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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.

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). 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-

* Corresponding author. Tel.: +1 605 688 5350.
E-mail address: michael.wimberly@sdstate.edu (M.C. Wimberly).

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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.

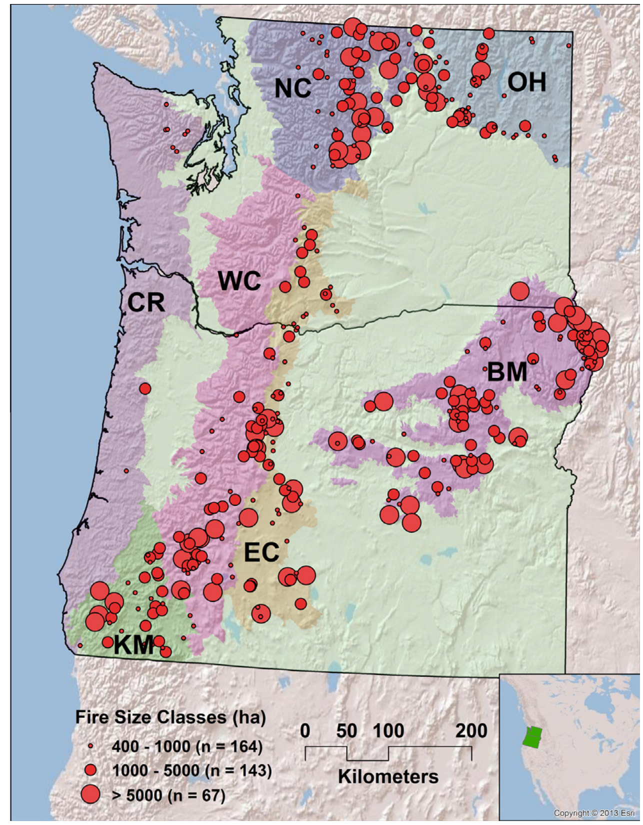

Fig. 1. Map of the Pacific Northwest region encompassing the states of Washington and Oregon along with the Bailey's ecoregion sections referenced in the report. Moist maritime conifer forests occur in the Oregon and Washington Coast Ranges (CR), the Western Cascades (WC), and in the western portion 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/).

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, 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 area 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 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%.

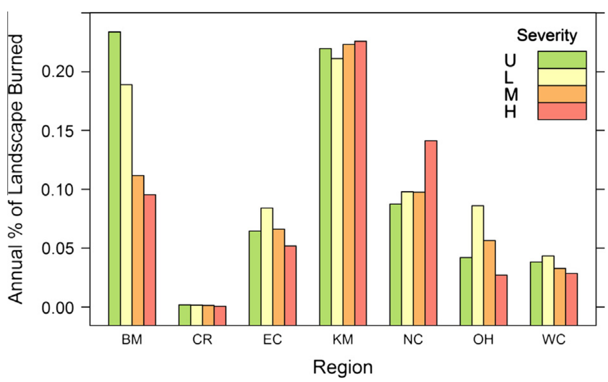

Fig. 2. Area burned by large fires (>400 ha) from 1984 to 2008 expressed as the mean percent of total ecoregion area burned per year by severity class (U = unburned, L = Low, M = Moderate, H = High). Data were obtained from the monitoring trends in burn severity (MTBS) project (http://www.mtbs.gov/).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

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

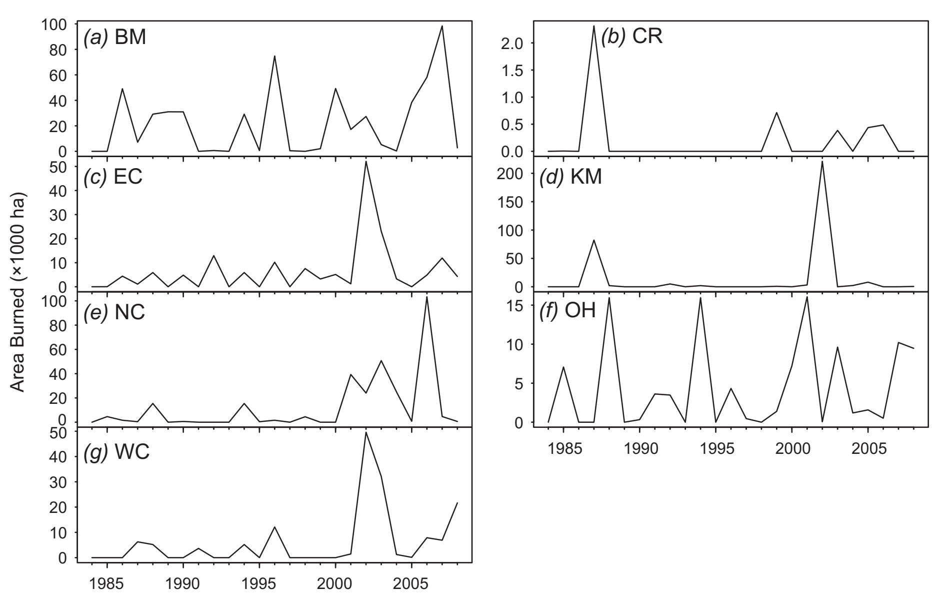

Fig. 3. Area burned by large fires (>400 ha) from 1984 to 2008 expressed as the time series of total ecoregion area burned per year. Data were obtained from the monitoring trends in burn severity (MTBS) project (http://www.mtbs.gov/). Abbreviations are defined in the caption for Fig. 1.

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 CO_2 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 files 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

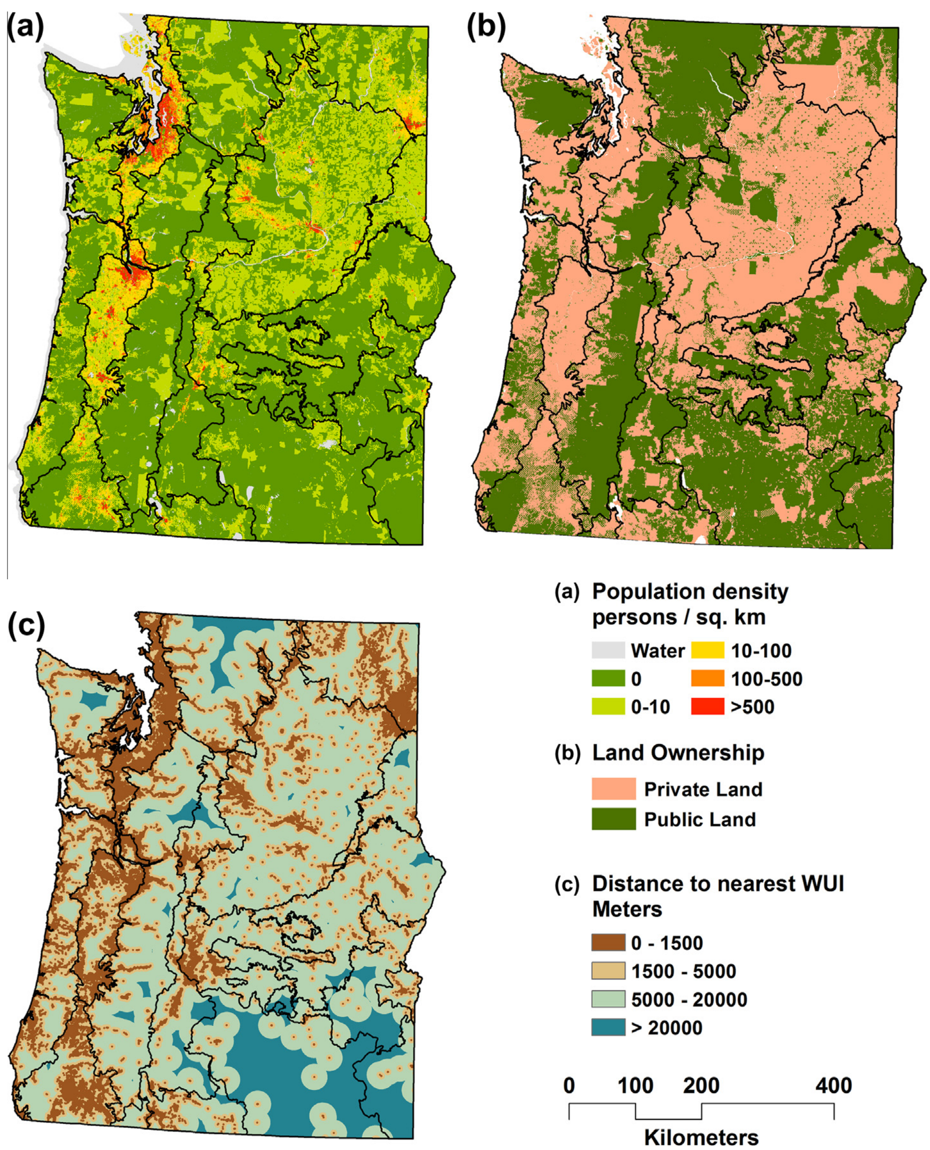

Fig. 4. Maps of human land use in the Pacific Northwest region: (a) population density (persons/km^2), (b) public and private land ownership, (c) distance to the Wildland–Urban Interface (WUI, meters). Population density and WUI data (Radeloff et al., 2005) were obtained from Spatial Analysis for Conservation 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-

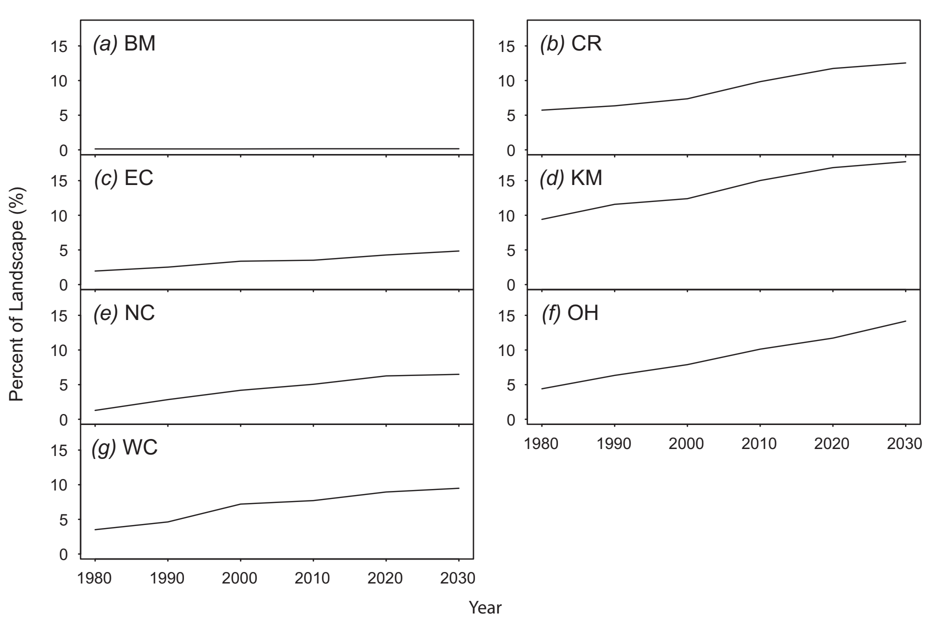

Fig. 5. Historical and projected trends of the percentage of human-occupied land, defined as at least one house per 16 ha (40 acres), within each Bailey's ecoregion section (Radeloff et al., 2010). Data were obtained from Spatial Analysis for Conservation and Sustainability lab at the University of Wisconsin–Madison (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 Hesseln, 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 Hesseln, 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 form 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 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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