steep environmental gradients. All of these examples imply that aggregating models, from individual fires to small regions, must be done with care. Investigating relationships between drivers and severity and spatial pattern of fire using a nested hierarchy of geographical extents may be necessary to identify the appropriate geographic and ecological domain of the relationships.

Although many metrics measure spatial pattern, some may be more sensitive to specific types of changes than others (Cushman et al. 2008), and therefore studies should ideally measure pattern using a variety of metrics. Our study and others have found that metrics that measure patch size or patch interior are more strongly correlated with area burned than finer-scale metrics based on pixel adjacencies (Keane et al. 2008, Haire and McGarigal 2009, Miller et al. 2012a). This may be because they are more sensitive to continuous aggregation of a severity class over distances greater than that of 1 pixel (i.e., matched to the spatial scale of landscape fire dynamics). Metrics based on pixel adjacency may also be more difficult to translate to definitive ecological effects. For example, the same PLADJ could represent all pixels having one neighbor, or a mix of pixels having all four neighbors (possibly within very large patches) and pixels with fewer or no neighbors (i.e., in very small patches). Because many of the ecological effects of the burn-severity pattern, such as seed dispersal, take place over a distance greater than 30 m (1 pixel), we suggest that patch-based metrics and measures of patch interior or distance to edge are not only at the appropriate scale for analysis, but also are more informative to other scientists and to managers.

Management implications

Our results show that severity and spatial attributes of fire regimes in the northern Cascade Range depend on fire size and the climatic variables that are correlated with fire-size distributions. In the Pacific Northwest, the climate associated with greater area burned and larger fire sizes, and therefore associated with the formation of large, high-severity patches, is projected to become more common in the future (Littell et al. 2010, Mote and Salathé 2010). If the statistical relationships found here between climate, fire size, severity, and burn-severity pattern persist under future climate, the landscapes of the future may be very different from the landscape

today. Increases in the size, severity, and homogeneity of the spatial pattern of fires probably would increase the proportion of the landscape in homogeneous early-successional patches, and potentially would decrease beta diversity (sensu Whittaker 1960) and ecological resilience (Holling 1973, van Nes and Scheffer 2005) across the landscape.

These changes in fire regimes and landscape pattern would affect a variety of other ecosystem process, including belowground processes (Neary et al. 1999), hydrology and geomorphology (Swanson 1981), and carbon storage (Meigs et al. 2009, Raymond and McKenzie 2012). Burn severity and burn-severity pattern affect a wide variety of wildlife species (Roberts et al. 2008, Wightman et al. 2010, Dudley et al. 2012, Buchalski et al. 2013). Accordingly, understanding how changes in fire regimes and their impacts on landscape composition and pattern affect these species should be a major focus for future research.

Ideally, maintaining resilient landscapes provides a buffer against abrupt changes in landscape pattern and process in response to rapid climate change. Adaptive management to increase resilience includes mechanical treatments and prescribed fire, both for fuel reduction and to maintain heterogeneous stand structures, designed at spatial scales that match patch-size distributions expected under changing fire regimes (Peterson et al. 2011, Prichard and Kennedy 2012). An alternative to managing to restore historical reference conditions associated with past (e.g., Little Ice Age) climate is using “climate analog reference conditions” (Churchill et al. 2013) that anticipate new disturbance regimes associated with future projections of climate (Littell et al. 2010).

In arid mountain landscapes of the Pacific Northwest (i.e., east of the Cascade crest), spatial heterogeneity of vegetation, patch sizes, and fire severity was widespread historically (Hessburg et al. 2005, 2007). Fire-size distributions reflected characteristic spatial scales of landforms, and therefore the spatial patterns of vegetation (fuels) that provided endogenous controls on all but the most extreme wildfires. We suggest that there may be a spatial hierarchy of endogenous controls that limit fire spread and behavior, ranging from variation within stands, particularly in low- and mixed-severity regimes (Larson and Churchill 2012), at the scale of local topography (Lydersen and North 2012), and at larger scales that reflect the influence of topography on local weather (e.g., diurnal canyon winds) and of variation in stand ages due to past disturbances. In a warming climate, with the larger high-severity patches associated with larger fires that our results suggest, there may be an increase in scale of the “relevant” endogenous controls, with larger-scale controls playing a more important role controlling fire spread and behavior. To promote resilience, one might try to create spatial heterogeneity at the future “relevant” scales under climate change. The problem is that we are uncertain about what this scale is

(although we expect that it will be larger). Therefore, we suggest that endogenous controls on fire might be best supported by maintaining spatial heterogeneity at multiple scales. Aiming for spatial heterogeneity across scales has the further advantage of maintaining options, in the form of landscape variability, in the face of extreme events, a strategy held to be robust, or “anti-fragile” (Taleb 2012) in a wider range of applications that includes protecting against natural hazards or economic failures.

With changes in fire regimes likely to be inevitable, actions to protect specific resources will take advantage of place-specific or scale-specific strategies. For example, increased use of wildfires to achieve resource objectives (van Wagtendonk and Lutz 2007, Hunter et al. 2011, Miller et al. 2012b, North et al. 2012, Haire et al. 2013), as has been implemented in the CWF within North Cascades National Park Complex, may be the only logistically feasible method for maintaining valued resources such as wildlife and water quality. Allowing wildfires to burn for resource benefit may be implemented with the least risk during years with mild climate, which probably would have the added benefits of burning with lower severity and more spatial heterogeneity. Other more specific actions include altering road infrastructure so that it is less vulnerable to erosion, removing nonnative species so that native communities can reestablish after high-severity fire, and incorporating future climate to determine the density, species, and genotypes of planted propagules (Peterson et al. 2011, Raymond et al. 2013).

The extent and scale of the changes in ecological systems expected under climate change present a significant challenge to management. Changes in the size, severity, and frequency of disturbances will not be preventable, and further uncertainty arises from multiple-fire interactions (Teske et al. 2012, Larson et al. 2013, Parks et al. 2014), other disturbance agents such as bark beetles (Hicke et al. 2012), direct stress from climate on vegetation (Anderegg et al. 2013), and changes in biotic interactions (HilleRisLambers et al. 2013). The need to incorporate uncertainty and non-stationarity explicitly in monitoring, recognizing, and adapting to change will be heightened under future climate.

CONCLUSIONS

We draw three main conclusions from this study. First, variation in burn severity within a fire has a more direct influence on the postfire landscape pattern than does fire size alone. Therefore, it is essential to know if observed and predicted increases in area burned translate into changes in the severity and spatial pattern of severity. We found a positive relationship between fire size and burn severity, and between fire size and high-severity patch size and patch interior in the northern Cascade Range. Thus, our second conclusion is that if these statistical relationships persist under future climate, and subse-

quent successional processes and overlapping disturbances do not ameliorate the homogenizing influences of large high-severity fires, then the landscape pattern as a whole may shift to a more homogeneous state, impacting ecosystem functions at both local and large scales. Third, the impact of changing climate and weather on the spatial attributes of fire regimes will be mediated by the strength of local bottom-up controls. In each of the four subsections within the larger study area, the relationship between fire size and both severity and severity pattern differed, reflecting each subsection’s vegetation, fuels, topography, and climate. Local controls may continue to be important in low- and mixed-severity fire regimes, even under climate conducive to large fires. In contrast, the change of controls from local bottom-up factors to top-down factors is likely to be most pronounced in high-severity fire regimes with already weak “bottom-up” local controls.

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SUPPLEMENTAL MATERIAL

Appendix A

Tables and figures showing supplemental methods including (1) study area maps, (2) topographical complexity metrics, (3) burn severity class descriptions, (4) spatial pattern metric equations, and (5) fires included in the study (*Ecological Archives* A024-060-A1).

Appendix B

Tables and figures showing supplemental results including (1) climate regression models, (2) subsection-scale severity pattern statistics and patch distributions, and (3) a sensitivity analysis of the effect of core area distances on regression model results (*Ecological Archives* A024-060-A2).

Research



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Author for correspondence:

Anthony LeRoy Westerling

e-mail: leroy.westerling@icloud.com

Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring

Anthony LeRoy Westerling

Sierra Nevada Research Institute, University of California, Merced, 5200 N. Lake Road, Merced, CA 95343, USA

ALW, 0000-0003-4573-0595

Prior work shows western US forest wildfire activity increased abruptly in the mid-1980s. Large forest wildfires and areas burned in them have continued to increase over recent decades, with most of the increase in lightning-ignited fires. Northern US Rockies forests dominated early increases in wildfire activity, and still contributed 50% of the increase in large fires over the last decade. However, the percentage growth in wildfire activity in Pacific northwestern and southwestern US forests has rapidly increased over the last two decades. Wildfire numbers and burned area are also increasing in non-forest vegetation types. Wildfire activity appears strongly associated with warming and earlier spring snowmelt. Analysis of the drivers of forest wildfire sensitivity to changes in the timing of spring demonstrates that forests at elevations where the historical mean snow-free season ranged between two and four months, with relatively high cumulative warm-season actual evapotranspiration, have been most affected. Increases in large wildfires associated with earlier spring snowmelt scale exponentially with changes in moisture deficit, and moisture deficit changes can explain most of the spatial variability in forest wildfire regime response to the timing of spring.

This article is part of the themed issue 'The interaction of fire and mankind'.

1. Introduction

Beginning in the mid-1980s, forest wildfire activity in western US forests underwent an abrupt and sustained regional increase, with nearly two-thirds of that increase concentrated in forests of the Northern US Rocky Mountains between 42° and 50° N latitude [1]. This change in Northern US Rocky Mountain forest wildfire has been linked to climatic factors such as warmer temperatures, dry summers, below-average winter precipitation or earlier spring snowmelt [1–4]. Analyses of reconstructed paleo wildfire and climate indices have also shown similar associations between widespread Northern US Rockies fire years and warm springs combined with warm and dry summers [5,6]. Warming temperatures increase vapour pressure deficit and evapotranspiration, with effects on wildfire, other disturbance and mortality, and forest productivity [7].

The timing, extent and severity of wildfire in western US forests is strongly influenced by climate: over seasonal to decadal time scales antecedent climate shapes fuel characteristics such as their amount, connectivity and structure, while seasonal to interannual climate variability governs fuel flammability [8–10]. A changing climate consequently alters forest fuels characteristics across multiple time horizons. Species compositions and productivity may change as higher temperatures affect potential growing-season length and changes in precipitation, and temperature-driven changes in evapotranspiration affect the moisture available for growth [8,9]. Flammability may be altered as changing evapotranspiration and precipitation affect the moisture available for wetting fuels. Dynamic interactions between wildfire (as well as other disturbance types) that kill standing trees, and altered climatic conditions for

subsequent germination, recruitment and growth of new vegetation, have the potential to produce abrupt, nonlinear transformations of the landscape in response to incremental climate changes [11].

Most of the western US is arid with a preponderance of annual precipitation coming in winter [12]. Much of the forest area is concentrated in mountain ranges where orographic effects enhance precipitation amount, and the elevation increases the likelihood for winter precipitation to fall as snow that can accumulate and carry moisture from cool season precipitation into the more arid summer [12]. Projections of future climate in the region indicate a potential for significant trends toward drier conditions and a reduced fraction of precipitation coming as snow [13,14]. Consequently, it is natural to hypothesize that an early consequence of a warming climate might be a rapid acceleration of wildfire activity in forests at elevations where snow plays an important role in the hydrology, but the climate is warm enough that a modest temperature increase could significantly shift the timing of the spring snowmelt. In 2006, Westerling *et al.* [1] found early indications that this might well be the case, attributing a majority of the regional increase in large forest fire frequency through 2003 to fires burning primarily in mid-elevation Northern US Rocky Mountains forests in early snowmelt years.

The work presented here seeks to answer several questions: has forest wildfire activity across the region continued to increase? Are there new 'hot spots' of increasing fire activity in the region's forests as temperatures have continued to increase? Has recent wildfire activity been associated with the same patterns of temperature and spring snowmelt timing? Can we explain spatial variation in the response of wildfire activity to recent climate trends within the regions' forests? Is fire activity changing in other vegetation types, and if so, is it also associated with the timing of spring snowmelt?

To address these questions, we update the 2006 analysis of Westerling *et al.* [1], extend it to consider the spatial variation in wildfire response to the timing of spring within western US forests, and take a preliminary look at wildfire trends in non-forest vegetation.

2. Data and methods

(a) Fire history

Fire histories for the US National Park Service (NPS), Bureau of Indian Affairs (BIA) and Bureau of Land Management (BLM) were obtained from the US Department of Interior (http://fam.nwgc.gov/fam-web/weatherfirecd/fire_files.htm) and for the US Forest Service (USFS) from the US Department of Agriculture (<http://fam.nwgc.gov/fam-web/kcfast/mnmenu.htm>), and used to update and extend Westerling *et al.*'s fire history [1,4]. We use the same methodology here to create a comprehensive history of actively suppressed wildfires greater than 400 ha reported burning in all vegetation types by USFS, NPS and BIA for 1970–2012, as well as by BLM for 1980 through 2012. USFS, NPS and BIA manage over 70% of the forest area in the western United States, and more than 80% of the forest area over 1370 m elevation (estimates derived from the federal lands and forest area datasets described below). BLM manages mostly rangelands, as well as an additional approximately

4% of western US forest area, and while shorter, the available record allows for more spatially comprehensive comparisons across wildfires in different vegetation types.

About 99% of recorded fires in our dataset are less than 400 ha in size, but comprise only about 25% of total burned area. Documentary records for less than 400 ha fires frequently have missing data fields and erroneous or incomplete location data. Fires greater than 400 ha comprise most burned area, while the smaller number of records facilitates quality assurance; larger fires also usually have better data quality (probably because their costs bring greater scrutiny) [1]. A small number of extremely large wildfires dominate the burned-area record, and multiple management entities often produce conflicting, duplicate records for suppression on the same fire. Fire records were compared using simple algorithms linking date, name, approximate size and location, and obvious duplications and errors were corrected [1,4]. Individual records corresponding to very large fires and fire complexes were compared with archived daily situation reports from the National Interagency Fire Center (<http://www.nifc.gov>), diverse media reports, and post-fire rehabilitation studies to help identify duplications and errors in the records, with particular attention to size and vegetation types.

(b) Land surface characteristics and forest masks

Gridded topographic information derived from the GTOPO30 Global 30 Arc Second (approx. 1 km) Elevation Dataset (elevation, slope, aspect) and coarse vegetation types using the University of Maryland vegetation classification scheme were accessed online from the North American Land Data Assimilation System (LDAS) (<http://ldas.gsfc.nasa.gov>) [15]. These data were combined with GIS layers of federal and tribal land ownership (accessed online from the US National Atlas http://nationalmap.gov/small_scale/atlasftp.html) using spatial packages in R (<https://cran.r-project.org>) to create masks of federal ownership, and gridded elevation and forest fraction for the western US on a 1/8 degree longitude/latitude grid. Masks for six western US forest areas managed collectively by BIA, NPS and USFS were created based on the methodology in [1] for: the Northern Rockies between 42° and 49° N latitude (NR); the Southern Rockies below 42° N (SR); mountain ranges of Arizona and New Mexico excluding the Southern Rockies (SW); the mountains of coastal southern and central California (SC); the Sierra Nevada and southern Cascades and Coast Ranges (SN); and the Cascades and Coast Ranges above 43.1° N (NW). SC fires burning predominantly forest vegetation were very few, with no statistically significant trends detectable. The largest SC wildfires tend to be wind-driven autumn fires that primarily burn in chaparral, but which can also burn significant forest area. Consequently, the small SC forest area is excluded in most subsequent analyses.

(c) Gridded hydroclimatic records

Gridded daily climate data derived from historical station observations using the index station method [16] for 1915–2012 were obtained from the University of Washington National Hydrologic Prediction System (NHPS) (<http://www.hydro.washington.edu/forecast/westwide/>). NHPS data did not incorporate all potentially available stations but were updated monthly, providing up-to-date time series using stations with high-quality records. We also

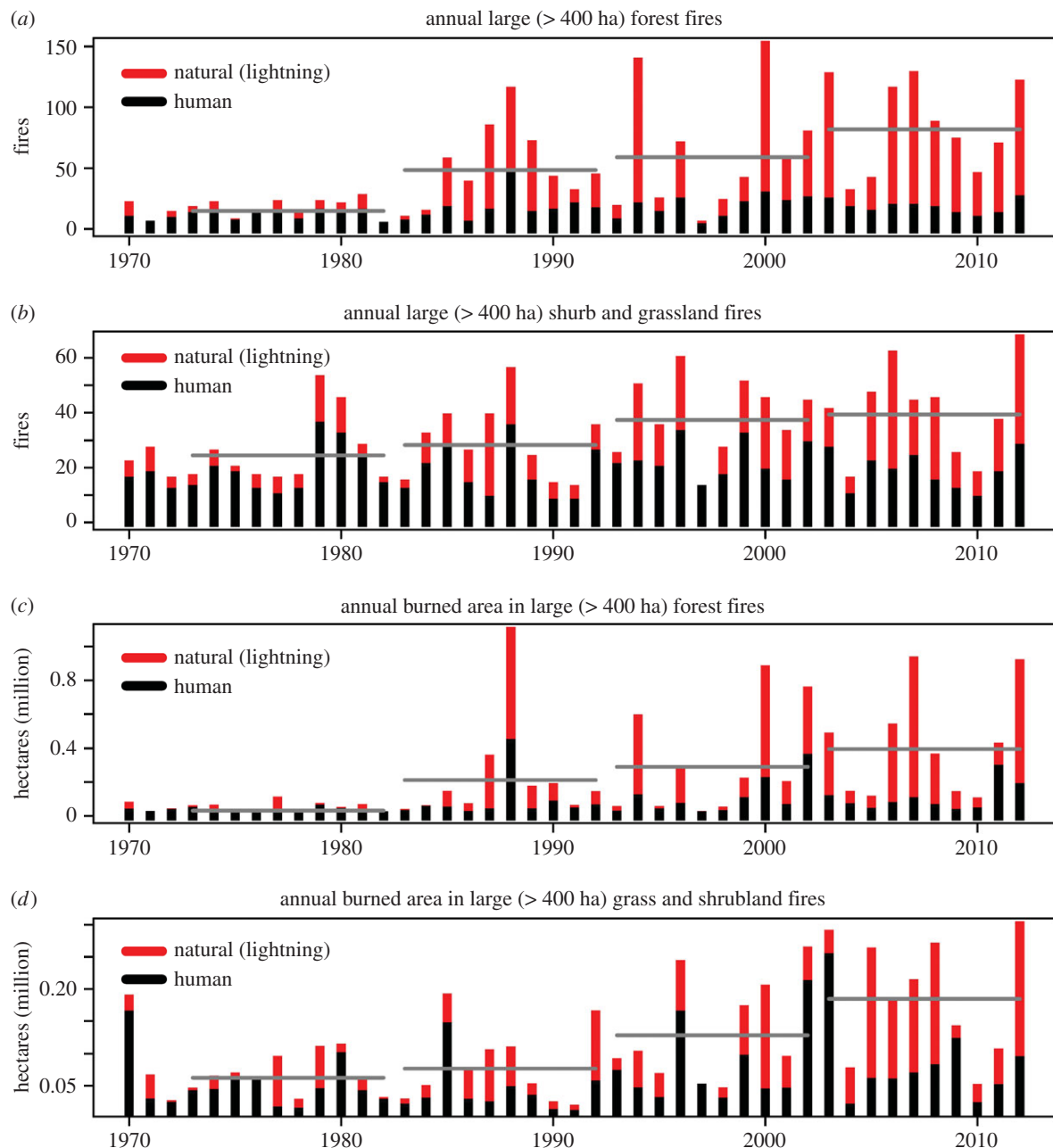


Figure 1. Human and lightning-ignited annual large forest fires, (a), grass and shrubland fires (b), forest burned area (c), and grass and shrub burned area (d), on Forest Service, Park Service and Indian Lands in the western US. Horizontal lines indicate decadal averages.

obtained from NHPS actual evapotranspiration (AET) and snow-water equivalent (SWE) simulated with the variable infiltration capacity (VIC) hydrologic model [17] at a daily time step in water balance mode forced with the daily climate data, LDAS vegetation and topography, and climatological winds. Potential evapotranspiration (PET) was estimated using the Penman–Montieth equation with the same forcing data, and used with AET to calculate moisture deficit ($D = PET - AET$) [1,18,19]. D was then aggregated to monthly cumulative values.

(d) Trend analysis of wildfire frequency and burned area by coarse vegetation type

Number and burned area of large forest wildfires on BIA, NPS and USFS lands were aggregated annually by coarse vegetation type ('forest' versus 'non-forest') derived from documentary fire history data. Annual large-fire frequency

and burned area time series were plotted by coarse vegetation type and reported ignition source (human- versus lightning-caused ignitions; figure 1). Decadal averages were calculated (figure 1) and pairwise comparisons using two-sided Mann–Whitney [20] tests of the null hypothesis that each subsequent decade's fire frequency and burned area distributions have the same mean as for 1973–1982 (tables 1–2). Trends were fit to annual burned area time series for forest and non-forest fires using linear regression techniques with the `glm()` function in R.

(e) Generalized Pareto log-fire size distributions

Generalized Pareto distributions (GPDs) [21] characterize the distribution of exceedances over a threshold. Here we binned individual fire records in the combined BIA, NPS USFS fire history by decade and coarse vegetation type, and fit GPDs to the logarithm of fire sizes exceeding a 400 ha threshold

Table 1. Per cent change in wildfires over 1973–1982 average.^a

	1983–1992	1993–2002	2003–2012
five forest areas	+259% (0.014)	+361% (0.013)	+556% (<0.001)
Northern Rockies	+532% (0.006)	+589% (0.087)	+889% (<0.001)
Northwest	+200% (0.046)	+514% (0.050)	+1000% (0.001)
Sierra Nevada	+219% (0.094)	+184% (0.110)	+274% (0.008)
Southwest	+71% (0.618)	+221% (0.040)	+462% (<0.001)
Southern Rockies	–6% (0.449)	+306% (0.180)	+256% (<0.001)

^a*p*-values for two-sided Mann–Whitney test in parentheses.**Table 2.** Per cent change in burnt area over 1973–1982 average.^a

	1983–1992	1993–2002	2003–2012
five forest areas	+640% (0.015)	+911% (0.035)	+1271% (<0.001)
Northern Rockies	+2093% (0.005)	+1784% (0.104)	+2966% (0.002)
Northwest	+428% (0.034)	+2149% (0.061)	+4979% (0.001)
Sierra Nevada	+270% (0.307)	+492% (0.161)	+324% (0.011)
Southwest	+42% (0.791)	+668% (0.046)	+1266% (0.004)
Southern Rockies	–31% (0.405)	+659% (0.054)	+331% (0.002)

^a*p*-values for two-sided Mann–Whitney test in parentheses.

using the `gpd.fit()` maximum-likelihood fitting function in the `ismev` library in R (figure 2).

(f) Regional spring and summer temperature index

Consistent with [1], a regional spring and summer temperature index was derived from temperature data accessed from the National Oceanic and Atmospheric Administration Drd964x Climate Division temperature dataset ([22]; <ftp://ftp.ncdc.noaa.gov/pub/data/cirs/drd/divisional.README>). Mean monthly temperature values (1970–2012) for 110 western US Climate Divisions for March through August of each year were averaged to produce an annual regional index of spring and summer temperature, which was compared with the annual large forest fire frequency on BIA, NPS and USFS lands (figure 3).

(g) Snowmelt timing

For the timing of the spring snowmelt 1970–2002, Westerling *et al.* [1] used the first principal component of the dates of the centre of mass of annual flow (CT1) for 240 snowmelt-dominated streamflow gauge records provided by the US Geological Survey (USGS) Hydro-Climatic Data Network and by Environment Canada [23–25]. We updated this index for 1970–2012, using a subset of stations provided by USGS. Both Westerling *et al.*'s [1] index and our updated reconstruction are presented here for comparison (figure 3). Missing values for each station were replaced with the 1970–2002 mean for that station. CT1 accounts for one-fifth of total variance in CT and is essentially the annual average CT value for western US stations, with a coherent regional signal in snowmelt timing [1]. The 14 earliest and latest

snowmelt years were extracted from the 1973–2012 record portion for comparison of wildfire and climate covariability with spring snowmelt timing (figure 3).

(h) Fire season length and fire burn time

Documentary wildfire discovery dates and control dates were converted to Julian day of the year using the `julian()` function in R (<https://cran.r-project.org>). The time between first discovery and last control of a large fire in each year was used to proxy for fire season length [1] (figure 3). The time between discovery and control for each large fire is assumed to be indicative of the period of its active spread, when wildfire activity is probably affected by climatological and meteorological factors.

(i) Snowmelt timing tercile analysis of burned area by coarse vegetation type

For a tercile analysis of burned area, wildfire area burned was summed to produce total annual burned area by year and coarse vegetation type (forest or non-forest)—using discovery year and fire type from the documentary fire history—for the common 1973–2012 record from the combined USFS, NPS and BIA fire histories, and for the 1980–2012 record for BLM. Box and whisker plots of annual burned area time series sub-setted by snowmelt timing tercile, coarse vegetation type and fire history were created in R using the `boxplot()` function. Sub-setted time series were compared using the Kruskal–Wallis rank sum test of the null hypothesis that the location parameters of all the sub-setted time series were the same, and multiple pairwise comparisons

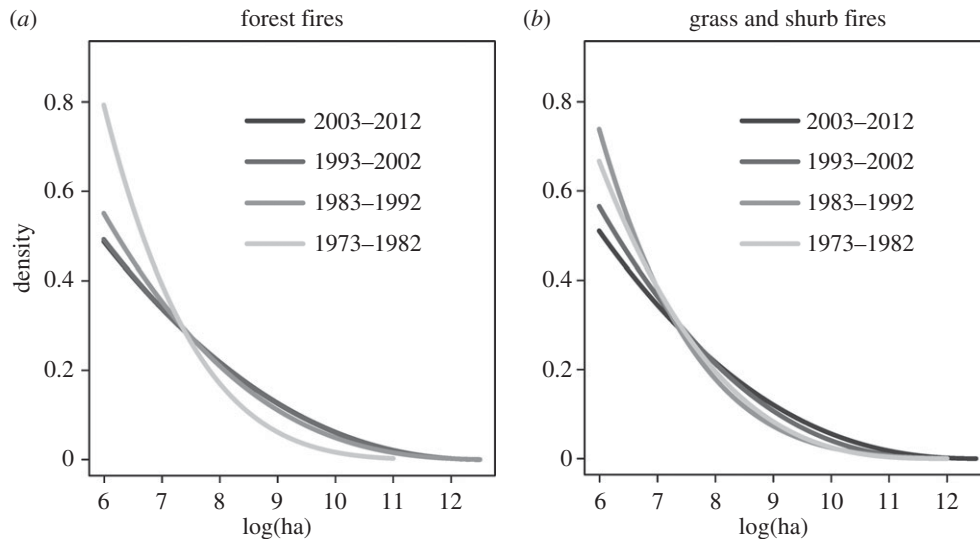


Figure 2. Generalized Pareto distributions fit to log(fire size), by decade for forest fires (a), and grass and shrub fires (b).

using Dunn's z-test statistic for pairwise differences in median using `dunn.test` package in R [26].

(j) Gridded forest-area weighted moisture deficit

Gridded forest-area weighted moisture deficit ($A\delta$) integrates both the intensity of drying and the forest area affected by drying associated with a shift from late to early springs. $A\delta$ was calculated for 300 m elevation bands in each gridded forest area (figure 5) as follows:

$$A\delta_{F,E} = \sum_{g(F,E)} a\delta_g, \text{ where } \delta = \frac{D^{\text{early}} - D^{\text{late}}}{D^{\text{early}} + D^{\text{late}}},$$

a = fraction of grid cell in forest vegetation; D^{early} = average of 14 early snowmelt years' cumulative water year moisture deficit; D^{late} = average of 14 late snowmelt years' cumulative water year moisture deficit; F = Forest {NW, NR, SN, SR, SC, SW}; E = Elevation in 300 m (1000 ft) bands, $g(F, E)$ = 1/8 degree longitude $\times$ latitude grid cells per forest and elevation band.

Elevation bands in each forest area for which the summed forest fractions in aggregate comprised less than one 1/8 degree grid cell in area were excluded from analysis. Forest areas in grid cells above 2895 m were also excluded from analysis, as a rough proxy for the regional upper treeline elevation.

(k) Determinants of cumulative forest-area weighted moisture deficit sensitivity to snowmelt timing

Average monthly SWE was derived from daily cumulative SWE simulated by VIC on a 1/8 degree grid, and intersected with the forest masks and elevation in R to produce a time series of monthly average SWE for the grid cells in each forest area and elevation band. Monthly average SWE < 1 mm was defined as 'snow-free' conditions. Over the grid cells in each combination of forest area and elevation band, the mean snow-free season length in months (SFI) was calculated for 1950–1999. Similarly, the average April through August cumulative AET was calculated over the same period for each forest area and elevation band. A scatter plot with point areas scaled to represent the forest-area weighted change in deficit ($A\delta_{F,E}$) for each forest (F) and elevation band (E) was plotted over mean SFI and mean

April–August cumulative AET (figure 5), to show how drying due to changes in snowmelt timing is a function of the length of the snow-free season and moisture limits on evapotranspiration in late spring and summer.

(l) Comparison of fire frequency to cumulative forest-area weighted moisture deficit

The aggregate number of large wildfires for 14 early and 14 late snowmelt years was calculated by forest and elevation as follows. Large wildfire locations were gridded to the nearest centroids of the 1/8 degree grid used here for analysis, and intersected with gridded masks for each of the six forest areas. Because of imprecision in documentary fire locations, large wildfires located within one 1/8 degree grid cell of a forest area were included in the total for that forest area. Elevations were derived from the documentary record for each wildfire, rather than the nearest 1/8 degree grid cell. For each forest area and elevation band, the number of large wildfires was summed by intersecting wildfire discovery year with years of early or late spring snowmelt. The difference between the early and late snowmelt year wildfire totals was graphed for each forest and elevation band versus the corresponding $A\delta_{F,E}$ (figure 5), and a diverse array of linear and nonlinear regression model functional forms were fitted in R using the `glm()` function. The model selected to describe the relationship between changes in wildfire frequency and changes in forest-area weighted moisture deficit related to a shift in the timing of spring had the best Akaike Information Criterion [27] combined with the best visual fit to the data and high R^2 value.

3. Results

(a) Trends in large-fire frequency and burned area

Our update on the Westerling *et al.* [1] analysis finds that the frequency of large forest wildfires has continued to increase, with each decade since the 1970s, showing an increased frequency of large wildfires at a regional scale compared with preceding decades (figure 1, table 1). We find a highly significant trend ($p < 0.0001$) over 1973–2012, equivalent to over 20 additional large fires per decade on USFS, NPS and BIA

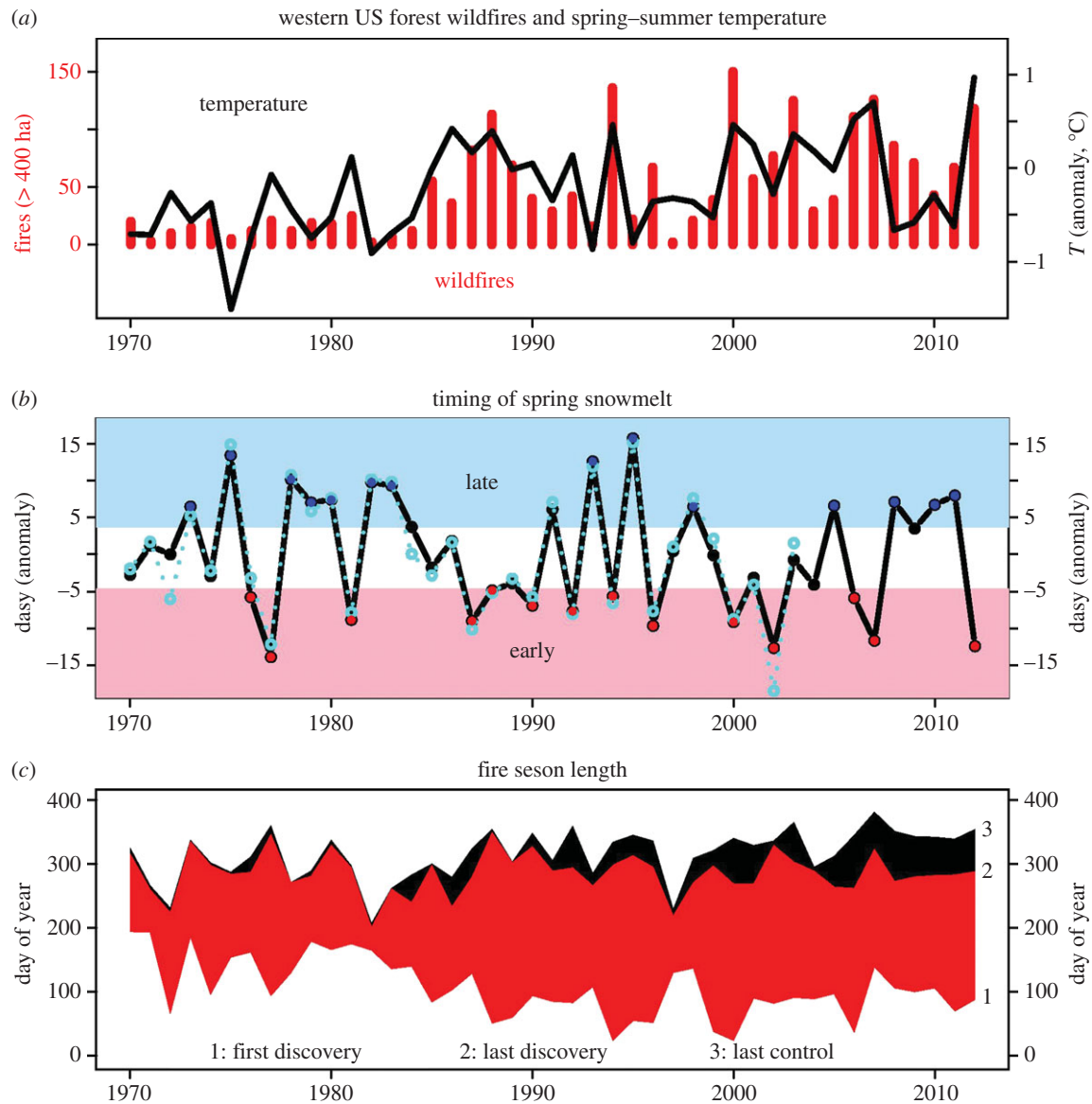


Figure 3. (a) Annual frequency of large (> 400 ha) western US forest wildfires (bars) and mean March through August temperature for the western US (line). Spearman's rank correlation between the two series is 0.69 ($p < 0.001$). (b) First principle component of centre timing of streamflow in snowmelt-dominated streams from Westerling *et al.* [1] (dashed line), and updated through 2012 (solid line). Low (pink shading), middle (no shading) and high (light blue shading) tercile values indicate early, mid- and late timing of spring snowmelt, respectively. (c) Annual time between first and last large-fire discovery and last large-fire control. This figure is an updated version of a previously published figure [1].

lands, or more than 140% of the annual average for the first decade of forest fires (not shown). The area burned in these large fires has also continued to increase (figure 1, table 2), with both a shift in the fire size distribution for fires exceeding 400 ha that was particularly pronounced in the 1980s (figure 2), and the ongoing increase in large wildfire frequency contributing to the overall increase in forest wildfire burned area. The fitted linear trend in forest wildfire burned area was also highly significant ($p = 0.002$, figure 2), equivalent to 123 000 ha per decade since the 1970s, or increasing on average by nearly 390% of the annual average for the first decade in each subsequent decade.

In addition, we find significant increases in wildfire activity in non-forest vegetation types within the same federal land management units (figure 1). The trend in the frequency of large non-forest wildfires is statistically significant ($p = 0.009$), and the trend in the area burned in these large non-forest wildfires is highly significant ($p = 0.0001$),

equivalent to 40 585 ha per decade since the 1970s, or 65% of annual average burned area for the first decade in large non-forest wildfires (figures 1 and 2). Unlike the case of forest wildfires where the rightward shift in fire size distribution was most pronounced in the 1980s, rightward shifts in large non-forest wildfire size distribution have been relatively gradual across the last three decades (figure 2).

A similar analysis with shorter duration BLM fire records produced comparable results. While there was no statistically significant change in the frequency of large BLM fires in non-forest vegetation ($p = 0.73$), there was a statistically significant increase in the frequency of large BLM forest fires ($p = 0.007$). Increases in burned area in large BLM forest fires were highly significant ($p = 0.007$), and increases in burned area in BLM non-forest wildfires were also significant ($p = 0.04$). Percentage increases showed the same pattern as USFS, BIA and NPS fires, with percentage increases in forest wildfires' burned area much larger than for non-forest wildfires.

Table 3. Fire season length and fire burn time by decade.^a

	1973 – 1982	1983 – 1992	1993 – 2002	2003 – 2012
five forest areas				
first discovery	154	150	115	120
last discovery	284	286	281	282
last control	292	316	316	342
season length	138	166	202	222
mean burn time	6	23	40	52
years with no fire	0	1	0	0
Northern Rockies				
first discovery	210	190	196	183
last discovery	250	260	256	263
last control	258	298	310	317
season length	49	107	114	134
mean burn time	7	27	48	59
years with no fire	1	1	2	0
Northwest				
first discovery	206	211	203	298
last discovery	223	243	234	252
last control	229	254	287	315
season length	23	43	84	116
mean burn time	7	13	41	54
years with no fire	5	2	3	1
Sierra Nevada				
first discovery	183	193	185	183
last discovery	241	256	269	268
last control	248	274	294	323
season length	65	81	109	140
mean burn time	8	17	27	49
years with no fire	2	1	1	0
Southwest				
first discovery	177	152	115	125
last discovery	243	220	248	250
last control	249	246	282	307
season length	72	94	167	182
mean burn time	3	20	37	41
years with no fire	2	1	1	0
Southern Rockies				
first discovery	183	175	165	169
last discovery	209	219	231	232
last control	214	227	263	286
season length	31	52	98	117
mean burn time	5	10	27	37
years with no fire	3	6	2	0

^aRounded to the nearest whole day, excluding years with no large fires.

Most of the increase in large wildfires is due to lightning-ignited wildfires (figure 1). Less than 12% of the trend in large forest fires on USFS, NPS and BIA lands is

due to changes in human-ignited wildfires. For non-forest fires, there was no significant trend at all in human-caused fires.

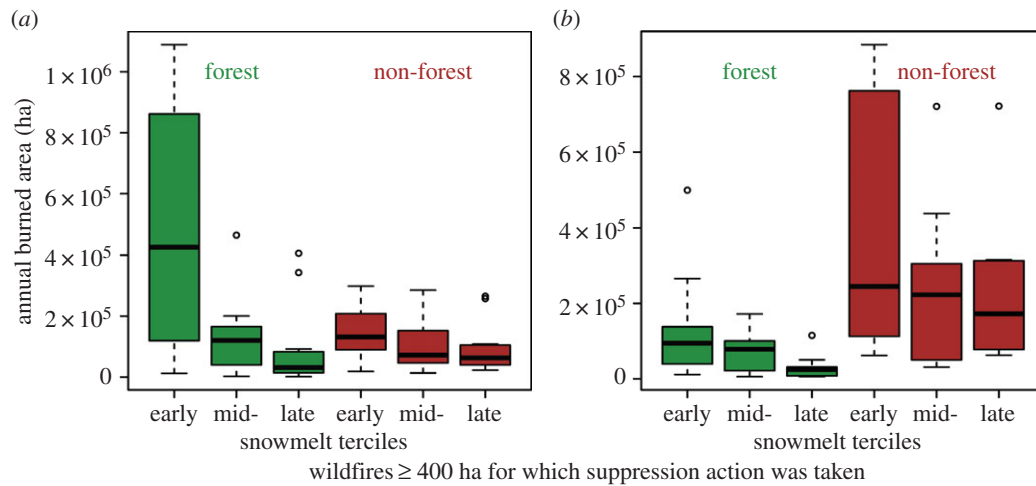


Figure 4. Annual burned area by coarse vegetation type and snowmelt tercile for USFS, NPS and BIA wildfires (1973–2012) (a), and BLM wildfires (1980–2012) (b).

(b) Forest wildfire, temperature and the timing of spring snowmelt

Annual large forest wildfire frequency in USFS, NPS and BIA forests is significantly correlated with spring and summer temperature (Spearman's rank correlation $\rho > 0.7$; figure 3). The largest fires years occur in years with warm spring and summer temperatures and early spring snowmelt dates.

Fire seasons in 2003–2012 averaged more than 84 days longer than in 1973–1982, reflecting a positive trend of just over three days per year since the 1970s (figure 3, table 3). While first discovery dates were over two weeks later on average in 2003–2012 compared with 1993–2002, later control dates more than compensated. This reflects the fact that over the last four decades, the average large wildfire burn time grew from nearly six days in 1973–1982, to nearly 20 days in 1983–1992, nearly 37 days in 1993–2002 and over 50 days in 2003–2012 (table 3).

The earliest third of spring snowmelt years accounts for more than 70% of the area burned in large forest wildfires, and 43% of the area burned in non-forest fires, in the 1970–2012 USFS, NPS and BIA record (figure 4). Early-tercile snowmelt years account for 57% and 50%, respectively, of the burned area for forest and non-forest wildfires in the shorter 1980–2012 BLM record (figure 4). Kruskal–Wallis tests for stochastic dominance were highly significant in each case, indicating that the distribution of at least one tercile deviated significantly from the other two. Dunn's tests indicated that the early- and late-tercile burned areas were significantly different for both forest and non-forest fires in the combined USFS, NPS and BIA fire history, as well as for the forest fires in the BLM fire history. However, the test could not reject the null hypothesis of no difference between early- and late-tercile non-forest BLM wildfire annual burned area. This may be due to the shorter time series available for BLM wildfires, which differentially reduced the sample of late-tercile wildfires more than for early-tercile wildfires, since the terciles were defined for the full 1970–2012 period. The BLM data do confirm, however, the significant effect of spring snowmelt timing on forest wildfire.

(c) Drivers of forest wildfire sensitivity to timing of spring

While the frequency of large forest wildfires regionally is sensitive to timing of spring snowmelt driven by warming

temperatures, analysis of the effects of spring snowmelt timing within western US forests revealed highly diverse responses of forest wildfire. Forests with historic mean snow-free periods of approximately two to four months and high cumulative spring and summer AET have been most sensitive to changes in moisture deficit associated with spring timing (figure 5). Mid-elevation forests in the US Rocky Mountains and the Sierra Nevada have had the largest forest areas with the most drying associated with early spring snowmelt timing compared with late spring snowmelt timing. These areas also show the greatest increases in large-fire frequency from early to late snowmelt seasons (figure 5). The best fit functional form for large forest wildfire frequency response to snowmelt timing was exponential: ($p < 2 \times 10^{-16}$, $R^2 = 0.91$). That is, more than 90% of the spatial variability in the shift in large wildfire frequency between early and late snowmelt years was explained by an exponential function of the change in forest-area weighted moisture deficit.

Note that while SW forests had highly significant trends in large fires and burned area (tables 1 and 2), dry years there with increased fire activity were not significantly associated with the index of spring snowmelt timing (CT1) used here. Unlike Rocky Mountain forests, the largest SW forest fires occurred in both early and late spring snowmelt years. The streamflow CT record is dominated by stations further north in the Rocky Mountains, whereas climate in the Southwest can diverge markedly from the Northwest. For example, 2011 was warm and dry in the Southwest and coincided with an extreme fire season there, but 2011 was a late spring snowmelt year as reflected in CT1, with a north–south climate dipole pattern consistent with a strong La Niña event [28,29].

4. Discussion

Previous studies have suggested forest wildfire activity in the western US is increasing due to a warming climate and earlier spring snowmelt, with Northern Rocky Mountain forests particularly sensitive to these changes [1,2,5,6]. Here, we see that wildfire frequency and burned area in Pacific Northwest forests have increased more rapidly, albeit from a low base, in the most recent two decades (tables 1 and 2). Wildfire

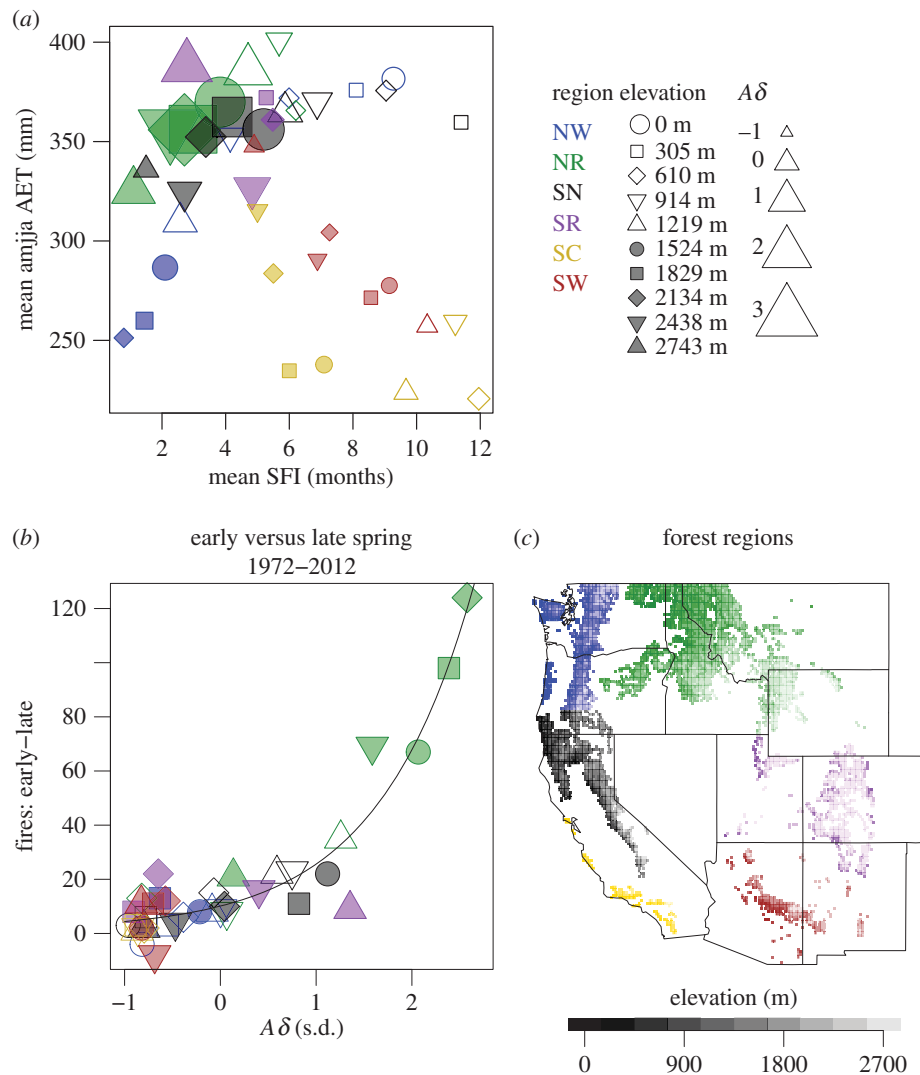


Figure 5. (a) Standardized per cent change in forest-area weighted moisture deficit ($A\delta$) from early versus late snowmelt years by forest area and elevation plotted against mean snow-free season and April–August AET; legend: point colour indicates forest area, shape indicates elevation in 300 m bands and size indicates $A\delta$ in standard deviations; (b) scatter plot of early snowmelt year minus late snowmelt year wildfires versus $A\delta$ with regression fit to $\exp(A\delta)$ (line); (c) map of western US forest area: shading indicates elevation, colour indicates forest region.

activity in other vegetation types may also be increasing (figure 1), and at least for some federally managed lands that increase is strongly associated with the timing of spring snowmelt (figure 4).

Within western US forest areas there is great diversity in the response of wildfire activity to changes in the timing of spring. The most sensitive forests are ones that historically had a mean snow-free season of just two to four months, and high spring and summer cumulative AET. A forest-area weighted moisture deficit index ($A\delta$) that integrates both the intensity of drying and the forest area affected by drying due to shifts in the timing of spring explains most of the spatial variability in changes in large forest wildfire frequency associated with early versus late spring snowmelt. Increases in fire frequency scale exponentially with changes in cumulative water-year deficit ($A\delta$). Given projections for further drying within the region due to human-induced warming, this study underlines the potential for further increases in wildfire activity [7,13,14].

Atmospheric circulation patterns have a broad regional footprint that produces high spatial correlation in temperature anomalies. Thus, early spring snowmelt years still imply warm, early springs in locations that receive little or

no snow. While greater AET in early months may extend the length of the summer drought, locations where AET in the dry season is more constrained by available moisture may not see as much change in cumulative moisture deficit over the full seasonal cycle (figure 5). Thus, sensitivity of large wildfire frequency to timing of spring is greatest in forests where snow is a significant portion of annual precipitation and moisture availability is less of a constraint on AET.

Increasing population and development in proximity to fire-prone lands are sometimes called out as potential drivers of increased wildfire activity. However, the very small contribution of human-caused ignitions to trends in wildfire (figure 1) seems to indicate that these do not play an important role in driving changes in western US wildfire.

Strong trends in Southwestern wildfire that do not appear to be associated with changes in the regional timing of spring index may lend support to the observations and argument that human-induced changes in forest composition, density and structure are particularly important to changes in wildfire in Southwestern forests [30]. At the same time, the start of the Southwestern fire season—as indicated by the date of first large-fire discovery—has shifted more than 50 days earlier since the 1970s, accounting for about one-third of the

increase in the length of the fire season there (table 3). The substantially earlier SW fire season start is consistent with warmer temperatures and earlier spring seasons leading to earlier flammability of fuels in SW forests. However, the spring snowmelt timing index used here (CT1) is dominated by observations recorded in Northern and Central US Rocky Mountain streams, and may not be consistently representative of the timing of spring snowmelt in Southwestern mountain forests.

Changing fire suppression tactics are frequently posited to have contributed to changes in wildfire activity in recent decades. While wildfires used in this analysis were recorded as actively suppressed ‘action’ fires, that does not rule out changing tactics over time altering the effectiveness of suppression and, consequently, the distribution of large

wildfire sizes. Western US wildfire is a coupled human and natural system, and it is reasonable to anticipate that the profound changes observed in climate and wildfire activity over recent decades could elicit changes in wildfire and land management practices that feed back into subsequent wildfire activity, shaping ecosystem sensitivity to further climatic change.

Competing interests. I have no competing interests.

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the timing of spring'

Anthony LeRoy Westerling

Phil. Trans. R. Soc. B **371**, 20150178 (Published 23 May 2016) (doi:10.1098/rstb.2015.0178)

An error was included in table 3. The first discovery date for the Northwest in 2003–2012 should be 198, not 298. The correct table is given below.

Table 3. Fire season length and fire burn time by decade.^a

	1973–1982	1983–1992	1993–2002	2003–2012
five forest areas				
first discovery	154	150	115	120
last discovery	284	286	281	282
last control	292	316	316	342
season length	138	166	202	222
mean burn time	6	23	40	52
years with no fire	0	1	0	0
Northern Rockies				
first discovery	210	190	196	183
last discovery	250	260	256	263
last control	258	298	310	317
season length	49	107	114	134
mean burn time	7	27	48	59
years with no fire	1	1	2	0
Northwest				
first discovery	206	211	203	198
last discovery	223	243	234	252
last control	229	254	287	315
season length	23	43	84	116
mean burn time	7	13	41	54
years with no fire	5	2	3	1
Sierra Nevada				
first discovery	183	193	185	183
last discovery	241	256	269	268
last control	248	274	294	323
season length	65	81	109	140
mean burn time	8	17	27	49
years with no fire	2	1	1	0

(Continued.)

Table 3. (Continued.)

	1973 – 1982	1983 – 1992	1993 – 2002	2003 – 2012
Southwest				
first discovery	177	152	115	125
last discovery	243	220	248	250
last control	249	246	282	307
season length	72	94	167	182
mean burn time	3	20	37	41
years with no fire	2	1	1	0
Southern Rockies				
first discovery	183	175	165	169
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last control	214	227	263	286
season length	31	52	98	117
mean burn time	5	10	27	37
years with no fire	3	6	2	0

^aRounded to the nearest whole day, excluding years with no large fires.

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Interactions of climate, fire, and management in future forests of the Pacific Northwest



Michael C. Wimberly*, Zhihua Liu

Geospatial Sciences Center of Excellence, South Dakota State University, Brookings, SD 57007, United States

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ABSTRACT

A longer, hotter, and drier fire season is projected for the Pacific Northwest under future climate scenarios, and the area burned by wildfires is projected to increase as a result. Fuel treatments are an important management tool in the drier forests of this region where they have been shown to modify fire behavior and fire effects, yet we know relatively little about how treatments will interact with changing climate and expanding human populations to influence fire regimes and ecosystem services over larger area and longer time periods. As a step toward addressing this knowledge gap, this paper synthesizes the recent literature on climate, fire, and forest management in the Pacific Northwest to summarize projected changes and assess how forest management can aid in adapting to future fire regimes and reducing their negative impacts. Increased wildfire under future climates has the potential to affect many ecosystem services, including wildlife habitat, carbon sequestration, and water and air quality. Fuel treatments in dry forest types can reduce fire severity and size, and strategically-placed treatments can help to protect both property and natural resources from wildfire. Although increased rates of burning are projected to reduce carbon stocks across the region, research to date suggests that fuel treatments are unlikely to result in significant increases in carbon storage. Prescribed burning combined with thinning has been demonstrated to be effective at reducing fire severity across a variety of dry forest types, but there is uncertainty about whether changing climate and increasing human encroachment into the wildland–urban interface will limit the use of prescribed fire in the future. Most fire research has focused on the dry forest types, and much less is known about the ecological impacts of increased wildfire activity in the moist forests and the potential for adapting to these changes through forest management. To address these knowledge gaps, future research efforts should build on the Pacific Northwest's legacy of integrated regional assessments to incorporate broad-scale climatic drivers with processes operating at the stand and landscape levels, including vegetation succession, fire spread, treatment effects, and the expansion of human populations into wildland areas. An important outcome of this type of research would be the identification of localized “hot spots” that are most sensitive to future changes, and are where limited resources for fire management should be concentrated.

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1. Introduction

The Pacific Northwest is renowned for diverse and highly productive forests that are a significant component of the terrestrial carbon sink the United States (Raymond and McKenzie, 2012). Fire was historically a ubiquitous landscape-level disturbance across the region that created a continually-shifting mosaic of forest age classes and structures (Agee, 1993). During the latter half of the 20th century, fire suppression has reduced the frequency of wildfires relative to historical fire regimes while timber harvesting and other forest management practices have increased, causing major changes in forest composition, stand structures, and landscape patterns (Hessburg and Agee, 2003; Wimberly et al., 2004).

* Corresponding author. Tel.: +1 605 688 5350.

E-mail address: michael.wimberly@sdstate.edu (M.C. Wimberly).

In the drier parts of the region, these changes have increased fuel loading and connectivity and contributed to the occurrence of uncharacteristically large, high-severity wildfires in recent years (Hessburg et al., 2005). In addition, increased spring temperatures and earlier spring snowmelt have lengthened the fire season across the western U.S., and these changes have been associated with increases in the area burned by large wildfires (Westerling et al., 2006). Projected scenarios of future anthropogenic climate change will further alter fire regimes, with implications for future forest vegetation, habitat for associated species, and forest management practices (Littell et al., 2010; Mote and Salathe, 2010).

Fuel management is an important component of current efforts to restore fire-resilient forest structure and mitigate the negative consequences of wildfires in the dry forests of the Pacific Northwest. It is expected that treatment of hazardous fuels will continue to play a major role in mitigating fire risk and conserving biodiver-

sity in future climates characterized by more wildfires (Spies et al., 2006, 2010). Although fuel treatments can potentially reduce both fire size (Cochrane et al., 2012) and severity (Prichard et al., 2010), they also have substantial impacts on carbon storage, wildlife habitat, and a variety of other ecosystem services (Stephens et al., 2012). These and other studies (e.g., Raymond and Peterson, 2005; Wimberly et al., 2009; Prichard and Kennedy, 2012) have increased our understanding of how fuel treatments and other forest management activities modify wildfire behavior and effects. However, we still know relatively little about how the direct and indirect effects of fuel treatments will affect fire regimes over large areas and long time periods in the context of a changing climate and expanding footprint of human population growth.

The main objective of this review was to explore the potential interactions between climate, fire, and human activities in future forests of the Pacific Northwest. We summarized recent studies documenting projected climate changes and their likely effects on future fire regimes, and reviewed the literature on fuel treatments to characterize current knowledge about their influences on fire behavior and effects and impacts on a variety of ecosystem services. Our synthesis of these topics highlighted some areas of scientific agreement with strong concordance across multiple studies, but also identified specific relationships and geographic areas where significant knowledge gaps exist. We further explored several novel linkages that have not been emphasized in previous assessments of climate change and fire management in the region, including (1) climatic constraints on the application of prescribed fire that could limit its use for fuel treatment and forest restoration activities, and (2) human population growth and expansion of the Wildland–Urban Interface (WUI) that may both increase fire risk and constrain fire management activities.

2. Background

The forests of the Pacific Northwest encompass an enormous range of climatic, physiographic, and floristic variability, but can be broadly classified into two geographic regions: a moist zone located west of the crest of the Cascade mountain range, and a dry zone located east of the Cascade crest and in southwestern Oregon (Franklin and Johnson, 2012). The moist zone has a Mediterranean climate characterized by relatively cool wet winters and warm dry summers with most precipitation falling as rainfall except at the highest elevations. The dry zone has a more continental climate generally characterized by colder winters, hotter summers, and lower precipitation with a higher proportion occurring as snow. The historical vegetation and fire regimes of these zones differed, as do current distributions of forest structure conditions, degrees of fire risk, and forest management practices. Consequently, this geographic stratification is important for understanding future responses to climate change as well as the potential for forest management activities to facilitate adaptation to these changes.

Moist maritime conifer forests dominated by Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) occur in the Oregon and Washington Coast Ranges, the Western Cascades, and in the western portions of the North Cascades (Franklin and Dyrness, 1973), (Fig. 1). Sitka spruce (*Picea sitchensis*) is dominant along the Pacific Coast and other species such as noble fir (*Abies procera*), subalpine fir (*Abies lasiocarpa*) and mountain hemlock (*Tsuga mertensiana*) are common at higher elevations. Historical fire regimes in these moist forests were characterized by large, high severity fires with long return intervals (200–500 years) (Agee, 1993; Long and Whitlock, 2002; Weisberg and Swanson, 2003), although fire regimes in some of the parts of the moist forest zone were mixed severity with shorter return intervals (Weisberg, 2004; Perry et al., 2011). In the modern landscape,

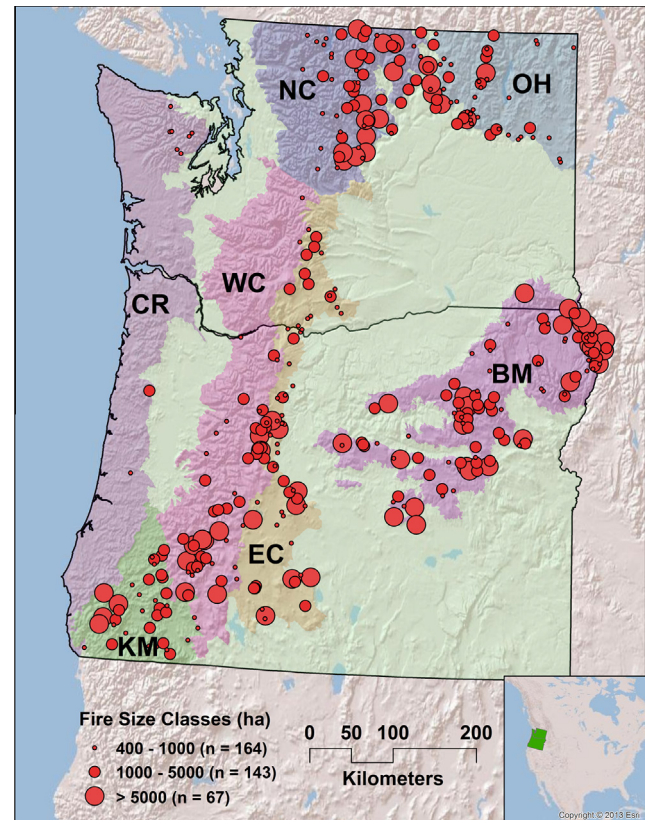


Fig. 1. Map of the Pacific Northwest region encompassing the states of Washington and Oregon along with the Bailey's ecoregion sections referenced in the report. Moist maritime conifer forests occur in the Oregon and Washington Coast Ranges (CR), the Western Cascades (WC), and in the western portion of the North Cascades (NC). Drier forest types predominate in the Eastern Cascades (EC), Okanagon Highlands (OH), Blue Mountains (BM), Klamath Mountains (KM), and the eastern portion of the North Cascades. Circles represent the locations of large historical wildfires (>400 ha) from 1984 to 2008 with size proportional to area burned by the wildfire. Fire data were obtained from the monitoring trends in burn severity (MTBS) project (<http://www.mtbs.gov/>).

most wildfires are suppressed and the disturbance regime is dominated by clearcut timber harvests, with much higher rates of stand-replacement disturbance and smaller areas of older forests dominated by large trees and other late-successional forest characteristics than would be expected under the historical disturbance regime (Wimberly et al., 2000; Kennedy and Spies, 2004; Wimberly and Ohmann, 2004). Other important disturbances in western Oregon and Washington typically result from winter storm events with high winds and precipitation and include flooding, debris flows, landslides, and wind-thrown trees. Their impacts can be substantial; the Columbus Day storm of 1962 blew down or damaged more than 10 billion board feet of timber, with most of the damage concentrated in the Western Cascades (Orr, 1963).

Drier forest types predominate in the Eastern Cascades, Okanagon Highlands, Blue Mountains, Klamath Mountains, and the eastern portion of the North Cascades (Fig. 1). Dry middle- and low-elevation forests encompass a variety of tree species, including Douglas-fir, grand fir (*Abies grandis*), Ponderosa pine (*Pinus ponderosa*) and dry-type lodgepole pine (*Pinus contorta*) (Franklin and Dyrness, 1973). In the Klamath region of southern Oregon, rugged mountainous terrain punctuated with serpentine soils supports diverse mixed coniferous-broadleaf forests. Historical fire regimes were highly variable, but were generally characterized by frequent fires (5–35 year fire return intervals) and mixed severities (Heyerdahl et al., 2001; Agee, 2003). Fire suppression, resulting buildup of fuels, and selective harvest of large individual trees since the early

1900s have resulted in landscapes that deviate from historical conditions and are at risk of high severity fire (Hessburg et al., 2005). In the modern landscape, rates of burning and the proportion of high-severity fire are generally higher in the dry ecoregions than in the moist ecoregions (Figs. 2 and 3). Insect defoliators and bark beetle outbreaks are also important disturbances in the dry forest zone, causing widespread tree damage and mortality and potentially influencing fuel loads and wildfire risk (Parker et al., 2006). However, some studies have found a lack of association between insect outbreaks and subsequent fire severity, suggesting that even severe forest insect outbreaks may not inevitably lead to larger and more severe wildfires (Crickmore, 2011; Black et al., 2013).

3. Climate change and fire ecology

3.1. Projected changes in climate and fire regimes

The future climate of the Pacific Northwest is projected to be substantially warmer than the present based on an analysis of climate simulations from the Intergovernmental Panel on Climate

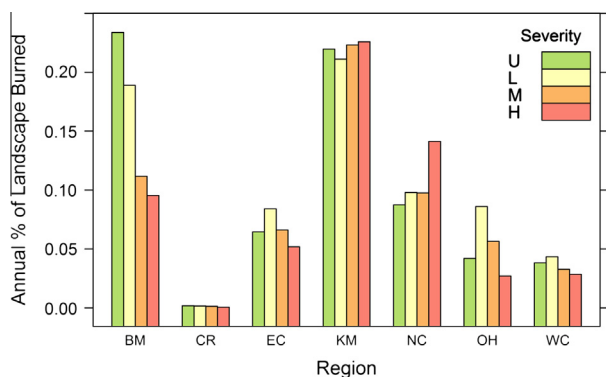


Fig. 2. Area burned by large fires (>400 ha) from 1984 to 2008 expressed as the mean percent of total ecoregion area burned per year by severity class (U = unburned, L = Low, M = Moderate, H = High). Data were obtained from the monitoring trends in burn severity (MTBS) project (<http://www.mtbs.gov/>). Abbreviations are defined in the caption for Fig. 1.

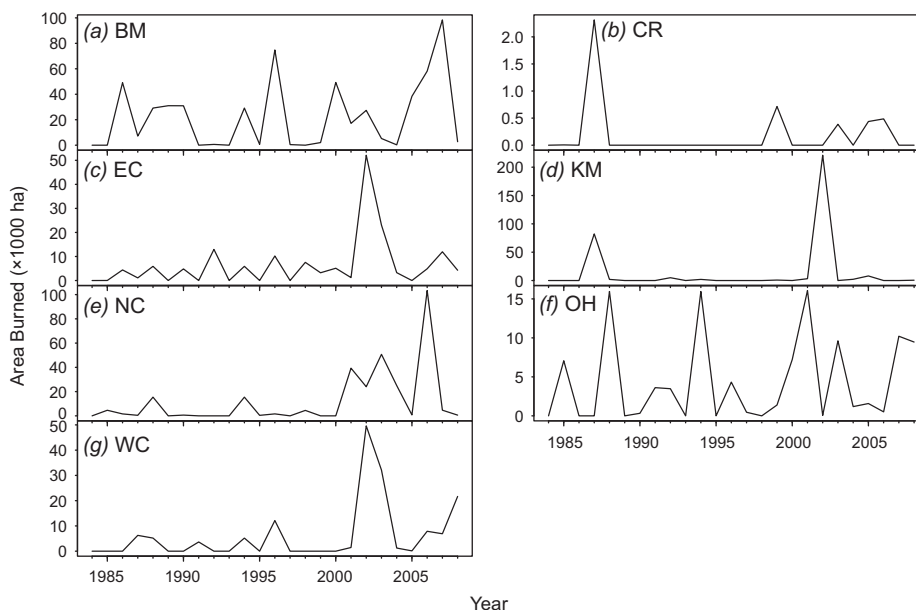


Fig. 3. Area burned by large fires (>400 ha) from 1984 to 2008 expressed as the time series of total ecoregion area burned per year. Data were obtained from the monitoring trends in burn severity (MTBS) project (<http://www.mtbs.gov/>). Abbreviations are defined in the caption for Fig. 1.

Change (IPCC) Fourth Assessment Report (Mote and Salathe, 2010). This assessment encompassed the states of Washington, Oregon, and Idaho as well as portions of Montana and British Columbia and analyzed results from 20 global climate models (GCMs), focusing on A1B as the higher emissions scenario and B1 as a lower emissions scenario. All of the models projected increases in mean annual temperature. The reliability ensemble average (REA) for temperature change by the 2080s was 3.5 °C for the A1B scenario and 2.5 °C for the B1 scenario. Warming was projected to be greatest in the summer months (June, July, and August) under all scenarios. Compared to temperature, projections of annual precipitation change had much higher uncertainty. The REA for precipitation changes through 2100 was close to zero, although individual models projected changes from –10% to +20% by 2080. Precipitation projections were most consistent during the summer months under both A1B and B1 scenarios, with 68–90% of the models projecting precipitation decreases of 20–40%.

Analyses of regional fire scar data from tree ring records have found that historical episodes of widespread, regionally synchronous wildfire tended to occur during dry years that had above-normal temperature in both the spring and early summer (Heyerdahl et al., 2002, 2008). Littell et al. (2009) similarly found that historical interannual variability of burned area in the Cascades ecoregion was associated with summer drought stress as well as lagged temperature and precipitation from preceding years. Projections of fire regime change in the region suggest that burned areas will increase as a result of future hotter and drier climates. Empirical models based on downscaled ECHAM and CMCM3 model projections under both B1 and A1B scenarios projected increased area burned across Washington through the 2080s (Littell et al., 2010). The mean area burned in forested ecosystems was projected to increase by a factor of 3.8 (compared to 1980–2006), with proportional increases expected to be highest in the Western Cascades and Blue Mountains. Similar results were obtained from process-based simulations for the entire region based on the CSIRA, MIROC, and Hadley CM3 models under an A2 scenario, with increases in burned area ranging from 76% to 310% above recent historical amounts (Rogers et al., 2011). All three models projected larger proportional increases in moist forests in the Western Cascades

and Oregon and Washington Coastal Ranges than in dry forests located east of the Cascade crest. The regional variation documented in both studies is consistent with national-level projections of the Keetch–Byram Drought Index (KBDI) using a downscaled Hadley CM3 model projection under the A2 emissions scenario, which showed that increases in KBDI would be largest in the moist coastal forests located west of the Cascade crest (Liu et al., 2013).

In addition to area burned, other important fire regime parameters such as fire size and severity may also be sensitive to climate change. Lower fuel moisture resulting from higher temperature and lower precipitation during the fire season should lead to increased fuel availability and more intense wildfires, which could in turn result in greater tree mortality and higher fire severity. In particular, live fuel moisture is an important factor influencing the potential for crown fire in conifer forests of the Pacific Northwest (Agee et al., 2002). These hypothesized relationships were reflected in a study that found associations between antecedent temperature and precipitation variables and a remotely-sensed classification of burned severity from 1984 to 2006 in the Pacific Northwest as well as other forested regions of the western United States (Dillon et al., 2011). A process-based modeling study projected increased fire severity (measured as biomass consumption by wildfires on a per-area basis) using three different climate models under an A2 emissions scenario, with greater proportional increases in biomass consumption by fire in moist forests than in dry forests (Rogers et al., 2011).

3.2. Ecological impacts of climate-driven changes in fire regimes

Interactions between climate and fire will affect the geographic distributions of major tree species and associated forest types in the Pacific Northwest. The distributions of most woody species are strongly associated with climatic gradients (Ohmann and Spies, 1998), and thus are expected to shift gradually in response to changing climate (Coops and Waring, 2011). For example, contraction of the range of Douglas-fir has been projected in parts of the region because of increases in severe water limitations at the edges of the species' current range (Littell et al., 2010). However, it is likely that changes in forest structure and function, including shifts in tree species composition, will be largely driven by the effects of fire and other large disturbances that kill mature trees and facilitate widespread seedling establishment (McKenzie et al., 2008; Littell et al., 2010). Whereas mature trees can be resistant to small changes in climate, fires catalyze species migrations by facilitating the widespread regeneration of a different suite of species for which the altered climate is more favorable. Species-level differences in the probability of fire-induced mortality (Prichard and Kennedy, 2012) will also contribute to shifts in forest composition resulting from changing fire regimes.

Another ecological consequence of changing fire regimes will be shifts in the age structure of forests. In particular, increases in fire frequency will likely reduce the area of older forests with late-successional characteristics, including large trees, accumulations of standing and down dead wood, and multi-layered canopies. During the latter half of the 20th century, timber harvest was the most important disturbance causing reductions in the area of large-diameter forests (Healey et al., 2008). Since the implementation of the Northwest Forest Plan in 1992, the overall rate of large-diameter forest loss has decreased throughout the entire regime. However, the loss caused by wildfires has increased, particularly on Federal lands in the dry forest zones. As a result of wildfires, the total loss of large-diameter forests in dry forests of the eastern Washington Cascades and the Klamath region in southeastern Oregon was higher after 1992 compared to the preceding two decades. Under future fire regimes with more frequent wildfires, the rate of older forest loss due to disturbance will likely increase and exceed

the rate of which these forests can be replaced through the comparatively slow processes of tree growth and forest succession.

These changes in forest structure also have implications for carbon storage, which is highest in old, undisturbed forests that contain large amounts of carbon in live tree biomass and dead wood pools as well as in understory vegetation and soils (Smithwick et al., 2002). At a stand level, fire results in immediate carbon loss due to combustion as well as longer-term losses resulting from autotrophic respiration as trees and other plants killed by the fire gradually decompose. The immediate carbon loss increases with the severity of the fire, although over time these losses will be offset by vegetation regeneration and growth in the herb, shrub, and tree layers (Meigs et al., 2009). Across broader landscapes consisting of multiple stands, the net ecosystem carbon balance depends on characteristics of the disturbance regime, particularly disturbance return interval and severity, as well as the rate at which recovering forests grow and accumulate carbon (Smithwick et al., 2007). Thus, future C storage in the region will reflect a balance between losses resulting from increased disturbance and the potential for increased productivity that may result from increased precipitation, longer growing seasons, and CO₂ fertilization (Rogers et al., 2011).

Forest growth rates were projected to increase as a result of higher temperatures and relatively constant annual precipitation based on an ensemble of GCM model forecasts and emissions scenarios, with increases ranging from 5% to 8% in moist forests and 15–23% in dry forests (Latta et al., 2010). In the process-based modeling study of Rogers et al. (2011), projections based on the CSIRO and MIROC models resulted in increased ecosystem carbon of 1.7–2.5 kg C m⁻² in moist forests and 22.1–24.7 kg C m⁻² in dry forests even after losses due to wildfire had been taken into account. In contrast, projections based on the relatively hotter and drier Hadley model resulted in decreases in ecosystem carbon of –23.9 kg C m⁻² in moist forests and –1.7 kg C m⁻² in dry forests. An empirical model of fire and carbon dynamics in Washington State based on a 20-model ensemble average of GCM projections also projected larger decreases in carbon storage in moist forests than dry forests (Raymond and McKenzie, 2012). Although net primary productivity was projected to increase in lowland areas of moist forest zone, this increase was overridden by higher wildfire consumption of live and dead woody biomass. These results emphasize that projections of carbon storage under future climate respond to multiple processes, including fire, plant growth, and decomposition, and are highly sensitive to variability in projected climate from different GCM simulations.

4. Fire management activities

4.1. Influences on fire regimes

Fire suppression can reduce the area burned by wildfires below the level that would be expected under a particular climate. However, suppression of fires over the long term in ecosystems with historically high fire frequencies incurs a “fire deficit” as fuels build up in the absence of fire, ultimately contributing to the occurrence of larger and more severe wildfires (Marlon et al., 2012). Successful fire suppression requires periods of moderate weather so that fire crews can complete containment fire lines prior to recurrence of adverse weather conditions. Statistical modeling of over 300 recent fires, with subsequent model testing of over 100 additional recent fires, indicated that the success of suppression effects in limiting daily fire growth depended on the duration of moderate fire behavior and the degree to which containment was already in place (Finney et al., 2009). Because fire behavior is highly dependent on weather, future climates that include more frequent periods of

extreme fire weather may reduce suppression effectiveness. This effect was simulated in the modeling study by Rogers et al. (2011), who found that fire suppression was less effective at reducing burned area and biomass consumed in projected future climate and fire regimes than under present-day conditions.

Forest management activities change the quality, quantity, and spatial patterns of fuels, thereby altering the behavior and effects of wildfires (Agee and Skinner, 2005). In the dry forest zone, treatments that involve prescribed burning, either alone or in combination with overstory thinning, have shown the greatest potential for decreasing both the intensity and severity of wildfire within the treated areas. Tree mortality and crown scorch following the 2006 Tripod complex fires in the East Cascades of Washington State were significantly lower in stands that had been both thinned and burned compared to stands that had only been thinned (but not burned) and control stands that had received no treatment (Prichard et al., 2010; Prichard and Kennedy, 2012). Similarly, tree mortality following the Biscuit Fire in the Klamath region of southwestern Oregon was lower in stands treated with thinning and prescribed burning than in control stands, whereas mortality was higher in stands treated with thinning alone than in the control stands (Raymond and Peterson, 2005). On the School fire in the Blue Mountains of eastern Washington, thinning alone and thinning combined with prescribed burning both reduced fire severity compared to untreated areas, but the reductions were greater for the combined thinning/prescribed burning treatment (Wimberly et al., 2009). Simulation experiments using the FFE-FVS model similarly showed that combined prescribed burning and canopy thinning reduced crown fire risk compared to thinning alone, but also emphasized that reductions in fire severity were greatest in the heaviest thinning treatments (Johnson et al., 2011).

In addition to influencing fire effects within the treated area itself, fuel treatments can also modify fire behavior and affect fire severity at a broader landscape scale. In a simulation modeling study encompassing dry forests in the Blue Mountains, treating 1–2% of the landscape per year was sufficient to reduce average fire spread rate, conditional burn probability, and fire size across the broader landscape (Finney et al., 2007). Another simulation modeling study of fuel treatment effectiveness in the Blue Mountains similarly found that implementing fuel treatments on a relatively small portion of the landscape (10%) resulted in a relatively large overall reduction (70%) of the expected mortality of large trees caused by wildfire (Ager et al., 2010b). A spatial analysis of fuel treatment effects on remote-sensing derived estimates of burn severity highlighted areas of anomalously low burn severity on the School fire that occurred near clusters of stands treated with thinning and prescribed burning, but outside the boundaries of the treatment units (Wimberly et al., 2009). However, these landscape-level effects will vary depending on the specific treatments, their sizes and spatial patterns, and the landscape context. In the dry forests of eastern Oregon, treatments were estimated to have reduced the size of the School fire by 3.6 times the total treated area, whereas treatments on the Otter Creek fire only reduced the size of the fire by one quarter of the total treated area (Cochrane et al., 2012).

The influences of forest management on fire regimes are difficult to generalize because of the wide range of potential management activities; their varied influences on ground, surface, and canopy fuels as well as the understory microenvironment; and the changes in fuels that occur after management as a result of succession, decomposition, and other processes operating over a range of time lags. As a result, some management activities can increase the potential for fire spread and high fire severity. For example, Cochrane et al. (2012) modeled the Kelsay and Boulder fires in the southern Oregon Cascades and found that clearcutting increased fire size relative to control simulations with no

management. Thompson et al. (2007) examined forests that burned initially in 1987 and were burned again by the 2002 Biscuit Fire in southwestern Oregon. They found that salvage logging and planting after the first fire resulted in higher burn severity in the Biscuit Fire compared to unmanaged areas. These results emphasize that efforts to understand the effects of forest management on fire regimes should expand beyond specific fuel treatments to consider the full range of activities that modify forest vegetation, fuels, and landscape structure.

4.2. Impacts on ecosystems and associated tradeoffs

Both wildfire and fuel management have diverse impacts on forest structure and as a result influence a variety of ecosystem services, creating numerous tradeoffs that must be assessed when developing forest restoration strategies and associated fuel management plans. Fuel treatments, particularly thinning, modify forest structure to achieve fire management goals. For example, one of the Fire and Fire Surrogates study sites located in central-eastern Washington dry forests dominated by ponderosa pine and Douglas-fir included thinning from below, prescribed burning, and thinning from below combined with prescribed burning as treatments (Harrod et al., 2007, 2009). Thinning reduced stand density of live trees by >60% and reduced canopy bulk density by >50%. In contrast burning alone caused only 8% mortality for all size classes of trees, with little effect on larger size classes or overall stand structure. The thin-and-burn treatment had higher fire-related mortality compared to burning alone.

The changes in stand structure caused by thinning treatments have the potential to increase local habitat suitability for species associated with the more open, low-density forest structure that typically results. The white-headed woodpecker (*Picoides albolarvatus*), for example, is associated with open stands with large live and dead pine trees (Lehmkuhl et al., 2007). However, thinning treatments implemented in the dry forests that create open stand conditions and reduce ladder fuels are likely to also reduce habitat for late-seral associated species such as the northern spotted owl (*Strix occidentalis caurina*) (Everett et al., 1997). A relatively high proportion of dry forests classified as highly suitable habitat for the northern spotted owl is also at risk for wildfire and currently classified as high or moderate priority for fuel treatment, and this risk will increase under projected future climates and associated increases in wildfire frequency (Gaines et al., 2010). Therefore, there is a need to develop management strategies that will preserve late-successional habitats for northern spotted owls and other associated species while reducing the risk of habitat loss to catastrophic wildfires (Spies et al., 2006).

Modeling studies have shown that treatment of only a portion of the landscape can protect late seral habitat across broader areas. In dry forests of the central Oregon Cascades, Ager et al. (2007) simulated spatial patterns of fire and northern spotted owl habitat loss in landscapes with treatments covering 0–50% of the forested area. These treatments, which consisted of thinning combined with removal of surface fuels and prescribed burning, were prioritized to be located adjacent to existing northern spotted owl habitat on the upwind side of simulated wildfires and in stands meeting a minimum stand density index criterion. Fuel treatments of just 20% of the landscape resulted in a 44% decline in the probability of northern spotted owl habitat loss. Likewise, Ager et al. (2010b) applied the Blue Mountains variant of FVS to assess wildfire risk probabilities in a dry forest landscape in eastern Oregon. Under a forest restoration scenario wherein treatments were applied based on level of overstocking, treating just 10% of the landscape resulted in an approximately 70% reduction in the loss of large trees and also a modest reduction in average burn probability around residential structures. In both of these studies, the strategic placement

of the treatments was critical for achieving a large reduction in large tree loss with a relatively small treated area.

The tradeoffs between stand-level impacts of fuel treatments and landscape-level effects on wildfire risk are also important in evaluating the potential for fuel treatments to enhance carbon storage. Mitchell et al. (2009) conducted a simulation experiment using the STANDCARB model and found that fuel treatments incorporating prescribed burning, tree removal, or both reduced fire severity in moist and dry forest types. However, long-term carbon storage was lower in treated than in untreated stands after accounting for the losses in carbon resulting from both treatments and wildfires. For fuel treatments to facilitate carbon sequestration at a landscape to regional level, carbon losses in the treated stands would need to be offset by increased carbon at other locations where fire frequency or severity is reduced through landscape-level treatment effects. Ager et al. (2010a) carried out a modeling study in a watershed in southeastern Oregon and found that even at a landscape scale, carbon loss directly caused by fuel treatments exceeded the carbon gain from reductions in fire severity resulting

from the fuel treatments. Campbell and Ager (2013) conducted a generalized simulation experiment that explored the sensitivity of fuel treatment effects on carbon dynamics to a variety of model parameters. They found that even though treatments reduced burned area, their effects on system-wide, long-term carbon storage were minimal because the areas impacted by wildfires were relatively small and because treatments and fires had similar effects on stand-level carbon dynamics.

The potential effects of fuel treatments and their interactions with wildfire risk on aquatic habitat and water quality remain poorly understood. Fire was historically an important component of the disturbance regimes of riparian forests across much of the Pacific Northwest. Riparian forests maintain high levels of fuel moisture in both live and dead fuels, and therefore are generally expected to have longer fire return intervals and lower fire severities than adjacent hillslopes (Everett et al., 2003; Olson and Agee, 2005). However, riparian areas are not completely resistant to high-severity fires, and the high fuel loads in riparian forests can lead to stand-replacing fires under extreme weather conditions

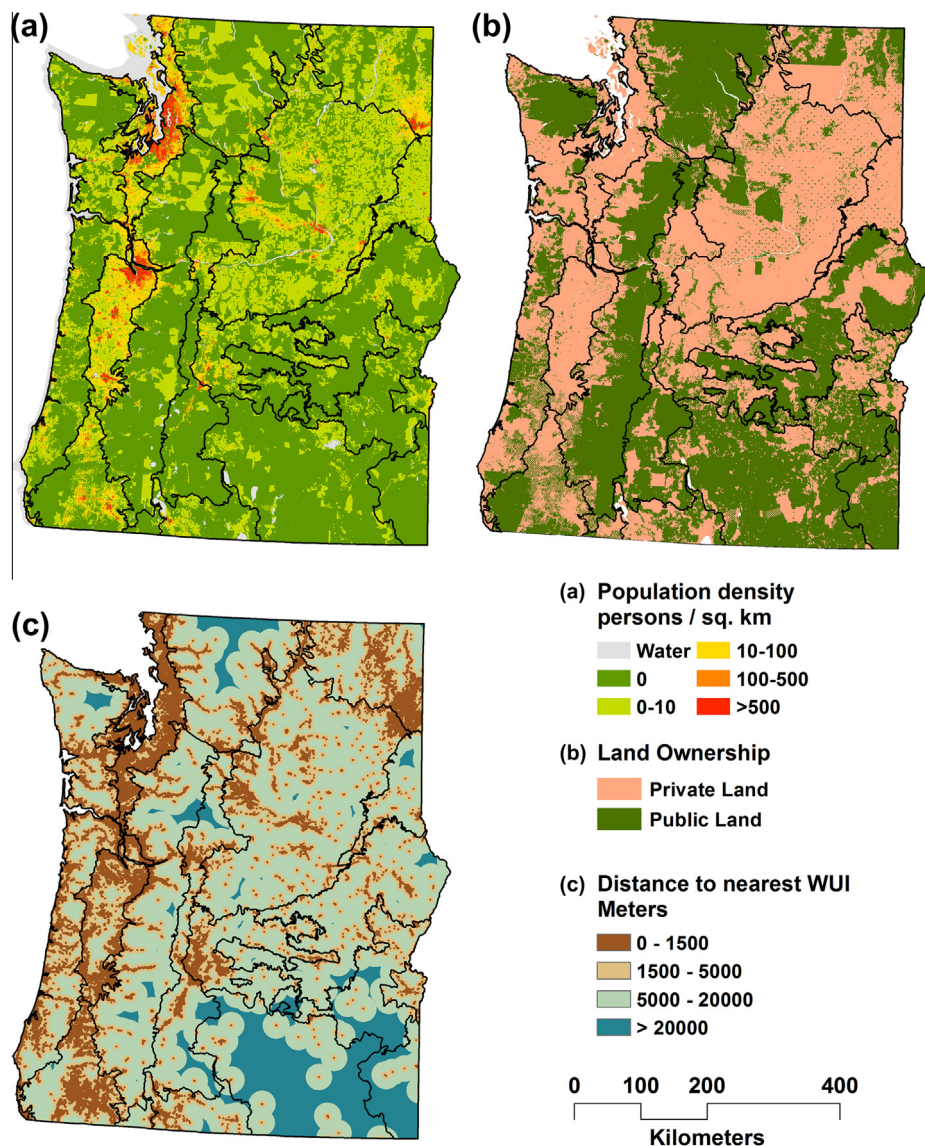


Fig. 4. Maps of human land use in the Pacific Northwest region: (a) population density (persons/km²), (b) public and private land ownership, (c) distance to the Wildland–Urban Interface (WUI, meters). Population density and WUI data (Radeloff et al., 2005) were obtained from Spatial Analysis for Conservation and Sustainability lab at the University of Wisconsin–Madison (<http://silvis.forest.wisc.edu>). Land ownership data were obtained from the U.S. Department of Interior Bureau of Land Management (<http://www.blm.gov/or>).

in both dry and moist forest types (Agee, 1998; Wimberly and Spies, 2001). An important component of the Northwest Forest Plan was the establishment of a system of riparian reserves in which management activities are restricted. In the dry forest zone, however, at least some of the riparian forests were historically characterized by relatively frequent, mixed-severity fires that created a patch mosaic of multi-aged stands dominated by fire-resistant tree species (Messier et al., 2012). As a result, comprehensive fuel reduction and ecosystem restoration activities would require more widespread riparian management activities that are currently being practiced throughout the region. Both wildfires and forest management have the potential to affect rivers and streams by increasing erosion and sedimentation, modifying riparian habitat characteristics, and increasing the risk of other disturbances such as landslides and debris flows (Wondzell, 2001; Wondzell and King, 2003). Although a conceptual model for risk assessment incorporating fire and fuel treatments in riparian areas has been proposed (O'Laughlin, 2005), we currently lack specific knowledge about the effectiveness of treatments in riparian areas and the effects that the treatments themselves may have on ecological processes.

5. Feedbacks and constraints

Climate change will alter wildfire regimes and potentially increase the need for fuel treatments in some areas, particularly in the dry forest zone where late-successional habitat and private property are at risk from wildfire (Spies et al., 2010). However, climate change may also impose greater constraints on the use of prescribed fire, including both planned ignitions and managed wildfires. Prescribed burning must be conducted during suitable burn windows that allow for appropriate fire behavior and effects within a given prescription (Quinn-Davidson and Varner, 2012; Ryan et al., 2013). These burn windows are highly weather dependent, encompassing conditions under which fire intensity is high enough for adequate fuel consumption but low enough to avoid risk of escaped fire and excessive mortality of overstory trees. The effects of prescribed fire on a range of ecological responses,

including plants, wildlife, and soils, can also vary with season of burning (Knapp et al., 2009).

In the Pacific Northwest these burn windows typically occur during in the Spring and Fall, outside of the main wildfire season (Knapp et al., 2009), and prescribed burning may not be possible in a given year if appropriate weather conditions do not occur. For example, prescribed burning in two of six experimental units was delayed for 2 years at the Mission Creek Fire and Fire Surrogates study site in the eastern Washington Cascades because of a lack of appropriate burning conditions (Harrod et al., 2007, 2009). Changes in climate that extend the length of the fire season could either increase or decrease the time during which there is suitable weather for prescribed burning. Although weather and its influence on burn windows is recognized as one of the main constraints on the ability to implement prescribed burns (Quinn-Davidson and Varner, 2012), our ability to quantify the underlying relationships is limited by a lack of comprehensive and consistent historical records of prescribed fire usage (Kolden and Brown, 2010). As a result, the potential effects of future climate on burn windows are unknown and represent a source of uncertainty about the future potential of prescribed burning for fuel treatment and ecological restoration.

Other regional dynamics, such as the expansion of human populations and infrastructure in the wildland–urban interface, will also affect future fire management in the region. The human population of the Pacific Northwest has expanded considerably over the past 30 years, with the total population of Oregon and Washington increasing from 6.8 million in 1980 to 10.6 million in 2010. Most of this population is concentrated in large metropolitan areas located west of the Cascade crest in the moist forest zone (Fig. 4a), and much of the forested land within the region is distributed in large blocks of public ownership that are distant from major population centers and protected from most types of development (Fig. 4b). However, there are also many smaller pockets of low-density development dispersed across most of the region, and as a result much of the forested land is located within several kilometers of the WUI (Fig. 4c). Although the percentage of human-impacted area within the most of the forested sub-re-

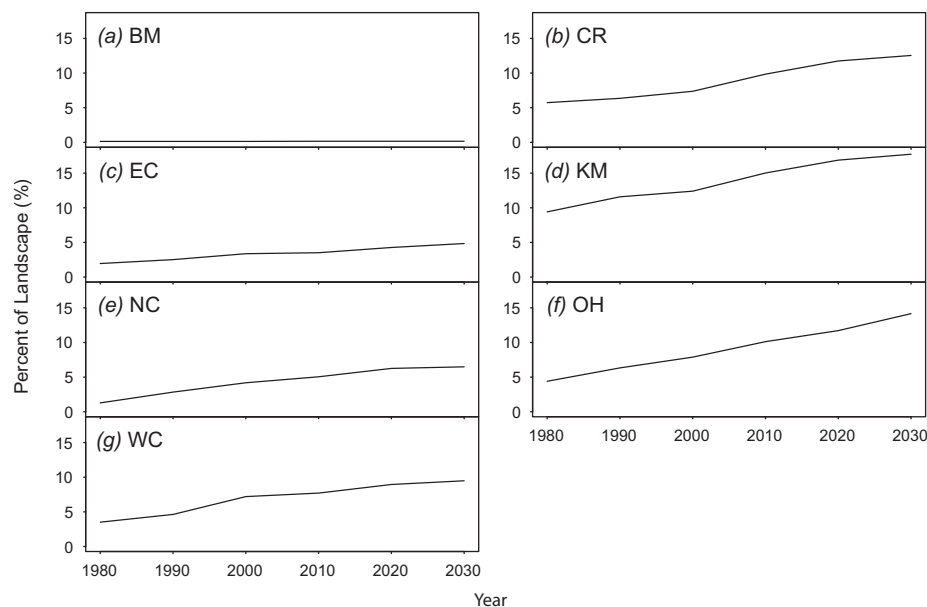


Fig. 5. Historical and projected trends of the percentage of human-occupied land, defined as at least one house per 16 ha (40 acres), within each Bailey's ecoregion section (Radeloff et al., 2010). Data were obtained from Spatial Analysis for Conservation and Sustainability lab at the University of Wisconsin–Madison (<http://silvis.forest.wisc.edu>). Abbreviations are defined in the caption for Fig. 1.

gions remains relatively low, these percentages have doubled within some of the sub-regions over the past 30 years and these trends are projected to continue into the future (Fig. 5). Future population and WUI growth is expected to be spatially heterogeneous, with most of the expansion occurring in the vicinity of large and medium-sized cities in both the moist and dry forest zones (Kline et al., 2003, 2010; Hammer et al., 2007).

The expanding human population and accompanying land use changes will interact with wildfire and fire management in a variety of ways. The necessity of protecting increasing densities of structures within the WUI will increase the cost and complexity of fire suppression (Gebert et al., 2007). The potential for using prescribed fire in fuel reduction and ecological restoration treatments will also be affected, because burn windows are highly sensitive to the surrounding physical and social landscape in addition to the climatic constraints discussed previously (Quinn-Davidson and Varner, 2012). For example, the range of weather conditions under which a burn is feasible may become narrower in situations where heavy fuel and rugged terrain increase the potential for extreme fire behavior, or where a dense human population in the surrounding landscape increases concerns about smoke or escaped fire. Furthermore, the costs of prescribed burning are higher in the WUI than in other areas (Berry and Hessel, 2004; Berry et al., 2006). Alternative methods of surface fuel treatment, including piling, mastication, and compaction of surface fuels, could be applied in conjunction with overstory thinning to enhance fuel treatment efficiency. However, as with prescribed burning, the costs of mechanical treatments are higher in the WUI than in other areas (Berry and Hessel, 2004), and to date there has been little research on their effectiveness in forests of the Pacific Northwest.

The interaction of changing climate and fire regimes with increasing and expanding human populations also has the potential to result in new management challenges for the region. For example, the health impacts of exposure to particulate matter from wildfire smoke, including increased levels of asthma symptoms, hospital admissions, and other respiratory health outcomes have been well documented (Bowman and Johnston, 2005). Future aerosol emissions are projected to increase as a result of larger burned areas in the western United States, with particularly high concentrations in the Northwest as a result of higher biomass consumption (Spracklen et al., 2009). Raymond and McKenzie (2012) projected based upon their models that much of this increased biomass consumption will be in moist forests west of the Cascade crest. Because the majority of the human population is also concentrated in this region, wildfire smoke may evolve into a more significant public health issue than it has been in the past.

6. Summary and conclusions

Studies based on both empirical and process-based models using multiple GCMs and emissions scenarios have projected that wildfires will occur more frequently and burn larger areas under projected future, warmer climates in the Pacific Northwest (McKenzie et al., 2004; Littell et al., 2010; Rogers et al., 2011). Increased fire occurrence has the potential to impact multiple ecosystem services including wildlife habitat, carbon sequestration, and water and air quality. Field-based and simulation modeling studies conducted in the dry forest zone have shown that fuel treatments that modify both canopy fuels (thinning) and surface fuels (prescribed burning) can reduce fire severity within the treated areas and can also reduce burn probability and severity across larger landscapes by modifying fire behavior and fire effects in untreated areas. Prescribed burning is a critical component of these treatments, and thinning without accompanying treatment of surface fuels has proved to be less effective than thinning and burning

at reducing the severity of large fires (Agee and Skinner, 2005; Raymond and Peterson, 2005; Wimberly et al., 2009; Prichard et al., 2010; Prichard and Kennedy, 2012). However, there is also considerable uncertainty about the degree to which changing climate and increasing human encroachment into the WUI may increase the cost and complexity of fire suppression and constrain the use of prescribed burning in the future. Increasing both funding and public support for prescribed burning will be critical for sustaining critical ecosystems processes and reducing fire risk in the dry forests of the Pacific Northwest (Ryan et al., 2013).

Fuel treatments themselves also have a variety of scale-dependent positive and negative impacts on a multiple ecosystem services. Fire was historically an important disturbance in many riparian as well as upland forests, but fuel treatment effectiveness in riparian areas and potential impacts on aquatic habitats and species are currently not well understood. Although fuel treatments in late-successional forests will reduce habitat quality for the northern spotted owl and other late-successional species within the treated areas, strategically-placed treatments do have the potential to reduce burned area and prevent the loss of late-successional forests to wildfire at a broader landscape scale (Ager et al., 2007, 2010b). In contrast, fuel treatments have not been demonstrated to increase carbon sequestration at either stand or landscape scales because fires are relatively uncommon, and thus the amount of carbon lost from the treatments has been shown to exceed the gain from reducing burn probability and fire severity even over large landscape and long time periods (Campbell et al., 2011). Because regional carbon stocks are expected to have limited sensitivity to fuel treatments over large areas and long time periods (Campbell and Ager, 2013), fuel management strategies are more appropriately focused on other management goals such as the development of resilient and functional ecosystems and protection of critical natural resources and property from destructive wildfires (Hurteau and Brooks, 2011).

At present, there is considerable uncertainty about the impacts of climate change on wildfire in moist forests and the appropriate management strategies that should be implemented in response. Proportional increases in area burned relative to current levels are projected to be highest in the moist forests (Littell et al., 2010; Rogers et al., 2011), although the overall rate of future burning will still be much higher in dry forests than in moist forests. These projected changes raise the possibility of once again seeing large, high-severity conflagrations such as the Tillamook, Yacolt, and other large fires that occurred during the first half of the 20th century (Gray and Franklin, 1997; Wimberly et al., 2004). The reemergence of large, destructive wildfires in the moist forest region, even on a relatively infrequent basis, could have enormous impacts on timber supply, carbon sequestration, and the conservation of late-successional and old-growth forests. More frequent occurrence of regional climate-associated disturbances such as the Columbus Day windstorm (Orr, 1963) would have similarly large impacts. Because of the relatively low levels of wildfire in the moist forests in recent decades, there has been relatively little research on fire management and fire ecology in contrast to the dry forests. As a result we have limited knowledge of how forest management influences fire regimes in the moist forests, and it is unlikely that fuel management as currently practiced in the dry forest zone would be feasible or desirable given the broader goals for ecological restoration within the moist forest zone (Franklin and Johnson, 2012).

Although the focus of this paper has been on wildfire, other types of disturbance also impact Pacific Northwest forests and will potentially interact with changing climates and wildfire. For example, mountain pine beetle (*Dendroctonus ponderosae*) causes mortality of *P. contorta* throughout the dry forests of the region, and attacks are associated with warm winters and drought conditions

(Preisler et al., 2012). Thus, future warmer and drier climates may lead to increased tree mortality from insects as well as wildfire. Bark beetle outbreaks change forest structure and fuel loads and as a result can affect the behavior, severity, and extent of subsequent wildfires. However, these effects are highly variable and are contingent upon numerous other factors, particularly the time that has elapsed since the outbreak (Hicke et al., 2012; Black et al., 2013). For example, neither bark beetle outbreaks nor defoliating insect activity had a significant influence on fire severity in B&B fire complex, which encompassed a variety of moist and dry forest types in the central Oregon Cascades (Crickmore, 2011). Fuel treatments also interact with insect disturbance, and have been shown to increase mortality from bark beetles relative to untreated controls (Youngblood et al., 2009), but also to reduce mortality from beetles following wildfire (Prichard and Kennedy, 2012). Given the significant acreage at risk for insects and disease outbreaks in the region and its considerable overlap with areas at risk for wildfire and development (Kline et al., 2013), improving our ability to understand and project climate effects on forest insects and diseases and their interactions with fire is a critical research need.

Most analyses and forecasts of fire in relation to climatic variability and climate change have focused on rates of burning across broad regions or physiographic zones (McKenzie et al., 2004; Littell et al., 2010; Rogers et al., 2011). In contrast, less is known about how other aspects of the fire regime such as fire severity and fire size are associated with climatic variability and other landscape-level constraints such as topography, vegetation and fuels. Increased future fire risk will be likely concentrated in specific “hot spots” where physiographic settings and vegetation types that are more conducive to burning intersect the expanding WUI where low-density development puts property and risk and potentially constrains the use of prescribed fire. The Pacific Northwest has a strong tradition of integrated regional assessment focused on forest management practices, fire, and landscape change (Haynes et al., 2001; Barbour et al., 2007; Spies et al., 2007), but to date such assessments have not explored these changes in the context of projected future climate and land use trends. The next generation of integrated assessments will need to incorporate novel data sources and models to more effectively integrate future climate projections with a variety of processes and constraints operating at the stand and landscape levels, including vegetation succession, fire spread, treatment effects, and the expansion of human populations into wildland areas.

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Review and synthesis

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



Paul F. Hessburg^{a,*}, Thomas A. Spies^b, David A. Perry^c, Carl N. Skinner^d, Alan H. Taylor^e, Peter M. Brown^f, Scott L. Stephens^g, Andrew J. Larson^h, Derek J. Churchillⁱ, Nicholas A. Povak^j, Peter H. Singleton^j, Brenda McComb^c, William J. Zielinski^k, Brandon M. Collins^l, R. Brion Salter^j, John J. Keane^m, Jerry F. Franklinⁱ, Greg Riegelⁿ

^a USDA Forest Service, Pacific Northwest Research Station, 1133 North Western Ave., Wenatchee, WA 98801-1229, USA

^b USDA Forest Service, Pacific Northwest Research Station, Corvallis, OR, USA

^c Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR, USA

^d USDA Forest Service, Pacific Southwest Research Station, Redding, CA, USA

^e Department of Geography & Earth & Environmental Systems Institute, The Pennsylvania State University, PA, USA

^f Rocky Mountain Tree Ring Research, 2901 Moore Lane, Fort Collins, CO 80526, USA

^g Department of Environmental Sciences, Policy & Management, University of California, Berkeley, CA, USA

^h College of Forestry and Conservation, University of Montana, Missoula, MT, USA

ⁱ College of Environment, School of Forest Resources, University of Washington, WA, USA

^j USDA Forest Service, Pacific Northwest Research Station, Wenatchee, WA, USA

^k USDA Forest Service, Pacific Southwest Research Station, Arcata, CA, USA

^l USDA Forest Service, Pacific Southwest Research Station, Davis, CA, USA

^m USDA Forest Service, Pacific Southwest Research Station, Fresno, CA, USA

ⁿ USDA Forest Service, Deschutes National Forest, Bend, OR, USA

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ABSTRACT

Increasingly, objectives for forests with moderate- or mixed-severity fire regimes are to restore successional diversity 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

* Corresponding author.

E-mail addresses: phessburg@fs.fed.us (P.F. Hessburg), tspies@fs.fed.us (T.A. Spies), dave_perry38@msn.com (D.A. Perry), rxfuego@gmail.com (C.N. Skinner), aht1@psu.edu (A.H. Taylor), pmb@rmtrr.org (P.M. Brown), sstephens@berkeley.edu (S.L. Stephens), a.larson@umontana.edu (A.J. Larson), derekch@u.washington.edu (D.J. Churchill), npovak@fs.fed.us (N.A. Povak), psingleton@fs.fed.us (P.H. Singleton), Brenda.McComb@oregonstate.edu (B. McComb), bzielinski@fs.fed.us (W.J. Zielinski), bmcollins@fs.fed.us (B.M. Collins), bsalter@fs.fed.us (R.B. Salter), jkeane@fs.fed.us (J.J. Keane), jff@u.washington.edu (J.F. Franklin), griegel@fs.fed.us (G. Riegel).

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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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).

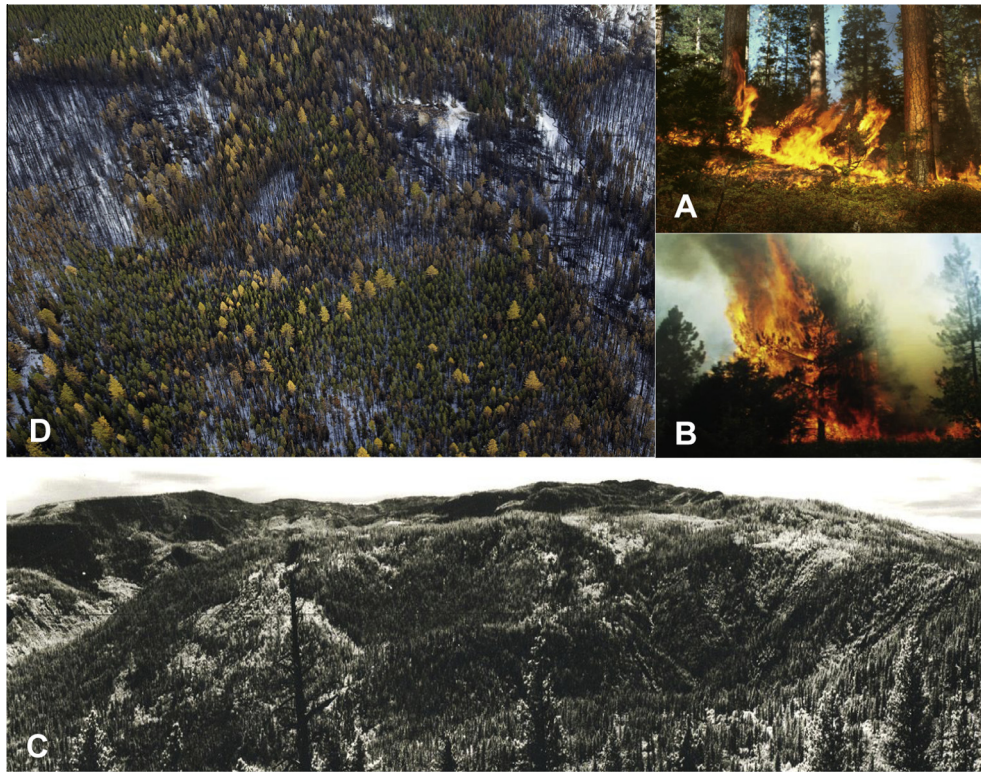


Fig. 2. Photographs of surface (A) and crownfire (B) behavior associated with mixed-severity fires, where 20–70% of the dominant basal area or canopy cover may be killed by the sum of all surface and crownfire effects. Panel (C) shows a typical mixed conifer forest in the eastern Washington Liberty-Beehive 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.

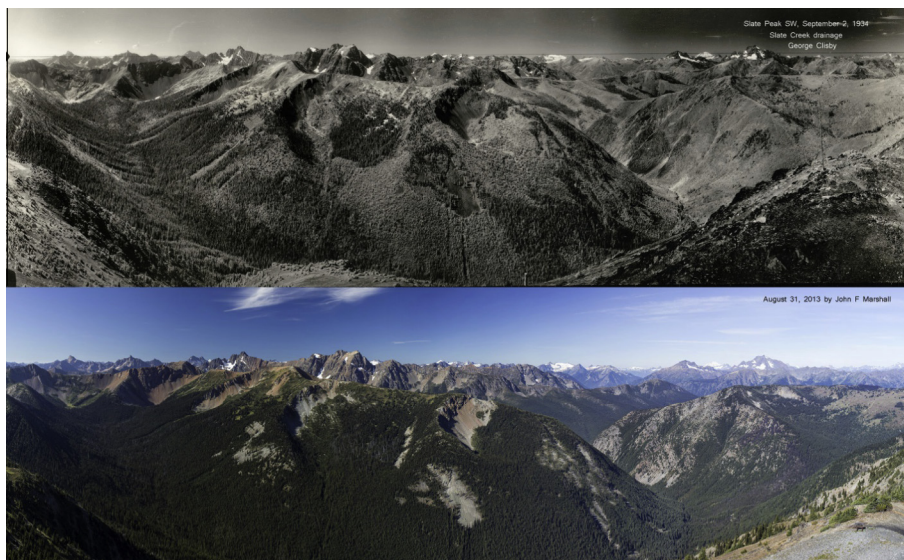


Fig. 4. Repeat panoramic photographs of the Slate Peak area, Slate Creek drainage, near Mazama, WA. In the 1934 black and white photo (above), cold site lodgepole pine, subalpine fir, and Engelmann spruce forests are apparent. These forests were burned by lightning ignited fires until well into the 20th 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 resilience¹ (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; Guarín 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), Haggmann 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 "strategically placed landscape area treatments" 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 defensible fuel profile zones (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 $\geq 50\%$ 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 $\times$ 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).

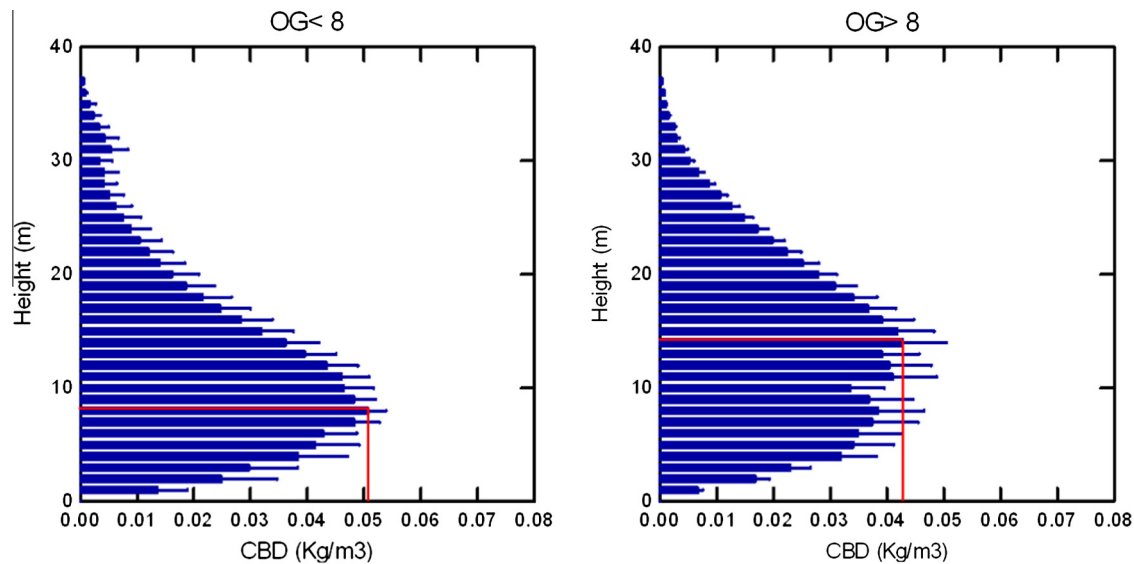


Fig. 7. Canopy bulk density (CBD) profiles as affected by the presence of living large (>80 cm) old growth trees or large stumps, where old growth trees were removed. Old growth (OG) is defined as older than 150 years. OG < 8/> 8 = fewer/more than 8 living large OG trees or stumps per 0.2 ha plot (8 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 large and old, early seral trees (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; Coddling 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 increase 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 reseed-ing 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 Wagtenonk 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.

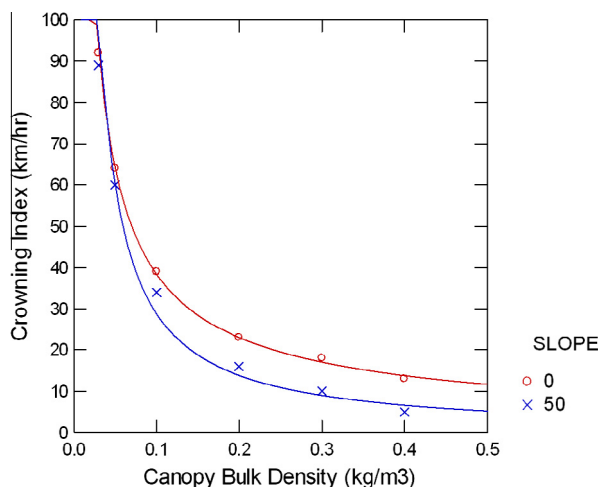


Fig. 8. NEXUS (Scott and Reinhardt, 2001) simulations of Crowning Index (wind speed required to initiate active crownfire) as affected by canopy bulk density and slope percent. Late summer fuel moisture conditions are shown. Canopy bulk densities represent a measured range on the Deschutes National Forest, 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 $<0.3 \text{ kg m}^{-3}$. 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 $\sim 0.15 \text{ kg m}^{-3}$, 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 $\geq 70\%$ canopy closure will crown in winds of approximately $10\text{--}30 \text{ km h}^{-1}$, 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

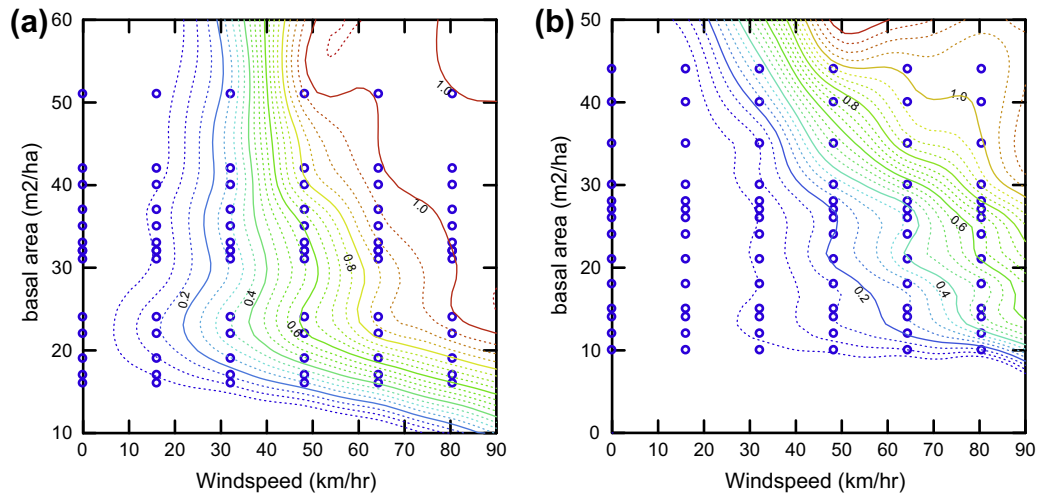


Fig. 9. Canopy fraction burned as a function of stand basal area (BA) and wind speed for two scenarios on the eastern slopes of the Oregon Cascades. This figure (a) is from 13 forest plots with measured crown bulk density (CBD), crown base height (CBH), and BA. This figure (b) is from the same plots 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 $20 \text{ m}^2 \text{ ha}^{-1}$. 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 m² 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 m² ha⁻¹ in the dry mixed-conifer and ponderosa pine zones, and 27–32 m² 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](#)).

[Rolooff 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, [Rolooff 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 brevicornis* 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.

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 reseeded 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

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.

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 $\leq 30\%$ tree canopy cover (sparse coniferous or hardwood) growing on a forest PVT^a; i.e., capable of producing forest cover ($\geq 10\%$ 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).

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

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

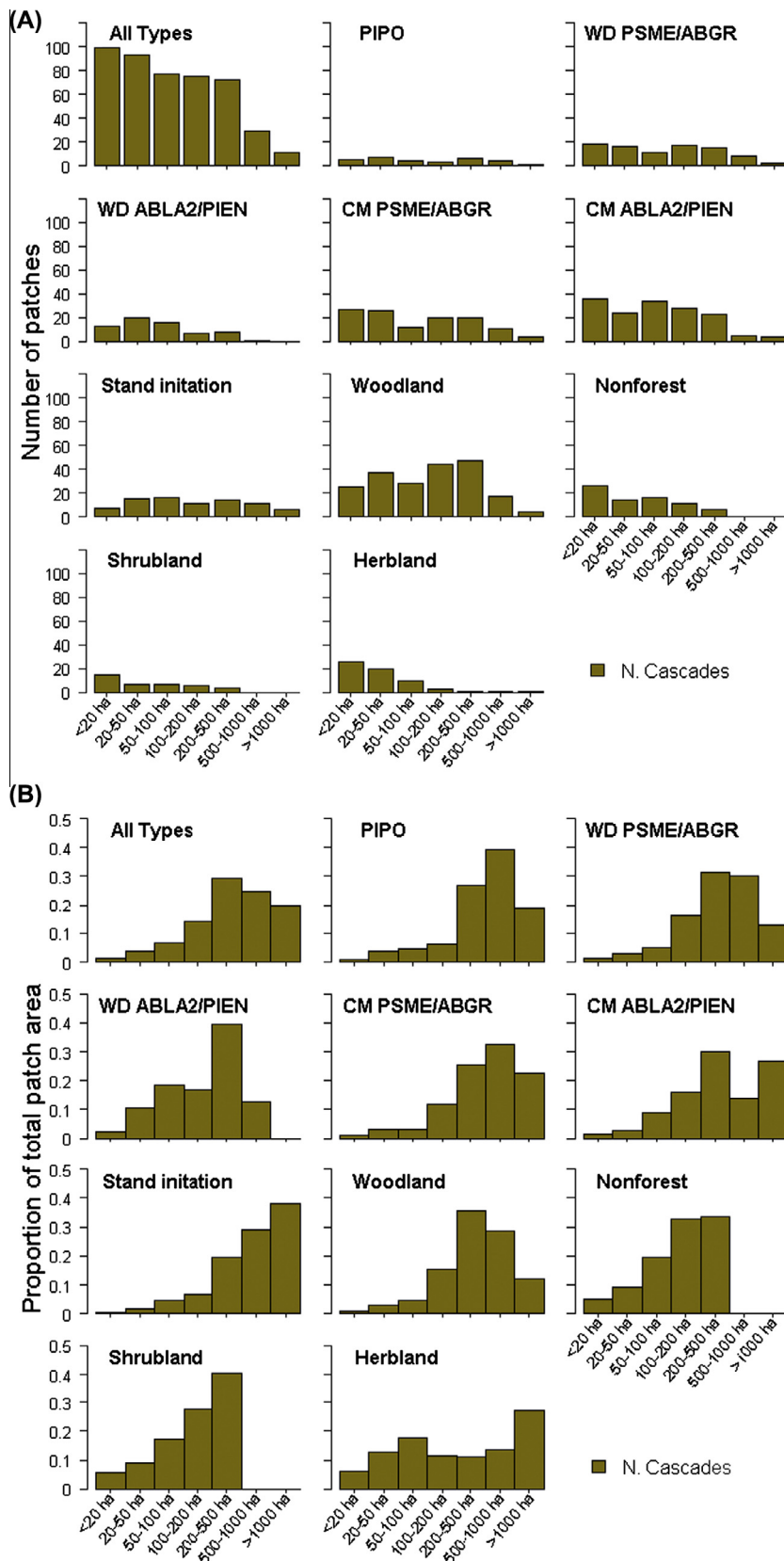


Fig. 10. (A) Patch size distributions of early 20th-century early successional conditions depicted in Table 1. Distributions from top left are: All types = pooled herbland, shrubland, woodland, bare ground (nonforest), and forest stand initiation structure patches of all forest PVT settings; PIPO, PSME/ABGR, 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).

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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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