Legacy of Repeated Disturbances in Mixed-Conifer Forests

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AUTHORIZATION TO SUBMIT DISSERTATION

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ABSTRACT

Managers and scientists alike are increasingly concerned with the impact of large disturbances on forests, especially under changing climate conditions. In this project, I aimed to understand the impacts of repeated disturbances, both wildfires and bark beetles, in mixed-conifer forests. Mixed-conifer forests are extensive throughout the western US, yet little is known about the impact of repeated disturbances on forest resilience. I addressed questions regarding vegetation responses following individual disturbances (either bark beetle or wildfires) and repeated disturbances (bark beetle and fire and repeated wildfires), as well as the impact of previous disturbances on the effects of subsequent wildfires. I used a combination of field work, remote sensing, and statistical analysis to answer questions at the stand and landscape scale. The interaction of bark beetles and wildfires did not result in different overall tree seedling density, surface woody fuel loading and stand structure than in areas only impacted by wildfire. Bark beetle outbreaks without subsequent fires also resulted in the highest seedling establishment. I found repeatedly burned areas to have reduced fuel loading and tree regeneration than once burned areas, indicating increased resilience. Also, past wildfires reduced burn severity of subsequent large wildfires, but many other factors such as day of burning weather and topography also influenced burn severity. My work informs our understanding of forest trajectories and forest resilience following repeated disturbances. This work furthers our understanding of changes in forest landscapes following single and repeated disturbances and advances our ability to manage forests for increased resilience in the face of future disturbances.

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DEDICATION

To all the wonderful people who helped me get here. From the mom who asked where I would go to grad school the semester I started undergrad and to all the friends who still think all I do is hug trees. You have kept me motivated and grounded.

To TSD for a lifetime worth of support and RWD for the final motivation to get this project done. ILYFEAANMW

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CHAPTER 1: THE IMPORTANCE OF REPEATED INTERACTING DISTURBANCES

PROJECT OVERVIEW

Past disturbances alter landscape heterogeneity (Allen et al., 2002, Binkley et al., 2007, Stephens et al., 2010, Churchill et al. 2013), forest structure (Hessburg et al. 2005), as well as the severity of subsequent disturbances (Collins et al. 2009, Holden et al. 2010, Parks et al. 2014). Both bark beetles and wildfires are recurring disturbance types that can result in extensive tree mortality and both are increasing in magnitude in the face of climate change (Jenkins et al. 2008), but little is understood about forest dynamics and ecosystem resilience following repeated disturbances. The western United States has experienced an increase in area burned and the number of large fires in recent decades (Westerling et al. 2006, Littell et al. 2009) and epidemic bark beetle populations have increased in recent years (Raffa et al. 2008, Bentz et al. 2010). As these large disturbance events more often overlap in space there has been a concern about the implications for subsequent fires, forest recovery and long-term resilience in the face of repeated disturbances (Thompson et al. 2007, Westerling et al. 2011). The increasing occurrence of large, extreme fires and extensive bark beetle-induced tree mortality of the past decade (Gedalof et al. 2005, Bentz et al. 2010), combined with the likelihood that such events will expand in the face of climate change, makes it critical for land managers and scientists to understand landscape changes following repeated disturbances, and their implications for long-term vulnerability and resilience to future disturbances.

Resiliency is defined as an ecosystem's ability to recover to a similar structure and species composition from a disturbance, and resistance is defined as an ecosystem's ability to remain relatively unchanged from a disturbance (Holling 1973). In this dissertation, I focus on several aspects of forest resilience and resistance following disturbances: tree seedling establishment, forest structure, surface woody fuel characteristics, and subsequent burn severity. Forests and ecosystems in general are not stagnant entities, but instead ever-changing. The density and tree species establishing following a disturbance indicates what kind of forest may be present on these sites in the future (e.g. Savage and Mast 2005). Remnant forest structure and surface fuels may increase or decrease the probability and severity of subsequent disturbances (Everett et al. 1999, Roccaforte et al. 2012), speaking to the resistant qualities of these areas. Dead woody material also provides wildlife habitat for cavity nesting birds, small mammals and increases site productivity (Brown et al. 2003, Hutto 2006), which may increase the resilience of the ecosystem as a whole. These post-disturbance fuel structures and tree regeneration patterns are explored in Chapters 2 and 3. Additionally, as I discuss in Chapter 4, the severity of an initial disturbance and thus the resulting vegetation and fuels can influence subsequent disturbances, changing the magnitude or severity (e.g. van Wagendonk et al. 2012, Parks et al. 2014) and even the occurrence of repeated disturbances (e.g. Teske et al. 2013, Parks et al. 2015). Thus in Chapters 2 and 3 I speak to the resilient ecosystem properties to disturbances, while in Chapter 4 I speak to the disturbance resistant properties.

Resilient and resistant ecosystems can be described through many visuals. In “state and transition” models, distinctly different ecosystem “states” are described with disturbances of other conditions that result in a transition from one state to the next (e.g. Stingham et al. 2003, Briske et al. 2008). Others describe resilience through a ball and cup model, in which an ecosystem stays within certain parameters, oscillating within conditions, until a disturbance is severe enough the ecosystem may move beyond the bounds of its original "cup" and result in an ecosystem shift to a different "cup" (e.g. Gunderson 2000). Similar to these previously published models, here I describe forest trajectory and resilience but with a more continuous visualization that can be applied to any of the aforementioned variables: tree seedling establishment, forest structure, and surface fuel characteristics (Figure 1.1). If a forest is resilient to a given disturbance it should recover from and continue along the same trajectory as an undisturbed forest. If the system is not resilient, it will follow a new trajectory, becoming structurally different and perhaps dominated by different species. Additionally, if an ecosystem is resistant to disturbances, the magnitude or severity of subsequent disturbances may be diminished and result in minimal ecosystem changes and quicker recovery. In the case of repeated disturbances, a forest may be resilient to an initial disturbance but a subsequent disturbance can overcome the forest's resilient capacity and result in a transition to a new trajectory.

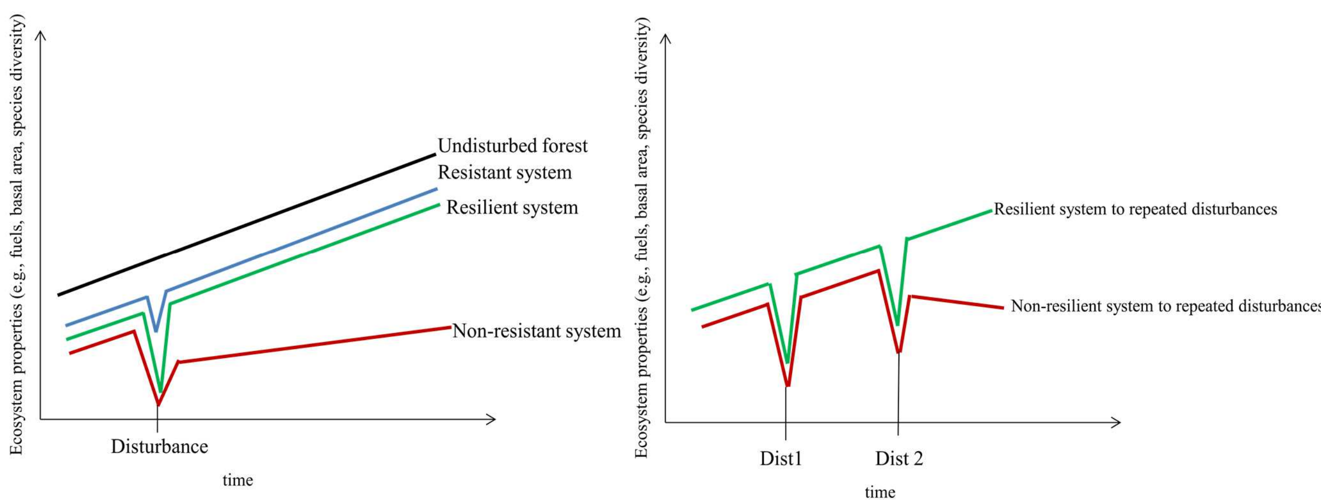

Figure. 1.1. Possible forest trajectory pathways through time, illustrating the effect of single disturbances (left side) and repeated disturbances (right side) on these pathways if the ecosystem is resilient or resistant as well as if it is not. Those that are resilient (green lines) and/or resistant (blue lines) recover to continue along a similar trajectory as undisturbed systems, but the non-resistant system diverges to an alternative trajectory. Forest ecosystems may also be resilient to an initial disturbance and then the frequency, severity, or other disturbance conditions may overcome forest resilience and result in a change in forest trajectory as shown by the red line in the right side graph. Though these lines indicate a

constant positive relationship between time and a given ecosystem property, this may not be the case for all variables infinitely through time. Instead these lines represent a forest trajectory not necessarily a constant, positive relationship.

While resilience theory is well established, the application of this theory to repeated disturbances in forested ecosystems is not well studied. In historical records and over long time periods we can observe the resilient properties of western U.S. forests to repeated fires (e.g. Fulé et al. 1997, Heyerdahl et al. 2008). However, after a century of fire suppression across the region that has lead to an increase in stand density, shifts in tree species composition, and increased vulnerability to large fires and other disturbances (Hessburg et al. 2005), we do not know if these forests retain their resilient properties in the face of large disturbances and climate change. For my dissertation research, I collected field and remote sensing data on multiple disturbances in mixed-conifer forests across Idaho, western Montana, and eastern Washington to examine these concepts of resilience and resistance. I explored how both previous bark beetle outbreaks and previous wildfires interacted with subsequent wildfires to alter forest recovery and resilience (Chapter 2 and 3), as well as how previous wildfires influenced subsequent wildfire severity and thus its resistant ecosystem qualities (Chapter 4).

In Chapter 2, I discuss a study I conducted in 2013 on four wildfires throughout Idaho and Montana that burned in 2007, all of which experienced previous bark beetle mortality. I present findings on the interaction between bark beetles and wildfires, in comparison to the occurrence of these disturbances without a recent prior disturbance to answer these questions: 1) How do tree seedling density and age structure differ between sites experiencing a single disturbance (fire or bark beetles) compared to sites experiencing repeated disturbances (bark beetle and fire)?; 2) How do stand structure and surface fuel loadings differ between undisturbed forests and those with high tree mortality from single or repeated disturbances?; and 3) How do bark beetle outbreaks followed by wildfire interact to influence surface fuels and seedling establishment?

In Chapter 3, I analyzed data collected in 2014 on wildfires that previously burned one to eighteen years before burning again in two large fires from 2007 to answer these questions: 1) How do fuel complexes and tree regeneration differ between areas burned in a single fire event and areas that have experienced two wildfires in recent history?, 2) How does the order of burn severity (high burn severity followed by low burn severity versus low burn severity followed by high) impact fuels and tree regeneration?, 3) How does time-between wildfires influence fuels and tree regeneration following subsequent wildfires?, and 4) Within burn perimeters, what distinguishes areas of increased forest resilience from areas that may be following an alternative successional pathway as a result of these repeated wildfires?

In Chapter 4, I discuss my findings from an analysis on the impact of weather, topography, vegetation, and previous burn severity on the burn severity of three large wildfires in Washington and Idaho. I answer these questions: 1) Was burn severity of subsequent wildfires influenced by previous wildfires?, and 2) Are climate/weather or topographic conditions mitigating factors in the effectiveness of past wildfires to reduce current fire behavior?

Finally, in chapter five I draw conclusions. I focus on the management implications, additional research needs. I emphasize the overall significance of my dissertation research.

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CHAPTER 2: BARK BEETLES AND WILDFIRES: HOW DOES FOREST RECOVERY CHANGE WITH REPEATED DISTURBANCES IN MIXED CONIFER FORESTS?

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ABSTRACT

Increased wildfire activity and recent bark beetle outbreaks in the western United States have increased the potential for interactions between disturbance types to influence forest characteristics. However, the effects of interactions between bark beetle outbreaks and subsequent wildfires on forest succession remain poorly understood. We collected data in dry mixed-conifer forests across Idaho and western Montana to test whether vegetation responses differ between sites experiencing single and repeated disturbances. We compared tree seedling density and age, surface fuel loading, and stand structure characteristics in stands that experienced either high severity wildfire, large-scale tree mortality from bark beetles, or stands that experienced high bark beetle mortality followed by severe wildfire within 3-8 years of attack. Tree seedling density was 300-400% higher in gray bark beetle-affected stands than burned sites, but there was no evidence that a beetle and wildfire interaction affected seedling densities. The age distribution of Douglas-fir and grand fir seedlings in stands with repeated disturbances differed from those that only experienced wildfire, suggesting that seed availability varies between these stands. Though both bark beetle outbreaks and wildfires resulted in the death of numerous large trees and surface woody fuel loads 100-200% greater than control sites, the creation of large snags and higher fuel loads across the landscape may have ecological benefits. Compounding effects of bark beetle activity and wildfires were not observed in surface fuel loadings or stand densities. Overall, the effects of high severity wildfire drove post-disturbance fuel complexes and succession whereas the effects of *Dendroctonus pseudotsugae* and *Dendroctonus brevicomis* outbreaks before wildfires resulted in minimal post-wildfire differences. We conclude that although seedling age structure is responsive to bark beetle and fire interactions, in terms of fuel complexes and tree densities these disturbances are non-additive and compounding effects on forest trajectory of dry mixed-conifer forests of the northern Rockies were not supported.

Key words: dry mixed-conifer forests, bark beetle and fire interactions, repeated disturbances

INTRODUCTION

The western United States has experienced numerous large fires (Westerling et al. 2006, Littell et al. 2009) and extensive tree mortality from bark beetle (Curculionidae: Scolytinae) populations in recent decades (Raffa et al. 2008, Bentz et al. 2010, Meddens et al. 2012). Understanding changes in vegetative structure and tree seedling responses following disturbances is critical for understanding temporal patterns of long-term vulnerability and forest resilience. The increased frequency of extreme fire events and large-scale bark beetle mortality during the past decade (Gedalof et al. 2005, Bentz et al. 2010), combined with increasing likelihood of such events under climate change, has created a need to understand how interactions between these disturbance types may alter ecosystems. Although most research in this area examines the effect of bark beetle mortality on subsequent wildfire burn severity (defined herein as relative tree mortality), post-disturbance succession is also of widespread interest to ecosystem managers (Simard et al. 2011, Hicke et al. 2012, Harvey et al. 2013, Carswell 2014).

There are several major concerns about the impact of bark beetle outbreaks on subsequent fire and forest structure. First, bark beetle outbreaks alter forest structure and the distribution and spatial arrangement of surface and canopy fuels (e.g. Hoffman et al. 2012, 2015, Schoennagel et al. 2012). However, many of these changes are dependent on time-since outbreak, relative tree mortality and rate of mortality, which can feed back to influence tree regeneration and forest structure (Donato et al. 2013). Initially following a successful attack, the "red phase" occurs as needles will turn from green to red and can last one to four years. The "gray phase" follows, lasting up to a decade and is characterized by many standing dead trees but with no needles remaining in the canopy; (e.g. Simard et al. 2011, Hoffman et al. 2012, Schoennagel et al. 2012, Donato et al. 2013). The "old" or "silver" phase occurs once standing dead trees begin to fall and break, and lasts up to 30-40 years post-outbreak (e.g. Schoennagel et al. 2012, Donato et al. 2013). Previous researchers suggested, through simulation modeling, that bark beetle activity followed by wildfires interact to influence long term forest stand dynamics (Simard et al. 2011, Hoffman et al. 2013, Hoffman et al. 2015), yet, there is little empirical data that quantified these changes.

Second, it is presently unknown how repeated disturbances influence woody fuel accumulation and stand structure (Harvey et al. 2013). Shifts in stand structure and fuels are important considerations for assessing potential fuel hazards, and coarse dead downed woody material and standing dead trees also play an important ecological role. Most tree killing bark beetle species attack mature seed-producing trees expediting forest structural changes upon successful colonization (McCullough et al. 1998, Bjorklund and Lindgren 2009). Additionally, wildfires increase the prevalence of both large and small diameter standing dead trees, especially in areas of high burn severity (Everett et al. 1999, Passovoy and Fulé

2006). Dead woody material from mature trees can improve wildlife habitat for cavity nesting birds (Bull et al. 1997, Hutto 2006), create canopy gaps for tree seedling establishment sites (Takahashi et al. 2000, Kennedy and Quinn 2001) and are sources of organic material that enhance forest productivity (Graham et al. 1994). The ecological benefits that arise from large tree mortality may be lost when forest ecosystems experience a high severity wildfire shortly after beetle mortality, degrading ecosystem functionality.

Our collective understanding about the impact of repeated disturbances on vegetation response, defined here as forest structure and seedling establishment, is limited. Much of the recent research has involved fire-on-fire behavior interactions (e.g. Peterson 2002, van Wagtendonk et al. 2012, Parks et al. 2013, Teske et al. 2013), whereas the literature on post-fire effects and the impacts of different types of repeated disturbances is lacking. A substantial body of research investigates the effects of “time-since-disturbance” following wildfire (e.g. Passovoy and Fulé 2006, Roccaforte et al. 2012, Stevens-Rumann et al. 2013) and bark beetle outbreak (e.g.Simard et al. 2011, Hoffman et al. 2012, Jolly et al. 2012, Donato et al. 2013, Hoffman et al. 2015) independently, but little data exists about forest structure resulting from the combined effect of these two disturbance types. Most research on bark beetle-fire interactions to date addresses high elevation forests (e.g. Bebi et al. 2003, Klutsch et al. 2011, Kulakowski et al. 2013), while little is known about low elevation mixed-conifer forests (Hicke et al. 2012). There is a need for forest type-specific research as high elevation spruce (*Picea*) and lodgepole (*Pinus contorta*) forests and low elevation ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii)* differ greatly in structure, fire regimes, bark beetle communities, and climate, making direct comparisons difficult. Though little is known about interacting disturbances in dry mixed-conifer forest types, these forests are extensive across the western USA, comprising at least 50% of US Rocky mountain forests (Schoennagel et al. 2004, Hessburg et al. 2005, Baker 2009). Dry mixed-conifer forests of the northern Rockies are dominated by ponderosa pine, Douglas-fir, and grand fir (*Abies grandis*) and are vulnerable to wildfires and bark beetle outbreaks following a century of fire suppression (Hessburg et al. 2005) and recent drought (Schoennagel et al. 2005). The only previous field study of interactions between disturbances from fire and bark beetles in dry conifer forests found that bark beetle outbreaks did not affect subsequent burn severity. However, low tree seedling densities on sites with repeated disturbances suggest legacy effects on forest succession (Harvey et al. 2013).

We investigated how repeated disturbances influence subsequent seedling response in dry-mixed conifer forests, examining four wildfires with adjacent and interior *Dendroctonus* bark beetle outbreaks to answer the following questions: (1) How do tree seedling density and age structure differ between sites experiencing a single disturbance (fire or bark beetles) compared to sites experiencing repeated disturbances (bark beetle and fire)?; (2) How do stand structure and surface fuel loadings differ between undisturbed forests and those with high tree mortality from single or repeated disturbances?; and (3) How do bark beetle outbreaks followed by wildfire interact to influence surface fuels and seedling establishment?

METHODS

Site selection

Data were collected in dry mixed-conifer forests in the interior northwestern US from the Continental Divide in Montana through Idaho. To determine potential plot locations, we used a combination of spatially explicit data sets describing wildfires, bark beetle outbreaks, and vegetation type. For wildfires, we analyzed two Monitoring Trends in Burn Severity

(MTBS) products: the fire perimeters for each wildfire, and the pre-processed Relative differenced Normalized Burn Ratio (RdNBR) spectral index data. To analyze the severity and extent of bark beetle outbreaks, we used USDA Forest Service Aerial Detection Survey (ADS) data. To determine vegetation type we used Environmental Site Potential (ESP), a LANDFIRE product based on the biophysical environment and standardized across the US, and selected only sites that were classified as Northern Rocky Mountain Dry-Mesic Montane Mixed-Conifer Forests (Rollins 2009). The MTBS-derived fire perimeters are constructed by analysts who rapidly and manually digitize the Landsat imagery. Burn severity thresholds created for forested and non-forested ecosystems (Miller and Thode 2007), were manually identified for each fire using interpretations of dNBR and RdNBR, raw pre- and post-fire satellite imagery, plot data (if available), and the analysts' own judgment (Eidenshink et al. 2007). The ADS data layers are created by manually mapping from aircraft the tree mortality attributed to different bark beetles or other mortality agents. We used the ADS data to identified areas affected by Douglas-fir beetle (*Dendroctonus pseudotsugae*) and western pine beetle (*Dendroctonus brevicomis*). Sites lie within Climate Division 4 in Idaho and 1 in Montana (NOAA 2014). These climates are defined by cold winters (-6 ^{o}C January mean) and warm summers (17 ^{o}C July mean) with 58 cm mean annual precipitation (averaged over 1900-2008; NOAA 2014).

We used MTBS, ADS, and ESP layers to stratify potential plot locations in dry-mixed conifer forests within wildfires that occurred in 2007 in stands that experienced bark beetle outbreaks in the previous 3-8 years. The 2007 fire year was chosen because there were multiple large wildfires that met our a priori requirements of forest type and disturbance history that year. We selected burn severity thresholds from the RdNBR data to identify areas that met our burn severity criteria of approximately >80% tree mortality or "high severity". Additionally, we only focused on areas that experienced moderate to high bark beetle induced mortality (25-90% total tree mortality). Given the diverse tree species composition, consistently high tree mortality from bark beetle induced mortality was not attainable as bark beetles select for larger trees of a given species. For all samplings we used sites for which burn severity, bark beetle caused tree mortality, from MTBS and ADS data, and ESP were verified through field observations. Of all visited sites, 92% were accurately determined by the three remote sensing data layers. On the remaining 8% of visited sites, ADS data had the highest level of inaccuracy, with several errors in the ESP for a given site as well. We did not collect data on the 8% of non-validated sites.

Based on information from local land managers we excluded sites that experienced commercial or non-commercial logging, tree planting, or other management treatments in the past 50 years, including post-fire rehabilitation and hazard tree removal. This was done to reduce the variability caused by the potential effects of management on both seedling density and fuel loading. The four wildfires that met our criteria were: the Rattlesnake (1292-2001 m) and Poe Cabin fires (1054-1711 m) on the Nez Perce-Clearwater National Forest, Rombo Mountain Fire (1419-2038 m) on the Bitterroot National Forest, and Middle Fork Complex (Lightning) Fire (1019-1399 m) on the Boise National Forest (Fig. 1). These fires burned in mixed severity and began in July of 2007 (Table 1).

Field methods

We sampled the effects of five disturbance types: 1) wildfire without prior bark beetle-induced tree mortality, 2) bark beetle-attacked stands in gray phase (3-8 years post-outbreak) when the wildfire occurred, 3) red phase (1-2 years prior to study), and 4) gray phase areas outside the fire perimeter, and 5) undisturbed (control) plots. Though bark beetle outbreaks at different phases are not different disturbances we treated these distinct time-since-disturbance periods as separate factors in the analysis. Within each treatment we randomly established eight to ten sites for a total of 40-50 sites per wildfire, or 180 plots across the four wildfires. Control sites were established in areas around the fire perimeter that had similar elevation, aspect and slope to disturbed plots. Pre-disturbance, all sites were dominated by mature ponderosa pine, Douglas-fir, and grand fir with a minor component on some sites of lodgepole pine , western larch (*Larix occidentalis*), and subalpine fir (*Abies lasiocarpa)*. Slopes were highly variable on plots, ranging from 2-70% slope. Parent material on the Poe Cabin Fire was loess mixed with basalt colluvium; both the Rattlesnake Fire and Rombo Mountain Fire had quartzite colluvium; and the Middle Fork Complex (lightning) Fire burned over quartz monzonite (Baker et al. 1983).

At each randomly located site we established a 0.04-ha circular plot and a 30-m long, variable-width belt transect. We recorded the slope, aspect, and elevation from the center of this circular plot. On the 30-m long transect, laid perpendicular to the elevation contour and following the dominant hillslope, total width of the belt transect varied between 1 and 10 m using similar methods to Droske (2012). Width was determined prior to sampling with the objective of sampling at least 30 tree seedlings of the most dominant species. If no or few seedlings were observed initially on the belt transect, a 10-m width was set as a default. Data were recorded for all tree seedlings within the belt transect(including species), distance between each primary branch node, and approximate age of each seedling determined by counting the total number of nodes. We also recorded distance to and species of the 10 nearest live seed sources from the center of the 30-m transect.

On each circular plot, we tallied all trees >5-cm DBH as alive or dead, including stumps, to estimate percent tree mortality induced by the fire. Pre-wildfire tree density was calculated by adding fire-killed stems together with live trees. We determined cause of tree mortality, either fire or bark beetle, prior to fire using similar methods to Harvey et al. (2013). Trees were recorded as 'killed by bark beetles prior to fire' if presence of exit holes on outer bark was detected, if fully excavated adult and larval galleries were present on the vascular cambium (>50% of bole circumference or remaining visible cambium), and if no needles were retained in the canopy which may indicate bark beetle induced mortality after wildfire. Dead trees with no evidence of pre-fire beetle activity or clear evidence of post-fire beetle activity were assumed to have been 'killed by fire.' We measured diameter at breast height (DBH) and recorded tree species of all living and dead trees (the latter referred to as snags) >5 cm DBH to quantify post-disturbance stand structure. We also quantified the density of standing dead trees >40 cm DBH because they provide greater ecosystem benefits and generally have the longest retention times within forested ecosystems (Bull 1983, Bull et al. 1997, Hutto 2006).

To quantify down woody debris, we measured fine woody debris and litter and duff depth on the 30-m transect using methods established by Brown (1974). Woody debris was broken up into size classes (0–0.64, 0.65–2.54, 2.55–7.62, >7.62 cm diameter), to correspond with 1-, 10-, 100-, and 1000-hr time-lag classes. The 1- and 10-hr fuels were measured along 6 m of transect and 100-hr fuels were tallied on a 10-m section of the transect. The 1000-hr fuels (coarse woody debris (CWD); logs ≥7.62 cm) were measured in 100-m^2 circular subplot located within the center of the overstory plot. Within this subplot the length and diameter at both ends was measured for every 1000-hr fuel and the fuel load was estimated following

Keane et al. (2012). The 1000-hr fuels were also categorized by sound and rotten classes (Fosberg, 1970).

Data Analysis

To test differences of fuel variables (pre-disturbance density, live tree density, large snag density, 1-, 10-, 100-, 1000-hr dead down and woody fuel loads, litter and duff depth) between disturbances we used a mixed effects analysis of variance (ANOVA), with site as a blocking variable (df=3) and disturbance treatment as the main effect (df=4). We checked assumptions of normality and equal variances through scatter plots of response variable residuals. To assess significant differences among disturbance classes when significance was observed in the overall test, Tukey's HSD test was used. Additional variables such as slope, aspect, and elevation were tested as covariates for all fuels categories, but were not found to be important. Finally, to test the influence of percent bark beetle mortality on fuel loading variables and seedling density we used percent bark beetle mortality as a continuous variable within each of the three disturbance types that experienced bark beetle mortality (red, gray, and bark beetle followed by wildfire). Analysis was performed in JMP (SAS Institute Inc. 2007) with $\alpha = 0.05$.

To examine differences in the combined fuel complexes and tree seedling density among disturbance types we used a Multi-response Permutation Procedures (MRPP; Mielke and Berry, 2001) in an Excel macro developed by the Rocky Mountain Research Station Statistician (King 2008). Live tree density (>5 cm DBH), large snag density, biomass of 1-, 10-, 100-, 1000-hr fuels, litter depth, duff depth and log-transformed seedling densities as response variables and grouped by disturbance type, with fire as the blocking variable in the analysis were included. Pre-analysis, data were standardized by dividing each observation by the mean for that variable. When overall significant differences were detected we used Peritz Closure (Petrondas and Ruben, 1983) multiple pairwise comparisons to separate disturbance type multivariate means.

We used total tree seedling density of all species to investigate differences in tree seedling establishment and age across disturbances, while age distribution analysis was performed on the three most abundant tree seedling species: ponderosa pine, Douglas-fir and grand fir. We log-transformed density data and added 1 prior to analysis to account for the non-normal distribution and zero-inflated nature of seedling density. Topographic variables such as slope, aspect, elevation and distance to seed source were analyzed as possible predictor variables in seedling density. Seedling age analysis was performed on the individual seedling rather than analyzing data at the plot level to accurately quantify the age structure of seedlings within a disturbance type rather than plot level averages. In these analyses we did not block by fire. The total number of tree seedling observations was 200, 1275, and 1060 for ponderosa pine, Douglas-fir, grand fir seedlings respectively.

RESULTS

Tree regeneration

Tree seedling species were predominantly ponderosa pine, Douglas-fir, and grand fir, and one wildfire (Rattlesnake Fire) had a greater density of lodgepole pine. On less than 10 plots, Englemann spruce (*Picea engelmannii)* and subalpine fir seedlings were observed, comprising a minor component of total seedling density.

Seedling density varied significantly among disturbance types, with seedling densities four times greater in gray phase stands compared to all other disturbances (Fig. 2a). No statistically significant differences were observed between burned and control sites, although control sites had a slightly higher mean density of tree seedlings. We found no significant relationships between topographic variables such as aspect and slope and tree seedling density ($F=1.79$, $P=0.095$ and $F=0.24$, $P=0.63$ respectively). Seedling density was positively correlated with elevation and negatively correlated with distance to seed source ($F=8.62$, $P=0.0039$; $F=14.8$, $P<0.0001$, respectively). However, these relationships were uniform across all disturbances and no interaction effect was observed. Seedling density was not significantly correlated with percent of bark beetle induced tree mortality in either the red ($F=2.17$, $P=0.15$, $R^2=0.07$) or gray red ($F=0.45$, $P=0.51$, $R^2=0.018$) phase stands or the bark beetle and fire affected stands red ($F=1.81$, $P=0.19$, $R^2=0.07$).

Tree seedling age structure differed not only by disturbance type but also by species. Ponderosa pine seedlings were a relatively minor component of the seedlings observed on control sites and bark beetle-affected sites, but densities were high on burned sites (Fig. 2a). Nevertheless, those present were significantly older than ponderosa pine seedlings on burned sites, with a mean age of six years on red phase bark beetle sites and up to seven years on control sites ($F=20.73$, $P<0.0001$; Fig. 2b). Ponderosa pine seedling ages were similar for bark beetle and fire-affected sites compared to fire-alone sites. We found more Douglas-fir and grand fir seedlings on control, red, and gray phase sites relative to ponderosa pine seedlings (Fig. 2c&d). Distributions of seedlings among disturbance types differed for both Douglas-fir ($F=28.30$, $P<0.0001$) and grand fir ($F=4.16$, $P=0.0024$). Though distributions on bark beetle affected sites were right skewed, both species had the oldest mean age on red phase sites, while gray phase sites had slightly younger mean seedling ages. Control and fire-alone sites had younger cohorts of seedlings for both species, but were not significantly different from one another. However, both Douglas-fir and grand fir also had a significantly older cohort of seedlings on bark beetle and fire-affected sites compared to fire-alone sites.

Fuel complex:

Across all wildfires, pre-disturbance stand density did not vary by treatment type (Fig. 3a). Yet, post-disturbance live overstory tree density and basal area were significantly lower, if existent at all, on burned sites compared to unburned sites (Fig. 3b & c). Mean live tree densities of 265-281trees ha^{-1} were observed in both red and gray phase bark beetle-affected stands compared to 450 trees ha^{-1} in control stands. The densities of large snags were significantly higher in disturbed stands although no significant differences between bark beetle-affected stands (red and gray phase) and burned stands (Fig. 3d) were evident.

Overall, total downed woody debris was up to 200% greater on burned sites compared to unburned sites (Fig 4). Although 10-hr and 100-hr fuels did not vary significantly among disturbances ($F=1.90$, $P=0.11$; $F=1.33$ $P=0.26$ respectively), there were slightly greater mean fuel loadings on burned sites. Conversely, fine woody debris (FWD; 1-hr fuels) and CWD were significantly different among disturbances (Fig. 4a &b). Rotten CWD was not a large component of the total CWD (<5%) and did not significantly vary among disturbances ($F=1.70$, $P=0.20$), thus sound and rotten CWD were pooled for all additional analysis. CWD and FWD fuel loadings were highest on bark beetle-affected areas in the gray phase, indicating the importance of time-since disturbance when compared to low fuels loadings in red phase stands. Similarly, high CWD loadings were observed in bark beetle and fire-affected stands, whereas fire-alone had slightly lower mean CWD loadings. Given the recent nature of the disturbance, red phase stands had CWD or FWD fuel loadings similar to control sites. Similar trends were observed in the analysis by percent bark beetle mortality. CWD and FWD loadings were positively correlated with percent bark beetle mortality in gray phase stands (F=5.54, P=0.025, R^2=0.14 and F=4.18, P=0.049, R^2=0.11), however no relationship was observed in red phase stands (F=2.14, P=0.15, R^2=0.05 and F=0.48, P=0.49, R^2=0.014), nor in stands of repeated disturbances (F=0.0004, P=0.98, R^2=0.0001 and F=0.05, P=0.83, R^2=0.001). The smallest FWD loadings were observed on burned sites, both fire-alone and with previous bark beetle.

Burned sites, both with and without prior bark beetle activity, had significantly smaller litter and duff depths compared to unburned sites (F=39.58, P<0.0001 and F=34.12, P<0.0001 respectively). This was the only significant difference in the pairwise comparisons as there were no significant differences between phases of bark beetle mortality and control sites, and litter and duff depths were similar among all burned sites.

Significant differences were observed in the multivariate analysis of combined fuels and stand structure characteristics among disturbances (P<0.0001). Based on the pairwise comparisons, red and gray phase sites were not significantly different from one another (P=0.31) but did vary significantly from burned sites (both fire alone and bark beetle and fire) and control sites (P<0.0001). Additionally, while bark beetle followed by wildfire sites and wildfire only sites did not significantly differ from one another (P=0.61), they were significantly different from all unburned sites (control, red and gray phase sites; P<0.0001). Further, control sites were significantly different from sites of any disturbance types (P<0.0001).

DISCUSSION

Our empirical data suggests that in areas with high tree mortality, regardless of the agent of mortality, there are no interactive effects on post-disturbance seedling density or fuel complexes in dry mixed-conifer forests of the Northern Rocky Mountains. Empirical studies on interacting disturbance types, specifically bark beetle mortality followed by wildfire events, in dry-mixed conifer forests have been largely limited to simulation modeling. We show that post-fire tree seedling age distribution was impacted by interacting disturbances. Still, total seedling density did not vary as a result of repeated disturbances, which did not support our hypothesis that seedling density would be reduced following repeat disturbances. Conversely, as predicted, there was no relationship between bark beetle outbreaks before wildfires and subsequent fuel loading post-wildfire, suggesting that disturbances do not interact to alter fuel complexes.

Tree regeneration: mixed interactive effects of fire and bark beetles

Tree seedling density was lower in both fire alone stands and in bark beetle and wildfire stands compared to unburned stands suggesting that there is not an interactive effect of fire and bark beetles on tree seedling densities when burn severity is high. These results contradict those of Harvey et al. (2013) and Kulakowski et al. (2013) who found lower seedling densities on bark beetle and fire affected sites compared to fire alone. Differences in forest type and site productivity between studies may be the cause of the mixed results (Kramer and Johnson 1987, Shatford et al. 2007). As no seedlings were detected on 95% of all burned sites, any additional reduction in seedling density from previous bark beetle activity may not be visible on these overall low regeneration sites. Unlike previous studies, we did not find observable differences in seedling density between fire alone and bark beetle and fire affected sites.

Although burned sites did not vary based on previous bark beetle activity, multiple differences in seedling densities were observed across disturbance types. The highest seedling densities on gray phase sites demonstrate the importance of time-since-outbreak on stand structure and in the establishment of tree seedlings (Collins et al. 2011). Seedling density on red phase sites was not significantly different from control sites, likely due to the recent nature of the disturbance (Collins et al. 2011, Schoennagel et al. 2012). Though seedling density was lower on burned sites compared to unburned sites, when seedlings were present the observed densities were not below recommended levels for stand replacement in this forest type (Shatford et al. 2007).

In addition to total tree seedling density, greater densities of individual species were observed within different disturbance types, likely a result of physiological differences in species. We found a greater abundance of Douglas-fir and grand fir seedlings on unburned sites compared to ponderosa pine. These results are in part explained by the relative abilities of Douglas-fir and grand fir to regenerate on both mineral soil and in litter and duff layers (Gray and Spies 1996, Malcolm et al. 2001), whereas shade-intolerance and a preference for mineral soil found after wildfires led to greater ponderosa pine seedling densities on burned sites (Romme 1982, DeLuca and Sala 2006). One of our four fires, the Rattlesnake Fire, had a higher proportion of lodgepole pine seedlings compared to the other wildfires. These sites, though identified as Northern Rocky Mountain Dry-Mesic Montane Mixed-Conifer Forests using LANDFIRE ESP data, had lodgepole pine in the pre-disturbance stand structure comprising <25% of its structure and the other three fires had <5% lodgepole pine. Lodgepole pine seedling densities were greatest on burned sites due to their serotinous cones and role as an early colonizing tree species (Schwilk and Ackerly 2001, Schoennagel et al. 2003).

Although tree seedling density did not vary between fire-alone and repeated disturbances stands, there were differences in the age distribution of both Douglas-fir and grand fir seedlings. In both cases bark beetle and fire-affected stands had significantly older seedlings compared to fire-alone stands. Most seedling age distributions are right-skewed with a majority of seedlings only a year or two old, and we expect to see continued seedling establishment through time (Shatford et al. 2007) with perhaps a decline in older seedlings, because survivorship is low in young tree seedlings (Peet and Christensen 1987, Gray and Spies 1996). This pattern can be observed in our control, red, and gray phase stands. However, bark beetle and fire affected stands had a significantly older cohort of seedlings that was approximately 3-6 years old. Neither distance to seed source nor any topographic variables could explain these differences as the influence of these bottom-up factors (i.e. slope, elevation, and distance to seed source) were consistent across disturbances. A decreased seed bank from repeated disturbances may be one explanation for the reduction in continued germination of Douglas-fir seedlings (Stark et al. 2006, Harvey et al. 2013). At the time of the wildfire these repeated disturbance stands were in the gray phase, thus we expect that these stands had similarly high seedling densities as we observed in gray phase stands. The germination of an older cohort of seedlings that established in-between the two disturbances left a limited seed supply for continued germination through time (Franklin et al. 2000, Larsen and Franklin 2005). Without a sufficient seed source and lacking a seed bank these sites may have less tree regeneration and lower seedling density in future years, especially on sites where seed trees were >150 m away (Bonnet et al. 2005, Donato et al. 2009, Harvey et al. 2013).

Vegetation responses can be influenced by many factors, including distance to seed source, elevation, climate and weather. We found distance to seed source was negatively correlated with seedling density and positively correlated with elevation, similar to Donato et al. (2009) and Harvey et al. (2013). Multiple authors demonstrate a variety of additional explanations for temporal patterns of seedling establishment including climate following disturbances (e.g. Brown and Wu 2005) and micro-climate variability (e.g. Gray and Spies 1997). However, we did not see similar age structures across all disturbance types, which would indicate favorable growing years compared to less favorable growing years if larger climatic drivers were responsible for variations in seedling establishment (Kitzberger et al. 2000). Further, we saw the same pattern within each disturbance type across a large geographic area (approximately 40,000 km^2), which would not occur if more localized weather was responsible for differences in vegetation responses. Lower seedling densities in burned areas may be explained by high levels of solar radiation due to the lack of canopy cover and high soil burn severity (White et al. 1996, Zald et al. 2008). However, more detailed information in specific micro-climate drivers of seedling establishment was not measured for our sites. Disturbance types and topographic variations were likely the strongest drivers of seedling densities and age structures in this study.

Fuel complex: higher fuels loads with longer time-since disturbance

Large snag dynamics across stands illustrated the mechanism of mortality of these two disturbances and live stand structure was predicted given our a priori requirements for each disturbance. Large snag densities in burned stands did not exceed densities of stands in either phase of bark beetle-induced tree mortality even though tree mortality in burned stands was close to 100% and as low as 25% on bark beetle affected stands. The two *Dendroctonus* bark beetles species in this study specifically targeted larger diameter trees (McCullough et al. 1998, Bjork and Lindgren 2009). This resulted in a greater number of large snags to total tree mortality seen in bark beetle affected stands compared to burned stands. While the death of large, old live trees is concerning to managers, large snags also serve a key ecological role and thus the creation of large snags is not necessarily a detriment to forested ecosystems (Bull et al. 1997, Hutto 2006). Our disturbed stands averaged 35-60 snags/ha, which well exceeded the recommended snag densities for restoration of 5-8 snags/ha (Nez Perce National Forest Plan 1987, Bitterroot National Forest Plan 1989). Still, small areas of high snag densities across the landscape may be beneficial especially during early succession following a disturbance (Haggard and Gains 2001, Kotliar et al. 2002, Hutto 2006). One large concern for areas of repeated disturbances is the potential longevity of large snags. In stands that experienced both disturbances large snags may have shorter retention times due to structural weakening from a fire after bark beetle induced mortality (Bull 1983, Bagne et al. 2008). The ecological function of snags may be short lived in areas of repeated disturbance.

High CWD loadings on gray phase sites and on burned sites (both repeated disturbances and fire alone) were consistent with a pattern of woody fuel accumulation in Douglas-fir and ponderosa pine forest types following bark beetle outbreaks (Jenkins et al. 2008, Hoffman et al. 2012) and wildfires (Passovoy and Fulé 2006, Roccaforte et al. 2012). Low CWD loadings in red phase stands were also consistent with woody fuel accumulation trajectories (Donato et al. 2013). Surface fuel accumulation is highly dependent on time-since-outbreak and minimal changes in dead down and woody surface fuel loadings occurred 1-2 years post-outbreak regardless of percent tree mortality (Donato et al. 2013).

We found a lack of relationship between bark beetle induced tree mortality and fuel loading in areas of repeated disturbances was a result of two factors. First, these repeated disturbance stands (bark beetle and fire) were in the gray phase when the wildfire occurred thus surface fuels were likely greater at the time of the wildfire (Jenkins et al. 2008, Donato et al. 2013) and a proportion of these original fuels were consumed during the wildfire. Second, and more importantly, these stands also experienced high tree mortality from a wildfire signifying the relationship of fuel loading to percent tree mortality by only one agent (bark beetles) does not adequately capture the relationship between total tree mortality and fuel loading.

Dead and down woody fuel loadings were greater on older disturbances compared to control sites, however CWD loadings did not exceed recommended loadings six years post wildfire and three to eight years post-outbreak (Brown et al. 2003). This recommended CWD loading outlined by Brown et al. (2003) is a combination of recommended loadings for ecological benefit such as soil productivity and small mammal habitat, while not creating excessive fire hazard and soil heating if a wildfire was to occur. Thus, at this period post-disturbance, woody biomass is providing a nutrient source to these recovering forests (Bull 1994, Kennedy and Quinn 2001) without presenting an excessive fire hazard (Brown et al. 2003). At the same time, tree mortality rates were high on these sites and only 16.4% of stems had fallen at the time of the study on burned sites and 11.5% on gray sites. We expect down woody fuel accumulation to continue as currently standing snags fall in the future. CWD loadings peak approximately 8-12 years post-wildfire (Everett et al. 1999, Passovoy and Fulé 2006, Roccaforte et al. 2012) suggesting that CWD loadings will continue to rise, perhaps beyond recommended ranges, over the next several years.

Smaller diameter woody fuel loadings varied by disturbance and lower loadings were observed on burned sites compared to bark beetles affected sites, consistent with the manner of these disturbances. While wildfires consume small woody material, bark beetle-induced mortality redistributes woody material, eventually redistributing fuels from the canopy to the forest floor (Schoennagel et al. 2012, Donato et al. 2013).

Though small woody fuel loading was consistent with fuel accumulations seen in other bark beetle affected sites, our findings were variable compared to previous wildfire studies. 10- and 100-hr fuels did not increase shortly after bark beetle disturbances (when in the red phase) and 1-hr fuel loadings in both phases (red and gray) were similar to control stands, like Hoffman et al. (2012). While mean 1-hr fuel loadings were lower on burned sites compared to gray phase sites, they did not vary between burned and control sites, similar to the findings of Stevens-Rumann et al. (2012). Inconsistent with other fuels studies following wildfires (Passovoy and Fule 2006, Stevens-Rumann et al. 2012) the 10- and 100-hr fuel loads were similar on burned areas and control sites. These varying results compared to published literature may be a result of time-since disturbance due to the high proportion of standing dead material that has not yet contributed to smaller diameter downed woody fuel loading in our stands. Conversely these varied results may be due to the Brown (1974) methods employed for these smaller woody debris size classes. These methods have come under recent scrutiny for their weakness in capturing fuel variability (Keane et al 2012).

CONCLUSIONS

Disturbance types impacted vegetation response and fuel structure. We did not find evidence of compounding effects of bark beetles followed by wildfires on fuel accumulation nor on tree seedling densities. Rather we note that the post-disturbance landscape is largely controlled by the changes resulting from high-severity wildfires. We only looked at areas with high tree mortality (>80%) within each wildfire, thus tree mortality and fire effects following the wildfire may have masked the structural changes from previous bark beetle outbreaks. In our study the most recent disturbance, whether bark beetle mortality or wildfire, creates similar vegetations responses and downed woody surface fuel complexes regardless of the previous disturbances.

Given the lack of empirical data on these interacting disturbances, especially in dry-mixed-conifer forests, multiple future studies are warranted. First, additional data collection on herbaceous and shrubby fuel components would be beneficial in improving our understanding of the fuel complex in these various disturbance types. Second, examining interactions between these two disturbances at various burn severity levels, rather than only at high severity, would address the possibility of interaction effects across the range of conditions. The impacts of bark beetle activity may be masked and could be observed in moderate to low severity burned areas by only observing high-severity areas of a wildfire. Third, given our understanding of the impact of wildfire severity at different phases of bark beetle outbreaks (Simard et al. 2012, Hoffman et al. 2013), examining the post-fire effects of bark beetle and fire interactions at different outbreak phases is critical to understanding long-term ecological trajectories. Finally, as with all post-disturbance empirical studies, observations post-disturbance limits our understanding of the initial cause of these disturbances and we cannot address questions of why these areas experienced wildfire and/or bark beetle outbreaks and the reason for varying severities. We are limited to observing current stand structure and surface fuels but cannot directly address causality of these changes nor make inferences on fire behavior or outbreak dynamics. Repeated measurements studies examining changes in fire behavior and whether these changes are a result of outbreak induced changes in stand structure or fire weather would address the causal questions (Hoffman et al 2013).

As temperatures increase with climate change both bark beetle outbreaks and fire events are expected to occur more frequently and over larger areas especially in dry mixed-conifer forests (Bentz et al. 2010, Westerling et al. 2011). We expect to see increasing overlap of these disturbances in space and in quick succession. Our study increases the understanding of interactions between these disturbances in dry mixed conifer forests. This knowledge informs managers on stand specific interactions and helps inform and validate model simulations the scientific community has heavily relied on to answer questions about disturbance interactions.

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TABLES

Table 2.1: Study fire information: fire dates fire size, and percent in each burn severity classification

Fire	Fire Start Date	Size (ha)	Unburn to low (%)	Low (%)	Mod (%)	High (%)	Area mask (%)*
Rombo Mountain	30/July/07	11,418	26	15	13	16	30
Middle Fork (Lightning)	17/July/07	2,851	33	22	16	6	23
Poe Cabin	18/July/07	24,160	8	35	38	19	0
Rattlesnake	13/July/07	41,485	36	35	16	13	0

* Non-processing area masks: areas in either pre-fire or post-fire reflectance imagery containing clouds, snow, shadows, smoke, significantly sized water bodies, missing lines of image data, etc.

FIGURES

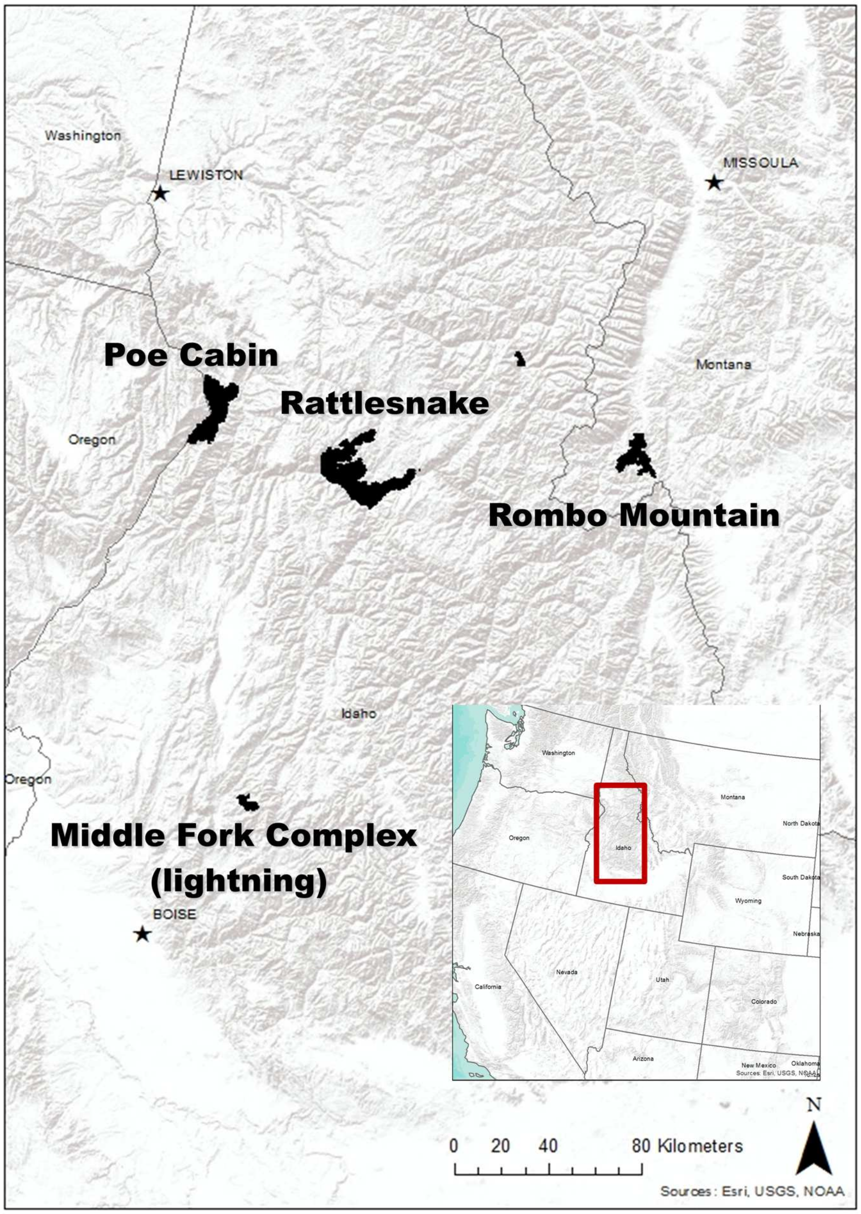

Figure 2.1. Map of the four wildfires studied: Poe Cabin, Rattlesnake, Rombo Mountain, and Middle Fork Complex (Lightning) fires in central and northern Idaho and western Montana. All were in dry, low elevation mixed-conifer forests in areas with prior bark beetle-induced tree mortality as mapped by the Aerial Detection System.

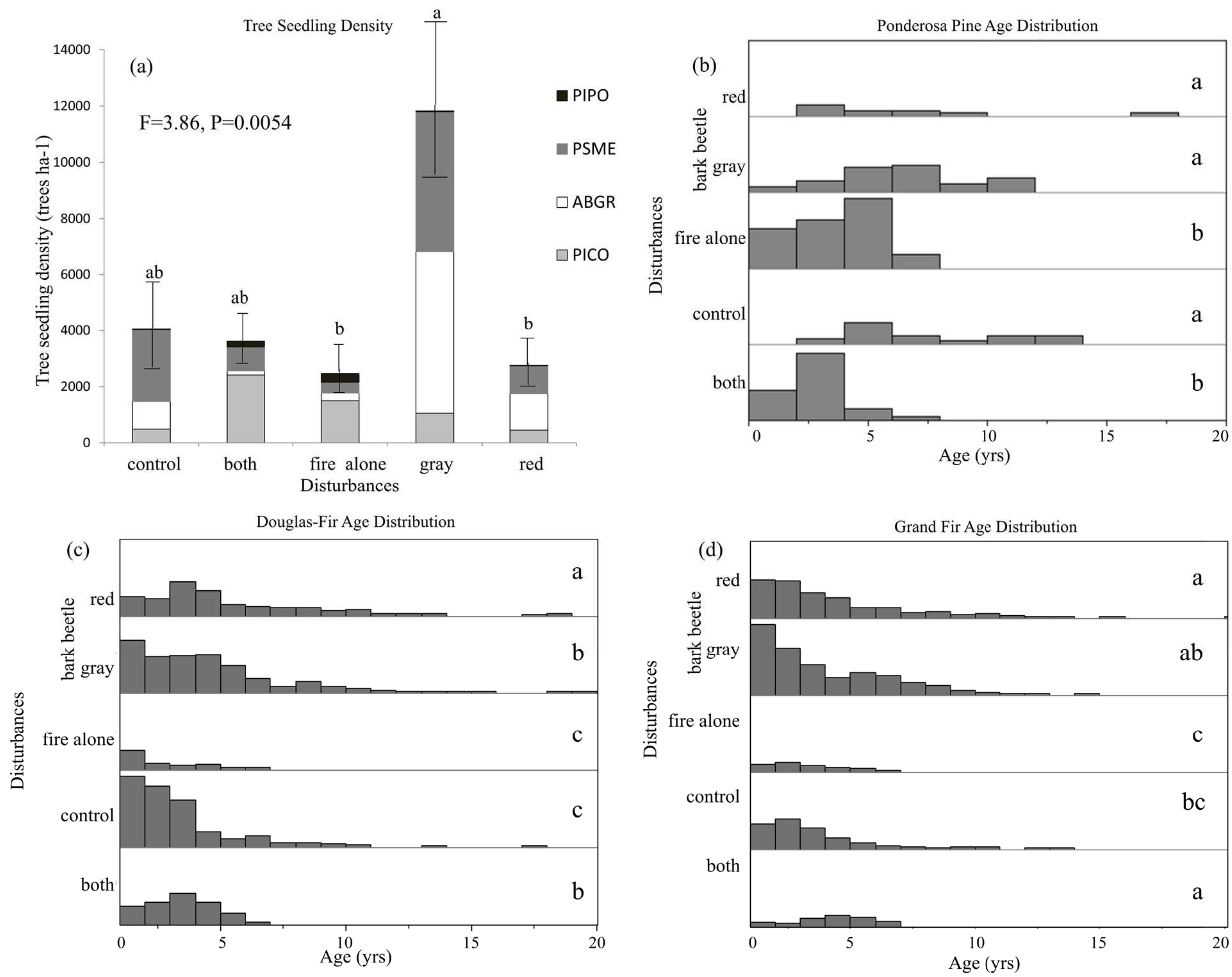

Figure 2.2. Seedling density and age distribution by species. (a) Seedling density per hectare for each disturbance, broken up by the four most dominant species: lodgepole pine (PICO), grand fir (ABGR), Douglas-fir (PSME), and ponderosa pine (PIPO). Error bars indicate the standard error for each disturbance type and F and P values are the results from the ANOVA (N=180). Histograms of seedling age distributions of (b) ponderosa pine (N=200), (c) Douglas-fir (N=1275), and (d) grand fir (N=1060). Letters above or next to each bar indicate results from a Tukeys HSD test.

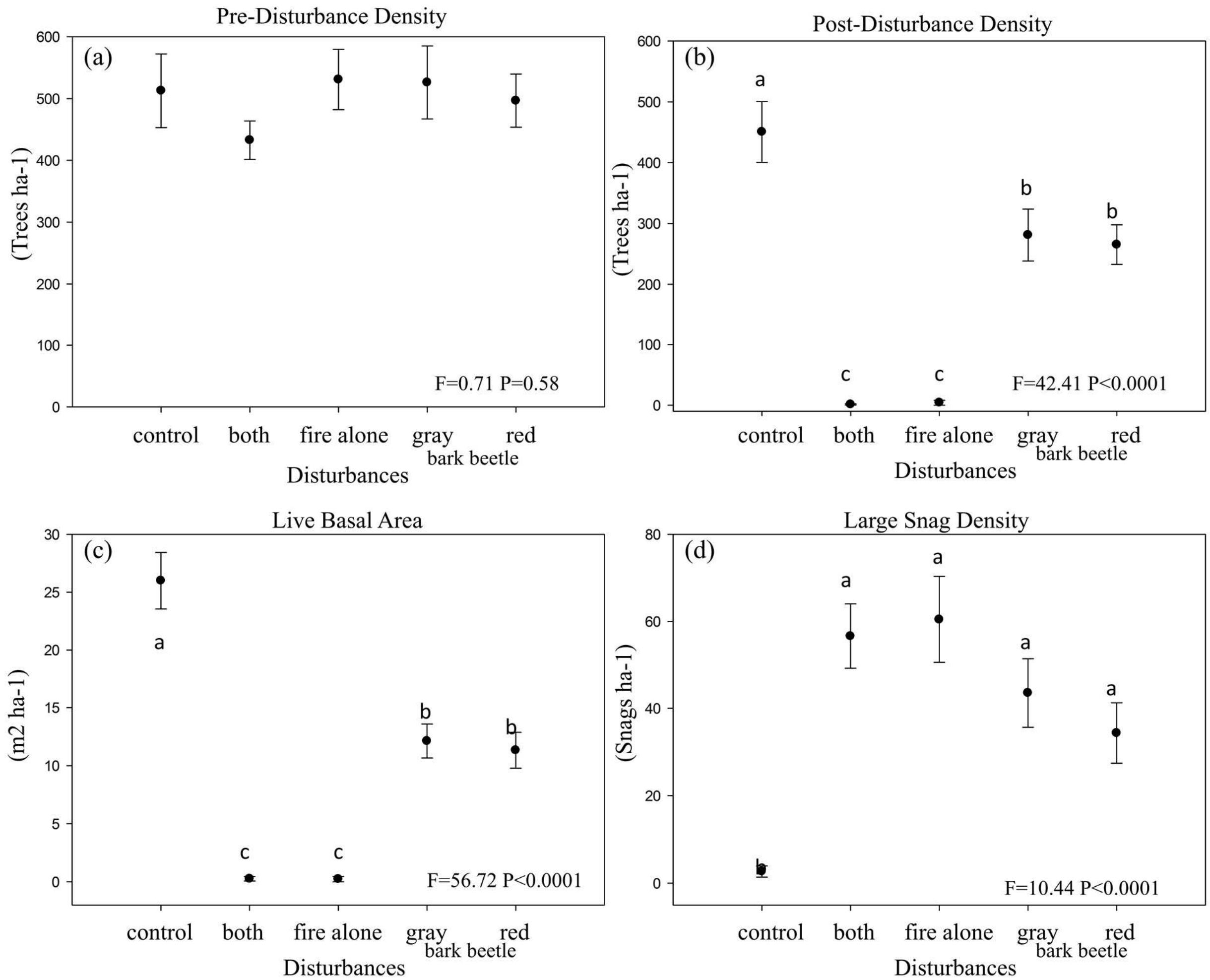

Figure 2.3. Stand structure characteristics from the five disturbances across all four fires. (a) Pre-disturbance tree density (trees>5cm DBH), (b) post-disturbance live tree density (trees >5cm DBH), (c) live basal area, and (d) large snag (>40cm DBH) density. Error bars indicate the standard error for each disturbance type. F and P values from the ANOVA (N=180) are also provided in each figure and letters above each point indicate results from a Tukeys HSD test.

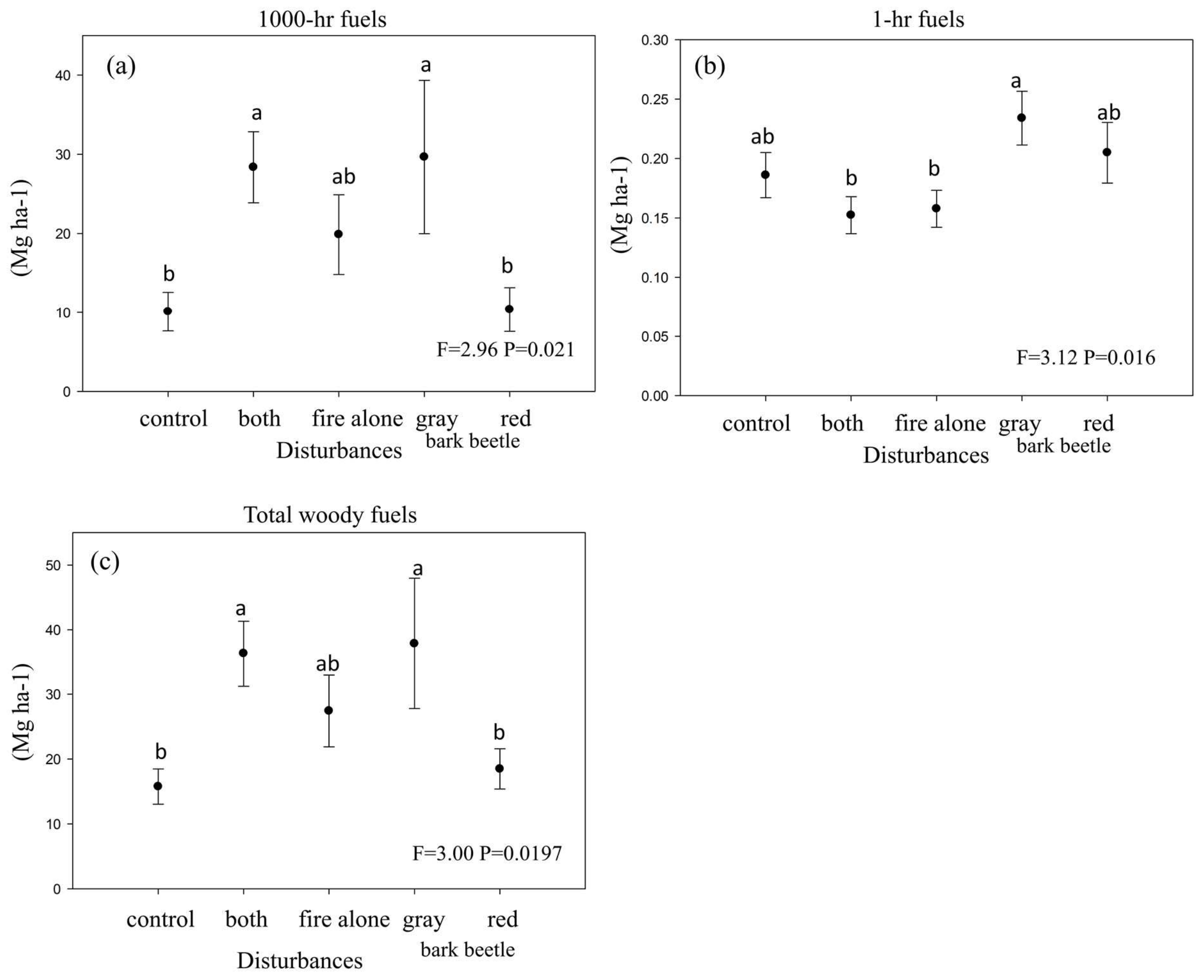

Figure 2.4. Woody surface fuel loadings of those size classes with significant differences between treatment groups: (a) 1000-hr fuel loadings (>7.62cm diameter), (b) 1-hr fuel loadings (<0.64cm diameter) and (c) total woody fuels. Error bars indicate the standard error for each disturbance type. F and P values from the ANOVA (N=180) are also provided in each figure and letters above each point indicate results from a Tukeys HSD test.

CHAPTER 3: FOREST ECOSYSTEM RESILIENCE WITH REPEATED WILDFIRES

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Submitted to Ecological Applications

ABSTRACT

Repeated wildfires influence forest structure, but their ecological effects and implications for resilience are not well understood. Given projections for more large fires in the future, repeated burns, such as those we study in the forests of US northern Rockies, are harbingers of the future. Thus, we were interested in how repeatedly burned areas at (1) different burn severities, (2) order of burn severities and (3) years between wildfires influenced forest recovery and fuel complexes. In 2014, we measured basal area, tree density, canopy closure, surface fuels, and tree seedling density in stands that burned one to 18 years before the subsequent 2007 wildfire. Repeatedly burned forests had less woody surface fuels and lower tree seedling densities compared with forests that only experienced one recent wildfire, which may result in different long-term recovery trajectories and effects of future fires. Time between disturbances had little effect, but order of burn severity (high followed by low severity compared with low followed by high severity) did influence forest characteristics. When low burn severity followed high, forests had lower canopy closure and total basal areas with fewer tree seedlings than when high burn severity followed low. Repeatedly burned areas are recovering differently than sites burned once, which we explore in a conceptual model as possible alternative pathways of forest dynamics. Repeatedly burned areas meet some vegetation management objectives, with reduced fuel loads and moderate seedling densities. These differences in recovery trajectories have implications not

only for the ecological resilience of these forests, but may reduce fire intensity and burn severity of subsequent wildfires, especially in the short term.

KEYWORDS: repeated fires, mixed-conifer forests, tree regeneration, fuels, forest structure, recovery trajectory, succession

INTRODUCTION

Wildfires serve an important ecological function in many ecosystems by consuming biomass and changing vegetation structure and composition. However there is concern about the recent increase in large wildfires across the western US (Westerling et al. 2006, Dennison et al. 2014) and the ecological effects repeated fires will have (van Wagtendonk et al. 2012, Parks et al. 2015). Further, we will likely see an increase in repeatedly burned areas, as there is evidence that climate change and past fire suppression policies will result in more extensive fires in the coming decades (Littell et al. 2009, Marlon et al. 2012, Stephens et al. 2014). Past wildfire perimeters can be used as a tool to hinder the progression of future wildfires (Teske et al. 2012, Parks et al. 2015) and to achieve vegetation management goals (Collins et al. 2009, Stevens-Rumann et al. 2012), but we need a better understanding of how forest fuels and tree regeneration respond to repeated fires to support ecologically-based fire management.

Recently burned forested landscapes can be resilient and resistant to future wildfires due to reduced tree density and surface fuel loads (Holling 1973, McKenzie et al. 2011), but increased forest density and fuels after a century of fire suppression has altered some of these fire resilient qualities of western US forests (Hessburg et al. 2005, Naficy et al. 2010). "Resilience" is an ecosystem's ability to recover to a similar structure and species composition from a disturbance and "resistance" is an ecosystem's ability to remain relatively unchanged from a disturbance (Holling 1973, Buma and Wessman 2013). We focus on two aspects of resilience/resistance: tree density, tree species abundance) and surface fuels as these components not only impact future forest structure, but also how future wildfires will burn through and affect forest ecosystems (Binkley et al. 2007, Churchill et al. 2013). Few researchers evaluate forest resilience and resistance to repeated wildfires, yet it is critical to understand if and how burned forests will interact with future large and potentially severe wildfires influenced by changing climate (Rogers et al. 2011, Marlon et al. 2012).

Repeated wildfires in a short period may result in a species shift (Turner et al. 1993, van Wagtendonk et al. 2012). If repeated wildfires result in high mortality of large trees and tree seedlings (Agee 1993) they could leave once forested landscapes without the potential to regenerate (Agee 2005, Johnstone and Chapin 2006). Turner et al. (1993) posited that if wildfires repeat before tree seedlings reach sufficient maturity to provide seed and/or survive fires, a forest may lose its ability to recover to a similar forest structure or perhaps a forested landscape at all. Our understanding is limited of when and where repeated fires will foster “resilient” forests or become alternative vegetation states.

Ecosystem resilience following repeated disturbances varies with vegetation type (Price et al. 2012, Haire et al. 2013), vegetation response (Peterson 2002) and burning conditions (Moritz et al. 2011). Van Wagtendonk et al. (2012) reported that changes in the effects of subsequent fires were partly due to shifting species compositions from forest to shrubland as a result of an initial wildfire burning with high severity in mixed-conifer forests of California. Forests have transitioned to alternative and persistent vegetation states following high severity or high frequency wildfires in California, the southwestern US and Australia (Savage and Mast 2005, van Wagtendonk et al. 2012, Fletcher et al. 2013). Conversely, Donato et al. (2009) found in southwestern Oregon that mixed-conifer sites with a short interval (15 years) between high severity wildfires maintained the same relative species abundances of trees and understory vegetation as sites with only one wildfire in the past 100 years, demonstrating strong plant community persistence. Further, Larsen et al. (2013) found that in ponderosa pine-dominated forests of the northern US Rocky Mountains, repeated wildfires burning with low to moderate burn severity helped maintain historical forest structure and composition by killing lodgepole pine (*Pinus contorta*) seedlings that dominated the understory after an initial fire event. These mixed results demonstrate the need for further understanding of how burn severity may impact forest recovery, species shifts, and resilience in the face of future disturbances. This is especially important as fire managers have the policy flexibility and the mandate to manage fires consistent with the vegetation management goals (USDA and USDI 2009). Managers are beginning to use previous fires as fuel breaks (Bobby Shindelar pers. comm., Parks et al. 2015), but they could also use wildfires as tools to foster forest resilience.

We collected empirical data in 2014, on a chronosequence of wildfires that burned one to eighteen years prior to burning again in two adjacent large wildfires in 2007. This study was performed to better understand resilience to future wildfires through the questions: 1) How do fuel complexes and tree regeneration differ between areas burned in a single fire event and areas that have experienced two wildfires in recent history? 2) How does the order of burn severity (high burn severity followed by low burn severity versus low burn severity followed by high) impact fuels and tree regeneration? 3) How does time-between wildfires influence fuels and tree regeneration following subsequent wildfires? and 4) Within burn perimeters, what distinguishes areas of increased forest resilience from areas that may be following an alternative successional pathway as a result of these repeated wildfires? We present a conceptual framework informed by theory and discuss implications for management of wildfires for forest resilience.

METHODS

Site selection and field methods

To assist in site selection we used Monitoring Trends in Burn Severity (MTBS) data. USDA Forest Service's Remote Sensing Applications Center developed the MTBS data, available nationally, which is comprised of remotely sensed data products for large wildfires (> 405 ha in the western US; Eidenshink et al. 2007). MTBS products include burned area delineations (referred to here as fire perimeters) and burn severity data layers interpreted by experts from LANDSAT 5 TM and LANDSAT 7 ETM+ satellite imagery at 30-m spatial resolution.

We selected sites using fire perimeters and categorical Relative Differenced Normalized Burn Ratio (RdNBR), a burn severity metric from MTBS data and Environmental Site Potential (ESP) layers from LANDFIRE. We define burn severity here as the long term effects of fires on vegetation and soils (Morgan et al. 2014). We selected only ESP sites that were identified as Middle Rocky Mountain Montane Douglas-fir Forest, Subalpine Mesic-wet Spruce Fir Forest, and Subalpine Mesic-dry Spruce Fir Forest (Rollins 2009). We grouped these into two "forest types": a "montane" forest type with only the Montane Douglas-fir Forest and a "spruce-fir" forest type with the latter two ESP types. Twenty-one wildfires from the past 25 years adjoined or were reburned in either the Cascade Complex Fire or East Zone Complex Fire that burned in 2007. These two 2007 wildfires burned over 242,000 ha on the Payette and Boise National Forests in central Idaho and were chosen due to the extensive overlap with previous wildfires and the limited pre- and post-fire logging and planting. Of the twenty-one possible adjoining and/or overlapping pervious wildfires, we selected six fires for our study (Figure 1) because they spanned multiple years since wildfires, represented relatively large areas of repeated fires, and were accessible for field sampling.

All sites had Idaho batholith as the soil parent material, which is granitic, resulting in coarse-textured, poorly developed soils (Barker et al. 1983). Elevation on sampled sites ranged from 1000-2400 m and climate is characterized by cold winters (-6 $^{\circ}$C January mean) and warm summers (17 $^{\circ}$C July mean) with 58 cm of mean annual precipitation (averaged over 1900-2008; Climate Division 4 in Idaho; NOAA 2014). The dry forests in this area have a long history of frequent wildfires (burning every 3-30 years) prior to 1900 (Heyerdahl et al. 2008b). However, these areas have experienced a century of fire suppression (Heyerdahl et al. 2008a,b, Morgan et al. 2014). While our study ares likely burned before the 20th century, no large wildfires were recorded since 1950 according to local fire history data compiled by Gibson et al. (2014).

We randomly selected potential sample plot locations within mixed-conifer forests and fire histories. Using categorical RdNBR from MTBS data, we selected sites that burned in either high burn severity or in low to moderate burn severities; we refer to the latter as “low severity”. We sampled in forests that burned once (at either burn severity class), in forests that burned twice, and unburned forests. We sampled across six treatment types (we refer to these as fire histories): 1) burned at low severity in a single wildfire (low), 2) burned at high severity in a single wildfire (high), 3) burned at low severity in the first and low severity in the second wildfire (low-low), 4) burned at low severity in the first and high severity in the second wildfire (low-high), 5) burned at high severity in the first and in the second wildfire (high-high), 6) burned at high severity in the first and low severity in the second wildfire (high-low). To determine the relative abundance of each fire history within our study, we calculated the total area in each of the six fire histories and compared that to the total area of repeated fires.

Potential sample plot locations were identified using random point generator in ArcGIS but constrained to be at least 100 m apart to reduce spatial autocorrelation and >100 m but < 2000 m from a trail or road to facilitate access in the often rugged terrain. We field verified both the fire history through tree mortality and seedling age and the ESP through prefire tree species composition at each randomly generated point. We established eight to eleven plots within each of the six fire histories, on each of the six past wildfires for a total of 307 plots across all wildfires. We excluded all sites that had post-fire planting or logging based on Payette and Boise National Forest records of management actions.

At each point we established a 0.04-ha circular plot at the center of a 30-m long, variable-width belt transect. We recorded the slope, aspect, and elevation at the center of each transect. We laid the 30-m long transect perpendicular to the elevation contour and following the dominant hillslope. The width of the belt transect varied between 1 and 10 m and was determined prior to sampling with the objective of sampling at least 30 tree seedlings of the most dominant species (similar to Stevens-Rumann et al. 2015). If no or few seedlings were initially observed, a 10-m width was set as a default. We recorded data for all tree seedlings within the belt transect, including species, distance between each primary branch node of each seedling, and approximate age of each seedling, which was determined by counting the total number of nodes. We classified a tree as a “seedling” if it established post-fire, regardless of height. We also recorded distance to and species of the 10 nearest live seed sources from the center of the 30-m transect. For distances beyond our laser rangefinder's reading capability (500 m) we recorded trees as > 500 m away.

On each circular plot, we tallied all trees >7.8-cm diameter at breast height (DBH, 1.37 m above the ground) as alive or dead, including fallen trees, to estimate percent tree mortality. Pre-wildfire tree density was calculated by summing fire-killed stems (fallen and standing) with live trees. We also measured DBH and recorded species of all living and dead standing trees (we refer to the latter as snags) > 7.8-cm DBH to quantify post-fire tree and snag density.

To quantify downed woody fuel, we tallied fine woody debris and litter and duff depth following methods outlined by Brown (1974) on a 30-m planar intercept transect at the center of the variable width transect Dead down woody debris was tallied by size classes (0–0.64, 0.65–2.54, 2.55–7.62, >7.62-cm diameter) to correspond with 1-, 10-, 100-, and 1000-hr time-lag classes. Along 6 m of the transect, we tallied 1- and 10-hr fuels and 100-hr fuels were tallied on a 10-m section of each transect. The 1000-hr fuels (coarse woody debris (CWD)) were tallied along the entire 30-m transect; we measured the diameter of each and identified each as sound or rotten (Fosberg 1970). These tallies were converted to biomass estimates using Brown's (1974) allometric equations.

Understory cover and forest canopy closure were also assessed along the 30-m transect. Understory canopy cover as a percentage of total ground cover was visually estimated for all graminoids, forbs, tree seedlings, shrubs, bare soil, and rock on the central 10 m of the transect in 2-m segments and averaged for each site. Overstory tree canopy closure was averaged from 3 measurements taken every 10 m along the transect using a spherical densiometer at breast height and facing upslope.

Data Analysis

We performed multivariate analysis to examine overall forest structural differences among fire histories using a Multi-response Permutation Procedure (MRPP; Mielke and Berry, 2001) in an Excel macro (King 2008). For this analysis we examined the effect of fire history on 11 variables: litter depth, woody fuel biomass grouped into fine woody debris (1-, 10-, 100-hr fuels; <7.62cm) and coarse woody debris (1000-hr sound and rotten fuels; >7.62cm), overstory tree canopy closure, live tree density, total basal area (live and dead standing trees), understory cover by percent (4 categories: forbs, grasses, bare soil and rock, shrub and tree seedling), and tree seedling density. We grouped woody fuels into only fine woody debris (FWD; < 7.62-cm) and CWD so the model would not be primarily driven by multiple categories of surface woody fuels. We excluded duff depth from analysis because duff layers were absent in many of our samples. Prior to analysis, we standardized data by dividing each observation by the mean for that variable. When overall significant differences were detected, we used Peritz Closure (Petrondas and Ruben, 1983) multiple pairwise comparisons to separate fire histories multivariate means. For all pairwise comparisons, we made a Bonferroni's correction of our Type 1 error to reflect the total number of comparisons (thus α=0.05/15=0.0033).

After completing the multivariate analysis, we performed univariate analyses to examine how individual forest characteristics differed among fire histories to better understand specific drivers of increased or decreased resilience. For this analysis we used a two way analysis of variance (ANOVA) with year (df = 4) and fire history (df = 5) as the main effects. We tested for assumptions of normality and equal variances visually on residuals as well as with a Shapiro-Wilks and Modified Levene test. Seedling density plus 1 was log transformed to correct for its non-normal distribution. We used a Tukey's HSD test to assess significant differences among fire histories when significance was observed in the overall test. To assess possible alternative causes for variation in tree seedling density we also tested the effect of distance to seed source, elevation, slope, forest type, aspect class (e.g. N, NW, W, SW, S, SE, E, NE), and slope across all fire histories.

Finally, to discuss resilience and possible alternative forest vegetation recovery trajectories, we compared tree seedling densities on plots that burned once and those that burned twice. We identified sites with "low seedling density" as those that had ≤ 100 tree seedlings ha^{-1} as this represented less than 5 seedlings observed on a single plot and is well below seedling density guidelines for restoration (Allen et al. 2002, Bailey and Covington 2002, Metlen and Feidler 2006) and for planting on these National Forests (Payette National Forest Plan 2003, Boise National Forest Plan 2010). "Moderate seedling density" ranged from 101-1000 seedlings ha^{-1}, as this did not exceed pre-fire tree densities on our sites and encompassed ecological planting guidelines of 225-500 seedling ha^{-1} (Payette National Forest Plan 2003, Boise National Forest Plan 2010). "High density" was categorized as any sites that exceeded 1000 seedlings ha^{-1} (Figure 2).

RESULTS

Forest structure differed with fire history. The MRPP on the 11 forest structure variables was significant ($P<0.0001$), and pairwise comparisons demonstrated significant differences ($P<0.0001$) among all fire histories except for three (Table 1). Areas that burned once at low severity and areas that burned twice at low severity did not differ. Areas that burned once at high severity were similar to areas that burned low-high. Meanwhile forests that burned at high burn severity followed by low burn severity (high-low) were similar to forests that burned twice at high burn severity (high-high).

Tree density, seedling density and fuels varied among fire histories, with different patterns than overall forest structure. First, pre-fire tree density did not vary significantly between fire histories ($F=0.67$, $P=0.86$), while post-fire tree density was higher on low and low-low fire histories than all other fire histories ($F=32.26$, $P<0.0001$). Vegetation and surface fuel variables, including litter depth, 1-, 10-, 100-, 1000-hr sound, 1000-hr rotten, fine and coarse woody debris, canopy closure, live tree density, bare ground cover and seedling density) differed significantly ($P<0.01$) among fire histories, with the exception of several understory cover classes, including shrubs and tree seedlings, graminoids, and forbs (Table 2).

The species composition of tree seedlings largely mirrored that of the pre-fire species composition. Tree seedlings were largely comprised of lodgepole pine, ponderosa pine, Douglas-fir, and subalpine fir, with a minor component of western larch (*Larix occidentalis*), Engelmann spruce (*Picea engelmannii*), quaking aspen (*Populus tremuloides*) and grand fir (*Abies grandis*). The notable exception was the abundance of lodgepole pine seedlings in sites previously dominated by ponderosa pine and Douglas-fir, especially in areas only burned one, either in 2007 or in one of the previous fires but did not burn repeatedly since 1984.

Tree seedling density was negatively correlated with distance to seed source and slope and positively correlated with elevation ($F > 10.0$, $P < 0.0001$ for all), but these relationships did not vary across fire histories nor with number of years between fires. Tree seedling density was not correlated with aspect class ($F = 0.60$, $P = 0.44$). We found many more tree seedlings in spruce-fir forest types than in montane forest types ($F = 95.31$, $P < 0.0001$) with median densities of 1200 seedlings ha^{-1} compared to 67 seedlings ha^{-1}.

The effect of years between wildfires was mixed. Between wildfire years, 1000-hr fuel loadings varied ($F = 3.55$, $P = 0.016$), with the greatest loadings on the older initial fires and the lowest loadings on fires one year apart. Fine fuels, including 1-, 10- and 100-hr size classes, did not vary with year between fires ($F < 2.08$, $P > 0.1$). Canopy closure on repeatedly burned sites was significantly higher in forests that had one year between wildfires than on all longer time-between wildfires ($F = 19.65$, $P < 0.0001$). However, no differences were detected among sites burned in older initial wildfires (2000, 1994, 1989) and again in 2007. Tree seedling density generally declined with years between wildfires, with greatest tree seedling densities in forests that burned one year apart and the lowest in forests that burned 13 and 18 years apart ($F = 22.22$, $P < 0.0001$). Understory vegetation cover by class (grass, forbs, shrub and tree seedling, and bare and rock) also varied over time ($P < 0.001$), however the variability of each class did not follow a clear increasing or decreasing trend with years between fires.

DISCUSSION

Lower tree seedling densities and fuels loadings across repeatedly burned sites compared to once burned areas suggests that the initial wildfire effects will continue to alter vegetation responses following a subsequent wildfire even when there is 18 years between wildfire events. Additionally, the order of high versus low severity burns influenced overall forest structure and vegetation response. Previous wildfires not only have the potential to impact subsequent burn severity (Holden et al. 2010, Parks et al. 2014) but also the subsequent post-wildfire vegetation recovery and forest structure. These differences in forest trajectories between fire histories have implications for resilience to future wildfires.

Areas of repeated wildfires have lower fuel loads and tree seedling density

Several key differences emerged between areas of single and repeated disturbances. Overall structure of forests burned low-low and others burned once at low severity were similar with two notable exceptions. First, though pre-fire densities were similar on once and twice burned sites, as expected live tree densities were lower on repeatedly burned sites (Figure 3; Larsen et al. 2013). Second, we found lower tree seedling density on low-low sites compares to once-burned sites, similar to Larsen et al. (2013). This demonstrates that wildfires burning into past burn mosaics at low burn severity may result in reduced densities of both the overstory and understory tree components. Tree seedling density was low on low-low sites and consisted primarily of lodgepole pine and Douglas-fir while lodgepole pine dominated on sites burned once with low severity as Larsen et al. (2013) also found.

Areas that burned with high burn severity within 18 years of an earlier fire at high burn severity (high-high) had a different regeneration pattern than all other burn severity combinations, except high-low. High-high sites had the lowest tree seedling densities, lowest mean 1-, 10-, and 100-hr loadings, and similar mean CWD to other repeatedly burned stands, which was much lower than single high severity wildfires we sampled. Additionally, high-high reburn sites had the highest bare ground percentage and this did not differ with years between fires. This suggests repeated high severity fires in northern Rockies mixed-conifer forests may take longer to recover than single burned areas or may be moving toward a different ecosystem structure and composition as there is a loss of trees and tree seedling, as well as available seed sources. This is especially true if burned patches are large enough that much of the burned areas are far from seed sources (Donato et al. 2009). These areas may become dominated by grasses and shrubs, similar to high burn severity wildfires observed by van Wagtendonk et al. (2012). Our findings differ from Donato et al. (2009) who found similar tree seedling density on single and repeatedly burned sites, 15 years between fires. This difference may be a result of lower site productivity on our sites compared to those of Donato et al. (2009).

Repeated wildfires at any burn severity reduced the woody fuel accumulation that resulted from the initial wildfire as expected (Larsen et al. 2013), and the highest CWD loads on single high burn severity sites. A second wildfire may help reduce future fire hazards by reducing surface woody fuels, beyond what an initial reintroduction of wildfire or prescribed fire will achieve, similar to evidence from repeated prescribed fires and the intended goals of many prescribed burn programs (Battaglia et al. 2008, Aponte et al. 2014). However, we may not have detected additional differences in fuels, especially in fine fuel biomass, due to the inadequacies in the widely used Brown (1974) methods in capturing the high spatial variability typical of fine fuels (Keane et al. 2012).

Order of burn severity and years between wildfires changed post-fire forest structure

The combination of once and twice-burned severities as well as time between fires influenced tree seedling density, likely due to the differences in live seed source trees after the initial disturbance. The lower seedling density in high-low areas compared to low-high areas may be a result of the early loss of the seed bank and seed sources which are important for continued tree regeneration (Bonnet et al. 2005, Donato et al. 2009). While we do not have field data from between wildfires, we infer from satellite-derived burn severity data that the low burn severity of the earlier fire left remnant live trees on and near sampled sites. In contrast, where fires burned with high burn severity, few nearby seed source trees survived. Thus, if the subsequent fire occurs before regenerating trees are large enough to survive the fire or sufficiently mature to produce seeds, even a low severity surface fire may result in a compounding negative effect of these repeated disturbances (van Wagtendonk et al. 2012). This compounding negative effect was observed when fires followed bark beetle outbreaks as well (Harvey et al. 2013, Stevens-Rumann et al. 2015). Conversely, when overstory trees survive an initial low burn severity wildfire, these trees continue to be a seed source between fires and, potential immediately following a subsequent high burn severity fire.

Canopy structure and surface fuel varied by order of severity. Canopy closure was significantly higher in forests that had the high severity wildfire in the second wildfire. This difference in canopy closure measurements was largely a result of standing dead material and likely a product of time-since-severe wildfire as much of the standing dead material created in 2007 had not yet fallen seven years post-fire as documented elsewhere (Everett et al. 1999). Differences in canopy closure between low-high sites and high-low sites were consistent with fall rates of trees killed by the earlier wildfires (Holden et al. 2006, Flannigan et al. 2004). However, surface fuel loadings did not differ between high-low forests and low-high forests, indicating that the subsequent fire, regardless of severity, alters post-fire downed woody fuel accumulation (Everett et al. 1999, Passovoy and Fulé 2006). The reduction in surface fuels and differences in canopy closure indicates that another fire may be less likely to burn severely in these already repeatedly burned landscapes (Collins et al. 2009).

The relationship of tree seedling density to years between fire, while time dependent, may also be a product of site- and climate-specific factors, as we also saw differences in understory cover. Tree seedling density decreased the longer time since initial fire, with the exception of 1994 repeatedly burned forests which had the lowest tree seedling density of all time-between wildfires. Some variability in seedling density may be explained by the climatic conditions in the years following wildfire (Brown and Wu 2005, Brown 2006). Mean annual precipitation was lower than the 30-year average during the five years following both the 1989 and 1994 fires, potentially contributing to lower seedling density than observed on the 2000 and 2006 fires. Also, 1994-1998 was hotter than the 5 years following the other fires, which may have resulted in less favorable growing conditions. The average annual maximum temperature between 1989-1993 was 2°C lower than the maximum temperature in 1994-1998 and there was a two degree difference in maximum temperature during summer months (Western Regional Climate Center last accessed November 18, 2014 http://www.wrcc.dri.edu).

Elevation, forest type and thus microclimate variability may have played a role in the observed seedling and understory cover differences. Tree seedling density was positively correlated with elevation and greater in the spruce-fir ESP sites, and the differences we observe as a "year effect" may be a result of site variability on the 1994 and 1989 fires which had a lower mean elevation (1768 m and 1701 respectively) than those that burned in 2000 (2076 m) and 2007 (1971 m). As variability in understory cover metrics did not follow a clear pattern with time-between disturbances, nor were the observed patterns consistent between variables, the differences were likely a result of varying site conditions of these different wildfire locations.

Limitations

In addition to the above explanations the decreases in understory vegetation and tree seedlings may be more a product of climate and fire history combined. First, our study sites spanned multiple years between wildfires yet we only looked at a single year of subsequent wildfires (2007). Examining differences through time by looking at multiple years since subsequent wildfire will be important, as some of the seedling density and understory cover characteristics may be explained by the climate in the years following the 2007 fires. Fire weather conditions were extreme at the time of both these fires (Hudak et al. 2011), which can limit tree seedling establishment and survival immediately post-fire (Bessie and Johnson 1995). Moreover, drought could be more pronounced in high burn severity areas, limiting longer-term tree seedling establishment and survival (Maher and Germino 2006). This is especially true in high-high sites, where soil burn severity is likely to be high and the lack of tree canopy cover increases solar radiation (White et al. 1996, Zald et al. 2008). Thus, a comprehensive analysis of seedling establishment dates and species and potential climate and/or topographic correlations would improve our understanding of the mechanisms of recovery following repeated wildfires (Gray and Spies 1997, Tepley et al. 2014).

Further, the observed differences between fire histories, here and elsewhere, may be due to grouping low to moderate burn severity. In combining these severity classes, we are likely missing some important site differences between areas of low and moderate burn severity (Stevens-Rumann et al. 2012). Analyzing repeated fires across the full range of burn severity may help us understand the influence of the order of burn severity on forest recovery.

Improved ecosystem resilience in areas of repeated wildfires

We developed a conceptual model of alternative ecosystem states and possible recovery trajectories with repeated fires (Figure 2) based on theory and on our observations and those of Savage and Mast (2005), Donato et al. (2009), and Larsen et al. (2013). This conceptual model can be used for an improved ecological understanding of possible pathways of both fire resilient systems and those areas where ecosystem shifts occur because an area experienced either too frequent or severe disturbances. This model can also serve as a tool for managers to better understand if, when, and how repeated wildfires may further vegetation management objectives of reduced fuel loads and tree density in the short term (< 20 years) post-fire.

Resilience may be increased or reduced by repeated wildfires. According to our conceptual model, multiple pathways can result in similar states but perhaps over different time scales and through different recovery trajectories. For example, state three, which is most resistant to future fires, represents a low density forest that can arise from either low tree seedling establishment following a low to moderate severity wildfire or from moderate tree seedling establishment following a high severity wildfire. In the first case, state three is achieved immediately post-fire as we saw in our low-low sites and these sites are resilient to future fires both in the short term (<7 years) and likely in the long term (>7 years) depending on the density of continued seedling establishment and growth. In the second case, state three is only achieved after years of tree establishment and growth, meaning the resiliency to future fires of these sites may be delayed until trees that established immediately post-fire are old enough to become sufficient seed sources. Tree seedlings need time to establish and grow to a size where they will not all be killed by fire, but as time between fires increases, there is greater vertical continuity of fuels from small regenerating trees to remnant large trees, contributing to high crown fire hazard (Agee and Skinner 2005). For these sites to result in an open stand structure without additional fires or management, seedling density and ultimately tree density must be low, as we observed on many high-low sites. With enough years between disturbances, forest structure may become similar despite different "fire histories" (Hessburg et al. 2005). Therefore, repeated wildfires may increase forest resilience depending on the burn severity of the initial event and whether the next fire occurs before the system recovers (Turner et al. 1993).

Resilience is influenced by the species and density of tree seedlings. Tree regeneration was highly variable and observed trajectory pathways varied accordingly. Differences in tree seedling density between areas of single and repeated disturbances indicate possible alternative ecosystem trajectories. Areas of repeated high severity wildfire had little tree regeneration which may alter or slow future forest recovery. Areas of repeated low severity wildfire also had low tree regeneration, but had an overstory forest structure beneficial in maintaining an open, forest structure where high tree mortality in subsequent fires is unlikely until ladder fuels accumulate (Hudak et al. 2011, Larsen et al. 2013). In these low-low areas, repeated wildfires and/or prescribed fires would help maintain this open forest structure, with mostly ponderosa pine and Douglas-fir seedlings, perhaps without the need of additional active management such as thinning (Larsen et al. 2013). In comparison, areas that burned only once at low severity had the highest tree seedling densities of all burned forests. These sites have increased fire hazards and may be returning to a dense, closed canopy forest structure if not burned again (Hessburg et al. 2005, Larsen et al. 2013), especially where lodgepole pine dominated. While small patches of high tree seedling densities meet some management objectives, large landscapes of dense, closed canopy forests do not contribute to overall forest resistance to future high severity wildfires (Hessburg et al. 2005, Larsen et al. 2013, Harvey 2014).

Surface fuels can play an important role in the resilience of a forest to future wildfires. Often fuel accumulation in recently burned areas is of particular concern to managers given the high accumulations of woody fuels over time from recently killed trees (Passovoy and Fulé 2006). In the event of a subsequent fire, areas of high surface fuels have the potential for severe soil heating and increased time and effort required for fire fighters to control a fire (Brown et al. 2003, van Wagtendonk 2006). Given that we observed lower surface fuels across all areas of repeated wildfires, repeatedly burned areas are showing signs of increased forest resilience using this metric.

Many of the pre-fire forest conditions and thus the resulting fire effects are at least in part due to the long absence of once-frequent fire in these forests(Heyerdahl et al. 2008b) until the large fires of recent decades (Morgan et al. 2008, 2014). As is seen across much of the western US, the pre-fire conditions were unlike historical structures (Hessburg et al. 2005, Nifacy et al. 2010) and the resiliency of repeatedly burned areas in the face of future fires is less certain and highly dependent on vegetation response and fuel loading (Churchill et al. 2013). However, our study demonstrates that though these forests are much changed from historical structures, they retain some elements of a resilient system and are recovering after repeated wildfires. The reintroduction of repeated wildfires, even at high burn severity may enhance the resiliency of these large landscapes by reducing fuel loads and the proportion of high density tree seedling regeneration.

Management implications

Overall, repeatedly burned areas appear to be more resilient to future large fires compared to similar areas that experienced one wildfire within the 18 years we studied. With the exception of high-high sites that are perhaps transitioning into an alternative vegetative state due to the low tree regeneration, other fire histories are regenerating to forested ecosystems. The reduction of both surface woody fuels and lower mean tree seedling density reduces the potential fire hazards (Larsen et al. 2013). In areas that experienced high severity wildfires, post-fire recovery is a long process and resilience to subsequent wildfire is hard to assess 7 years post wildfire. Given a sufficient time for tree growth before an additional wildfire these forests may be more resilient due to reduced fuels and tree density than areas of a single wildfire. However, if another wildfire occurs before tree maturity, high tree mortality is likely and resilience will need to be reassessed. In low-low burned forests the reduction in seedling density and overstory density is restoring more open stand structures for ponderosa pine and Douglas-fir dominated forests which increases the resilience of these forests to future wildfires (Larsen et al. 2013). Managing wildfires to burn into past fires, especially under less severe fire weather conditions that will result low-low and high-low may continue to improve forest resilience to future fires, even if those future fires burn under extreme conditions.

Managers have a choice about how, when, and where to suppress wildfires, so understanding the time frame and types of conditions in which past wildfires will aid in favorable future burning conditions and achieving vegetation management goals is critical for making sound management decisions (USDA and USDI 2009). Many of the repeatedly burned areas had more resilient forest structures and surface fuels compared to areas of only a single wildfire. Where repeated wildfires create diverse forest structures and thus a variety of habitats and recovery trajectories, they may be strategically useful in landscape management and in maintaining landscape forest resilience (Agee 2005, Hessburg et al. 2005, Donato et al. 2009).

Heterogeneous landscapes are critical for maintaining ecosystem function and resilience (Allen et al., 2002, Binkley et al., 2007, Stephens et al., 2010, Churchill et al. 2013, O'Hara and Ramage 2013). Thus, repeated high severity wildfires may not be detrimental at a landscape scale if wildfires are burning with mixed severity resulting in diverse forest structures. However, proximity to seed sources (Donato et al. 2009) or patch sizes of these forests that burned high-high, and overall percentage of these various reburn severity groups should be considered. Of the 242,000 ha burned in the East Zone and Cascade Complexes in 2007, 50 percent burned with low to moderate burn severity and 21 to 30 percent experienced high burn severity. Repeatedly burned areas, within our 6 previous wildfires, comprised a total of 42,390 ha of this large burned landscape. Of the reburned area only three percent of the total area was high-high, nine percent was low-high, seven percent was high-low, and 81 percent was low-low. The forest conditions resulting from previous fires influenced both severity and the vegetation trajectory. Though small, the repeatedly burned areas altered the effects of these larger wildfires such that forests generally burned at lower burn severity, suggesting that forest resilience increased in terms of fuels, remnant forest structures and future forest age structures, as suggested by others (Thompson and Spies 2010, Churchill et al. 2013). In the face of projected fire extent (Westerling et al. 2006, Dennison et al. 2014), repeated fires will happen but these areas may not be of large concern for vegetation management and forest resilience. Managers have the potential to use prescribed burning and manage future wildfires to foster resilience in repeatedly burned areas.

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TABLES

Table 3.1. The results from the multivariate analysis on forest structure. P-values from MRPP pairwise comparisons across all fire histories across all variables (FWD, CWD, canopy closure, live tree density, total tree basal area, understory cover by percent (4 categories: forbs, grasses, bare and rock, shrub and tree seedlings), and tree seedling density. Bolded values indicate significant differences using a Bonferroni's correction (α=0.05/15=0.0033).

	Low	High	High High	Low High	High Low	Low Low
Low	**-**	**$3.35E^{-12}$**	**$4.95E^{-20}$**	**$1.03E^{-10}$**	**$3.97E^{-19}$**	0.1011
High		-	**$1.73E^{-8}$**	0.0052	**$7.36E^{-5}$**	**$2.62E^{-8}$**
H-H			-	**$2.99E^{-7}$**	0.0391	**$1.59E^{-14}$**
L-H				-	**$1.48E^{-6}$**	**$4.21E^{-8}$**
H-L					-	**$6.51E^{-14}$**
L-L						**-**

Table 3.2. Mean canopy characteristics and surface fuels across different fire histories: once burned areas of low or high burn severity and areas of repeated fires of different burn severities (high-high, low-high, high-low, and low-low). Values in parentheses are standard errors and letters in superscript indicate results from the Tukey's HSD test on fire history differences when significant pairwise comparisons were detected. Those sharing the same letters were not significantly different.

		Low	**High**	**High-high**	**Low-high**	**High-low**	**Low-low**
Canopy	Live tree density (trees ha^{-1})	260 (34.7)	15.7 (11.2)	6.2 (4.4)	0 (0)	9.1 (6.9)	159 (21)
	Live basal area (m^2 ha^{-1})	12.7 (1.1)^{a}	0.4(0.3)^{b}	0.8(0.6)^{b}	0 (0)^{b}	0.7(0.5)^{b}	15.1 (1.7)^{a}
	Total basal area (m^2 ha^{-1})	20.6 (1.2)^{ab}	13 (1.1)^{cd}	8.9 (1.5)^{d}	15.4 (1.3)^{bc}	7.1 (1.2)^{d}	22.2 (2.0)^{a}
	Canopy closure (%)	60 (3)^{a}	31 (3)^{c}	15 (3)^{d}	35 (4)^{c}	14 (2)^{d}	58 (4)^{ab}
Fuel depth	Litter depth (cm)	1.0 (0.1)^{a}	0.6 (0.1)^{cd}	0.3 (0.1)^{d}	0.6 (0.1)^{bcd}	0.4 (0.1)^{d}	0.9 (0.1)^{ab}
	Duff depth (cm)*	0.5 (0.1)^{b}	0.1 (0.0)^{c}	0.04 (0.0)^{c}	0.01 (0.0)^{c}	0.01 (0.0)^{c}	0.2 (0.1)^{bc}
Fuel load	1hr (Mg ha^{-1})*	0.6 (0.1)^{a}	0.3 (0.1)^{ab}	0.1 (0.0)^{b}	0.2 (0.0)^{b}	0.2 (0.1)^{b}	0.5 (0.1)^{a}
	10hr (Mg ha^{-1})	2.3 (0.3)^{a}	2.4 (0.4)^{a}	0.7 (0.1)^{b}	1.5 (0.2)^{ab}	1.4 (0.2)^{ab}	1.9 (0.3)^{a}
	100hr (Mg ha^{-1})	4.0 (0.7)^{ab}	4.9 (0.7)^{a}	1.7 (0.3)^{ab}	3.4 (0.5)^{ab}	3.5 (0.5)^{ab}	3.0 (0.5)^{ab}
	1000hr-sound (Mg ha^{-1})	17.9 (2.5)^{b}	34.3 (4.3)^{a}	11.1 (2.9)^{b}	23.2 (3.9)^{ab}	29.6 (6.5)^{ab}	20.6 (5.4)^{ab}
	1000hr-rotten (Mg ha^{-1})	4.2 (1.0)^{bc}	10.5 (2.1)^{ab}	16.1 (2.9)^{a}	12.5 (2.6)^{ab}	10.0 (3.5)^{abc}	1.8 (0.6)^{c}
Understory cover	Bare ground (%)	18 (2)^{d}	31 (3)^{bc}	45 (3)^{a}	30 (3)^{bc}	39 (4)^{ab}	25 (3)^{cd}
	Shrubs & seedlings (%)	12 (2)	12 (2)	14 (3)	21 (3)	16 (4)	16 (2)
	Graminoids (%)	17 (2)	21 (2)	15 (2)	13 (2)	18 (2)	13 (2)
	Forbs (%)	12 (1)	11 (1)	11 (2)	15 (2)	9 (2)	11 (2)

*standard errors of 0.0 indicate very small standard errors (>0.05).

FIGURES

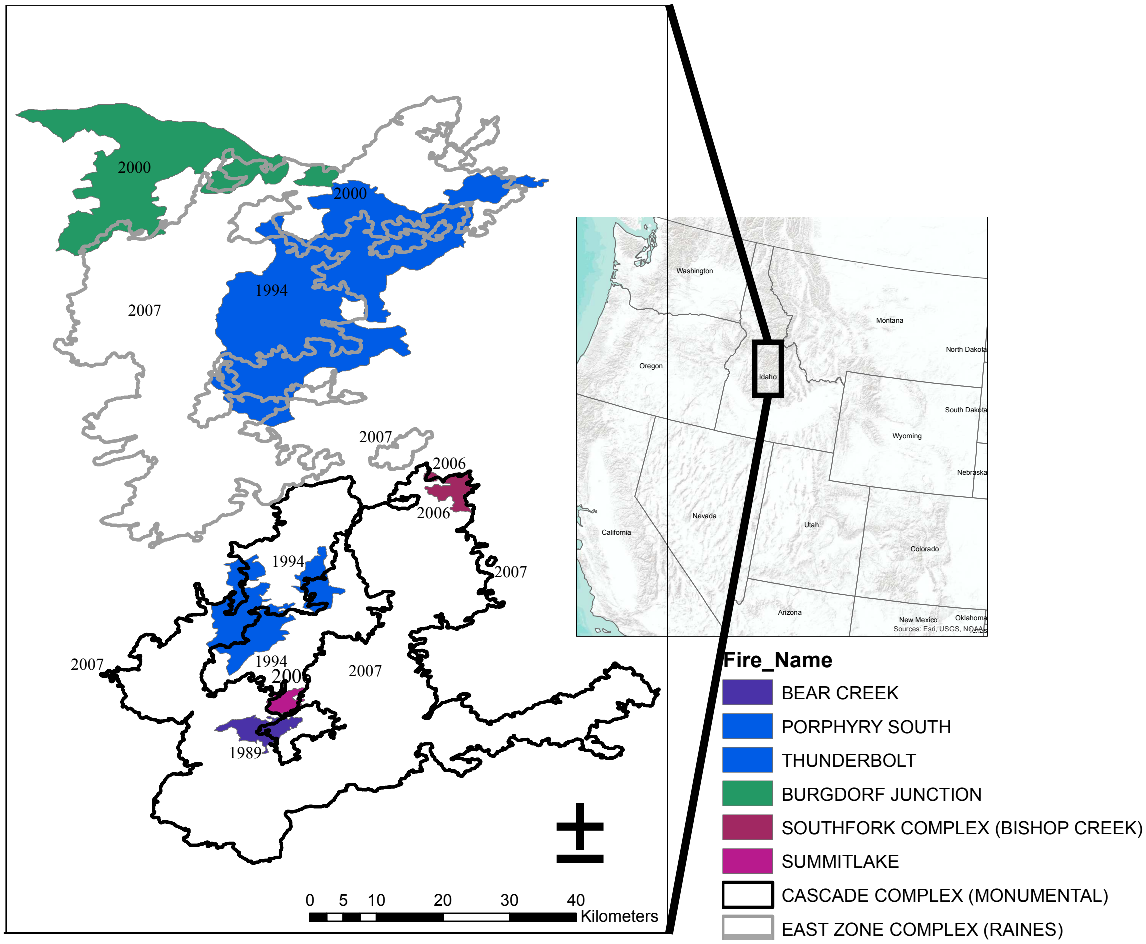

Figure 3.1. Map of eight study fires. Labeled by fire year in map, while the legend shows fire names.

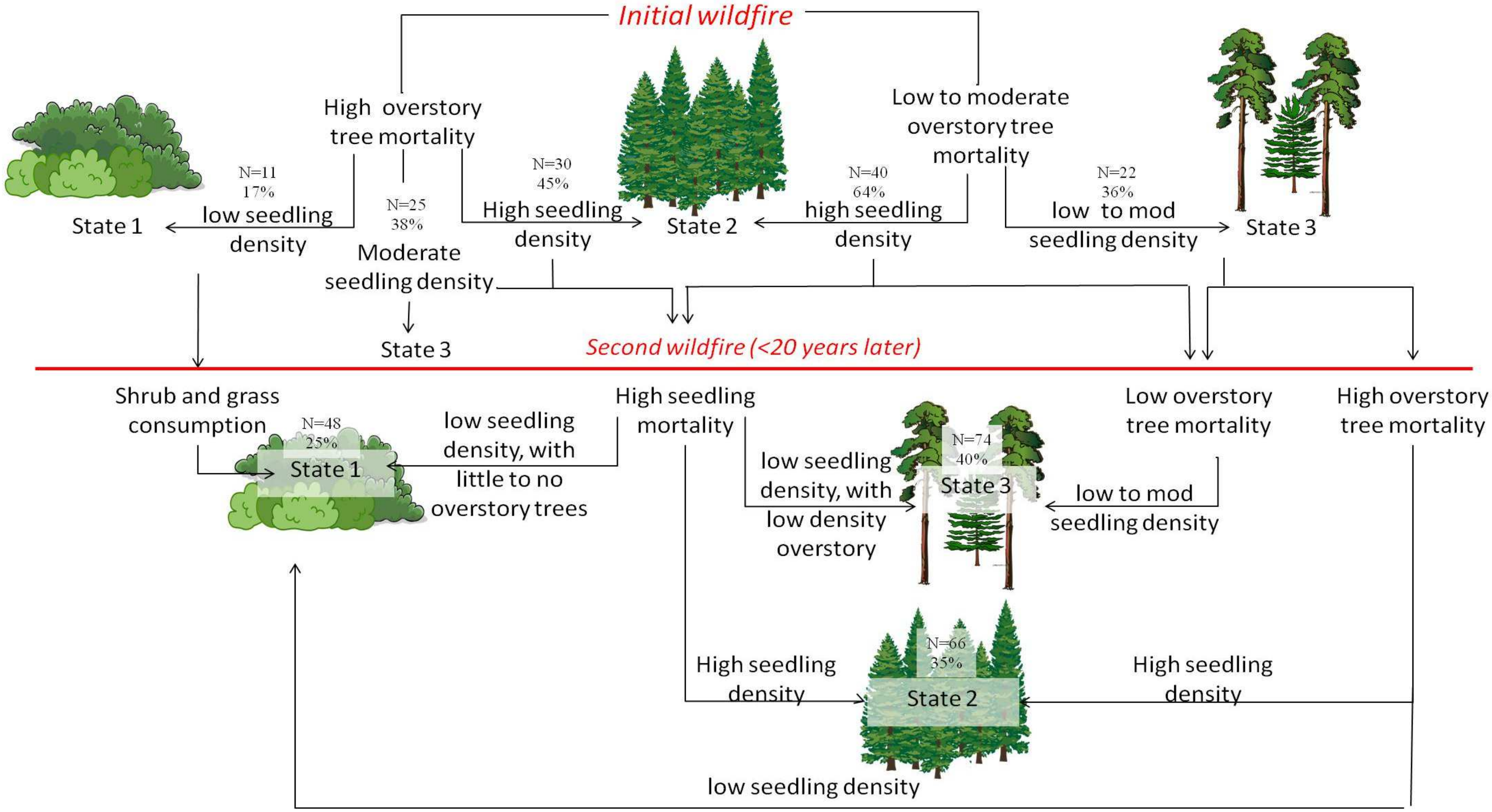

Figure 3.2. Conceptual model of possible forest recovery pathways following single and repeated disturbances. The model is based on tree regeneration and does not include surface woody and herbaceous fuel. State 1 represents a shrub and grass dominated system with little to no overstory. State 2 represents a high tree density, closed canopy forest, most commonly dominated by lodgepole pine in the northern Rockies. State 3 is an open canopy, low density forest of ponderosa pine, Douglas-fir, and/or western larch. State 3 can arise from either a low to moderate level of tree mortality from a single wildfire (right side of the figure), or from low seedling density following a high severity wildfire (left side of the figure). In the first case the overstory forest structure is established immediately following the initial wildfire, in the second case this is a long recovery pathway and is dependent on seedling establishment, density and survival.

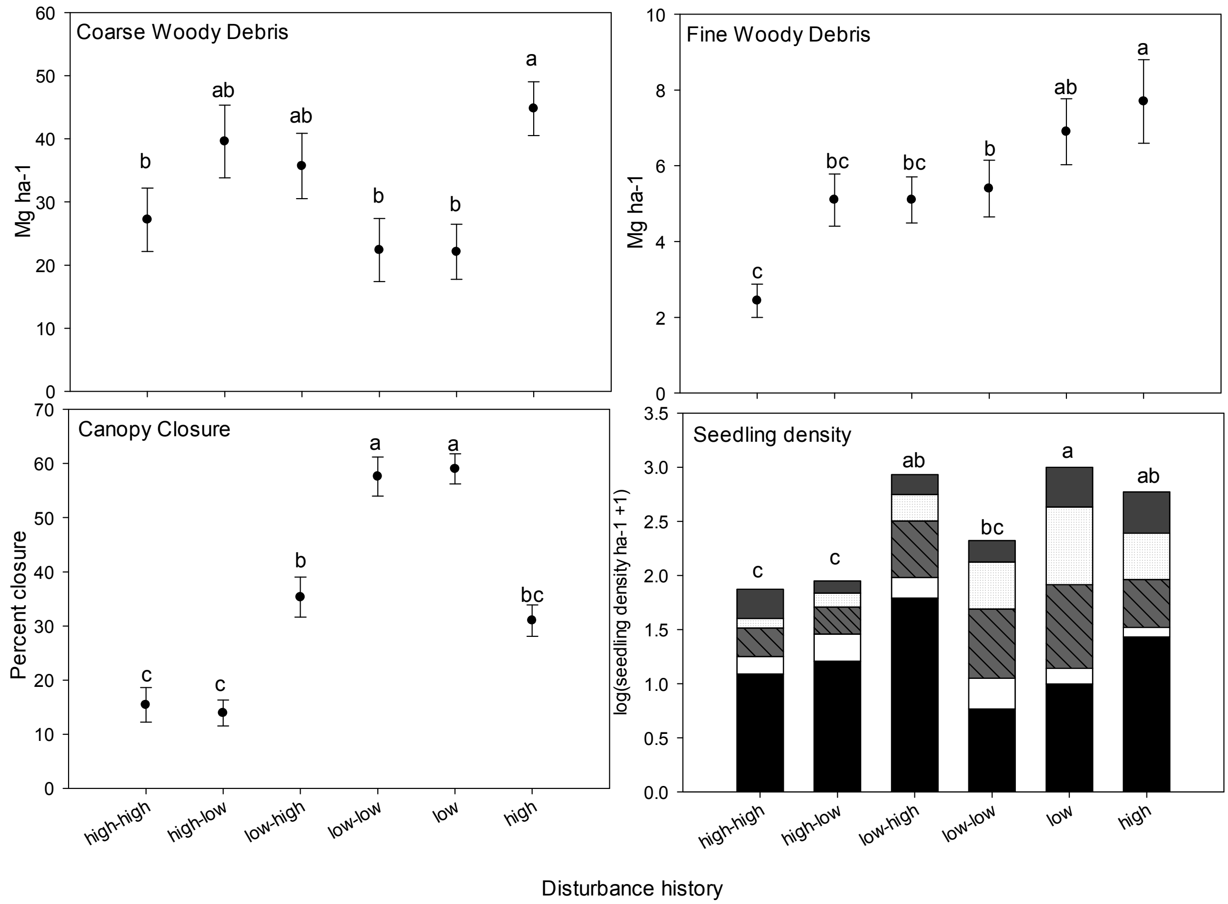

Figure 3.3. Coarse woody debris (> 7.62 cm downed woody material), fine woody debris (< 7.62 cm downed woody material), canopy closure and log of seedling densities by treatment group. Error bars indicate standard error and letters above each point are results from the Tukeys HSD test. In seedling density graph black bars show lodgepole pine densities, white indicate ponderosa pine, gray striped bars represent Douglas-fir, gray spotted bars are subalpine fir, and solid dark gray indicate all other species (grand fir, western larch, aspen and Engelmann spruce).

CHAPTER 4: PRIOR WILDFIRES INFLUENCE BURN SEVERITY OF SUBSEQUENT WILDFIRES

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ABSTRACT

As more large fires interact in coming decades, past wildfires influence behavior, effects, and where subsequent fires burn, with implications for ecosystem resilience and fire management. This study is one of the first to examine how previous burn severity, topography, vegetation, weather, and their interactions influenced burn severity. We examined three wildfires, two in Idaho and one in Washington that were some of the largest-to-date and cost over one billion dollars in fire suppression alone. We used spatial autoregression to analyze burn severity, defined here as the effects of fire on vegetation and inferred from satellite imagery, on two large wildfires in Idaho and one in Washington. These three study fires reburned over 50,000 ha previously burned by 38 wildfires between 1984 and 2006. We found that areas previously burned in the last 23 years, at any severity, had lower burn severity in the subsequent fire. Topography, vegetation, and weather also influenced burn severity, with the best model including maximum temperature on the day of burning, vegetation cover type, slope, and elevation as predictors. These site-specific factors should be considered in decisions about how to use these past burn mosaics in managing fires and forest ecosystems as they experience repeated fires. However, across all study fires and burning conditions within them, burn severity was reduced, suggesting that these previously burned landscapes can effectively mitigate subsequent fire effects even under extreme fire weather conditions under which these three fires burned. Our findings show that reburning within a short time period reduces the proportion of high burn severity, demonstrating the

forests' ability to "self-regulate" against repeated high severity disturbances even when those fires burn under extreme weather conditions.

Key words: burn severity, forest self-regulation, reburn, repeated wildfires, spatial autoregression

INTRODUCTION

Forests of the western US are adapted to fire, but frequency, fire return interval, and the severity influence how forests burn and vegetation recovers following a wildfire. As a self-regulating process, previous fires may limit the progress and/or severity of subsequent wildfires over short time periods, because of limited burnable fuels and changes in forest structure and tree survival (Agee 1999, Peterson 2002, McKenzie et al. 2011, Parks et al. 2014a, 2015). Over the past century, exclusion of all but the largest fires and other human land use changes have influenced forest development over much of the western United States (Hessburg et al. 2005, Naficy et al. 2010). Beginning in 1940, fire suppression was very successful, and most fires were actively suppressed. With the onset of warmer, drier summers and warm springs, the number and size of wildfires is increasing in the western US and globally (Westerling et al. 2006, Littell et al. 2009, Williams et al. 2009). However, after this long fire-free interval in many western forests, our understanding of fire effects and wildfire interactions with previously burned areas is limited yet key to inform management of fires.

Others have found that burn severity of subsequent wildfires was influenced by the burn severity of prior fires, but these studies were in wilderness areas (e.g. Collins et al. 2009; van Wagtendonk et al. 2012; Parks et al. 2014). Burn severity is defined here as the post-fire effects one year following the fire on vegetation and soils (Morgan et al. 2014). In studies of past fire interactions in the Sierra Nevada Range, van Wagtendonk et al. (2012)

and Collins et al. (2009) found that areas previously burned with low to moderate severity in a 30 year period, tended to burn at similar severity in a subsequent fire. However, if an area had previously burned in a high severity fire, a high proportion of the area burned again at high severity in a subsequent fire. They attributed this to the fire-induced shift in vegetation from forests to shrublands rather than simply a function of post-fire fuel accumulation (van Wagtendonk et al. 2012). Similarly, Holden et al. (2010) found that in wildfires 3 to 14 years apart there was a threshold for burn severity above which burn severity is likely to increase in the subsequent fire. Based on satellite imagery, low severity fires often resulted in subsequent low severity fires, but high severity fires resulted in high severity fires (Holden et al. 2010, Parks et al. 2014a). Time-since-previous fire played a role in how effectively a previous fire reduced subsequent fire size (Teske et al. 2012, Haire et al. 2013, Parks et al. 2015) and burn severity (Parks et al. 2014a). Parks et al. (2015) demonstrated that previous fire was an effective barrier for subsequent fires for six years in southwestern US forests and up to 18 years in the northern Rockies. Here we focus on non-wilderness areas as fires in these areas are often in drier forest types than wilderness areas, tend to have the highest suppression cost with both past and present active suppression, have high public interest due to active use, and present the largest scientific uncertainty with respect to effects of repeated fires.

Topography, vegetation, fire weather and climate influence burn severity of wildfires (Schoennagel et al. 2004, Lentile et al. 2007, Dillon et al. 2011, Birch et al. 2015), but whether these variables will supersede or will compound the impacts of previous wildfire burn severity is not well understood. Various researchers have found mixed results regarding the relative importance of top-down drivers of fire, such as maximum temperature, relative humidity, and wind speeds, and bottom-up drivers, such as vegetation and topography. Turner and Romme (1994) and Gedalof et al. (2005) demonstrated that extreme weather conditions can override bottom-up factors, resulting in larger wildfires regardless of fuels and forest types. In contrast, Dillon et al. (2011) and Birch et al. (2015) found that bottom-up factors influences burn severity more than climate and weather. Though multiple researchers have examined bottom-up versus top-down drivers of burn severity, few have analyzed the interaction of these factors in previously burned areas. Especially when wildfires are burning under very hot, dry, and windy conditions, these large and often high intensity fires are most likely to overcome the fuel breaks created by previous wildfires (Pollet and Omi 2002, Graham 2003). To fully understand the self-regulating capacity of these landscapes we must understand when and why past wildfires alter subsequent burn severity and when environmental factors or day of burning conditions override these potential fuel breaks.

In this study, we examined the drivers of burn severity within reburned areas in non-wilderness forests of the interior northwestern US. We studied the Tripod Complex Fire (central Washington), the East Zone Complex (central Idaho), and Cascade Complex Fires (central Idaho), each of which were unusually large wildfires in terms of size, severity and suppression costs relative to those of the last century, and burned through numerous past fires. The effectiveness of fuels treatments, including prescribed fires, have been previously studied on two of these wildfires (Hudak et al. 2011, Prichard and Kennedy 2014), however neither included previous wildfires that may have also modified burn severity. This study was guided by two key questions: 1) Was burn severity of subsequent wildfires influenced by previous wildfires? and 2) Are climate/weather or topographic conditions mitigating factors in the effectiveness of past wildfires to reduce subsequent burn severity? These questions are important for understanding how post-fire conditions will influence subsequent burn severity and when/where these past burn mosaics can be used in wildland fire management that is critical for achieving federal vegetation management goals. Additionally, we speak to how climate, topography, and vegetation interact with past wildfires to influence subsequent burn severity and therefore the self-regulation of forest disturbances. Such self-regulation is least likely and most important when large fires burn under extreme conditions.

METHODS

Study areas

We focused our study on three recent, large wildfires in Idaho and Washington. These wildfires were chosen due to their large size, high suppression costs, and large areas of interactions with previous wildfires. Combined, these three complexes burned a total of 313,000 ha and cost over $1.4 billion dollars (US Secretary of Agriculture 2008). In all three cases these wildfires were complexes comprising of multiple ignitions that burned into one another and were managed as a single fire, thus we treated each as a single study fire.

The 2006 Tripod Complex on the Okanogan-Wenatchee NF in Washington was, at the time, the largest (71,000 ha) fire event in Washington State in over 50 years (MTBS 2010, Prichard and Kennedy 2014). It burned July-September 2006. Over 65% of the area burned was classified as moderate to high severity burns with high tree mortality. The wildfires initiated in high elevation forest of lodgepole pine (*Pinus contorta*) and Engelmann spruce (*Picea engelmannii)* with many dead trees due to recent mountain pine beetle and spruce beetle outbreaks. The wildfires then spread into surrounding mixed-conifer forests of Douglas-fir, ponderosa pine and western larch. As the Tripod Complex spread northeast with prevailing winds, it skirted three 2003 fires, three 2001 fires, and burned a small portion of one 1994 fire (Figure 4.1).

In 2007, the East Zone and Cascade Complex fires each burned over 121,000 ha on the Boise and Payette National Forests in Idaho (MTBS 2010, Hudak et al. 2011). The East Zone and Cascade Complexes burned in July-September 2007 with mixed severity, with 21 to 30% of each wildfire classified as high burn severity (Stevens-Rumann and Morgan *in review*). These two complexes burned through a variety of forest types from subalpine forests and meadows at high elevation to lower tree line dominated by ponderosa pine woodlands. These two wildfires interacted with 31 previous wildfires that burned between 1984 and 2006 (Figure 4.2 and 4.3). These previous wildfires were both throughout the interior of the 2007 fires and along their perimeters. Though the Cascade and East Zone Complexes abutted and could be treated as one continuous landscape, we analyzed these fires separately given their large size and the computational resources required to analyze these large landscapes.

Data

We used data from multiple sources to examine the key drivers of burn severity (Table 4.1). We assessed the impact of previous wildfires by evaluating burn severity using a continuous Relative Differenced Normalized Burn Ratio (RdNBR) which was obtained from the Monitoring Trends in Burn Severity (MTBS) project (Eidenshink et al. 2007). This project provides mapped burn severity for large (>400ha) wildland fires since 1984 in the U.S. at the 30-m spatial resolution. These images are obtained by comparing satellite imagery pre- and one year post-fire with Landsat 4 and 5 Thematic Mapper and Landsat 7 Enhanced Thematic Mapper-Plus (Parks et al. 2014a). We chose to use RdNBR over other satellite imagery metrics of burn severity because it generally a good of a predictor of field-validated burn severity (Miller et al. 2009, Cansler and McKenzie 2012, Prichard and Kennedy 2014) and is especially good for heterogeneous vegetation (Parks et al. 2015). Additionally, field-based composite burn index (CBI) values on the Tripod Complex Fire were highly correlated with RdNBR (R^2=0.71; Prichard and Kennedy 2014). We converted continuous RdNBR values for past fires into categorical variables of “unburned”, “low”, “moderate”, and “high” using thresholds established by Miller and Thode (2007). For our analysis, we used unburned as the baseline for this categorical variable.

We used the MTBS data for three potential predictor variables. First, we assigned categorical RdNBR values for burn severity of past wildfires. Second, time since fire was assigned for each pixel that experienced 2 or more years. For points not previously burned we assigned “100” as time since previous fire. These point may have burned in less than 100 years, but we only had burn severity data available after 1984 thus we assigned a longer time since previous fire value. For points that were reburned more than once (i.e., burned in three or more wildfires between 1984 and 2007), the most recent fire year and burn severity were used to calculate time since previous fire. This occurred on 1,491 pixels (2%) of 68,394 reburned pixels of the Tripod Fire. On the Cascade Complex Fire this occurred on 7,366 pixels (3%) of 281,762 reburned pixels and on the East Zone Complex Fires 22,677 (4%) of 556,007 pixels. Third, to understand possible edge effects, such as fire suppression, on burn severity we also used a distance-to-edge metric which we created as the distance of each study point to the nearest burn perimeter. Although fire management actions during wildfires likely altered fire extent and severity, we did not account for them as the records of actions were incomplete.

To examine the impact of weather on the day of burning we acquired fire progression maps from the Okanogan-Wenatchee, Boise, and Payette National Forests. These progression maps allow us to narrow the time frame within which each point burned to a 10-96 hour window depending on the frequency progression maps were made during the fire incident. We then assigned weather characteristics during each progression interval to each study point based on the date each point burned. During each progression window we assigned maximum and average wind speed in kilometers per hour, which was the open wind speed taken at 6.1m above ground. We also assigned maximum and average air temperature in Celsius, and minimum relative humidity (RH). These data were acquired from nearby Remote Area Weather Stations (RAWS), the First Butte station for the Tripod and the Tea Pot Idaho station for the Cascade and East Zone (Western Regional Climate Center, http://www.raws.dri.edu/, last accessed January 13, 2015). Both stations were on the western border of the fire perimeters and within 5km of the burned edge.

Vegetation and fuels information was derived from LANDFIRE products (Ryan and Opperman 2013). We used 2001 data to reflect the best data for conditions prior to the three study wildfires. Crown height (CH), crown bulk density (CBD), existing vegetation height (EVH), fire regime group (FRG) and canopy cover (CC) were all acquired. We also converted existing vegetation type (EVT) to "cover type" categories, to group similar vegetation types. These categories varied slightly between the different study areas due to varying initial EVTs. For the Tripod Fire these cover types were "lodgepole pine", "ponderosa pine", "subalpine forest", "riparian", dry-mesic mixed-conifer", "Douglas-fir/western hemlock", "deciduous shrubland", "shrubland", "grassland", "no vegetation", and "avalanche area". On the Cascade and East Zone Fires we grouped EVTs into: "lodgepole pine”, “ponderosa pine”, “subalpine forest”, “riparian”, “dry-mesic mixed-conifer”, “deciduous shrubland”, “Douglas-fir”, “shrubland”, “grassland”, “no vegetation”, and “subalpine grassland”. Thus, we reduced the number of categories from 40 to eleven on the Tripod and 39 to eleven categories for the East Zone and Cascade Fires.We used “dry mesic mixed-conifer” as the base contrast for burn severity comparison among these different categorical variables for all study areas.

Topographic and landscape indices were evaluated, including potential incoming solar radiation summarized over one calendar year period (Fu and Rich 1999), elevation (m), hillshade (ESRI 2011), slope (degrees; ESRI 2011), and steady state topographic wetness index (TWI). TWI was derived using Evans’ (2003) script, similar to methods from Gessler et al. (1995) and Moore et al. (1993). Additionally, three topographic position indices were included (slope position, referred to here as “TPI”, ridgetop/ridge-like settings, and valley/valley bottom-like settings). These final three indices were calculated at a 100-m nearest-neighbor and were derived using methods developed by Weiss (2001).

Data Analysis

A Spatial Autoregression (SAR) model was used to address our questions (Wimberley et al. 2007, Beale et al. 2010). Our response variable was burn severity of East Zone, Cascade and Tripod Fires, represented by continuous RdNBR values. Candidate predictor variables included: weather variables assigned by fire progression interval, burn severity classification of past wildfire events (e.g., unburned, low, moderate, and high), time since fire, topographic variables, vegetation types and fuel characteristics (Table 4.1). We examined correlation between possible predictor variables visually and with a linear regression and then excluded correlated variables from the same model. The SAR model to predict RdNBR based on the above variables was published by Wimberly et al. (2009) and Prichard and Kennedy (2014) and was constructed in the R programming language (R Development Core Team 2011). Individual variable models were compared using Akaike's Information Criterion (AIC; Akaike 1974), and final multivariate models were selected based on lowest AIC values. We tried multiple models but removed additional variables when the AIC value was not reduced by more than 50 with the addition of any given variable.

Computational limits required subsampling by Wimberley et al. (2009), however both Beale et al. (2010) and Prichard and Kennedy (2014) used the full data set for this spatially explicit data set. Prichard and Kennedy (2014) demonstrated that the nearest neighborhood distance (30 m) minimized both AIC and Moran's I, as both increased with increasing neighborhood distance. Thus, similar to other SAR applications, we did not subsample and fit all models using the nearest neighbor distance to define the SAR neighborhood weighted matrix for the Tripod Complex, assigning point data information to each 30-m pixel (Kissling and Carl 2008). In the Cascade and East Zone Complex a full data set was impossible due to a failure of the Landsat 7 EMT+ scan line correction mechanism (known as SLC off condition; Howard and Lacasse 2004). This malfunction resulted in missing areas of data within the perimeter of each fire (Figure 4.4). In these two wildfires we used the full available datum, skipping the 150-m scan line areas and treating pixels surrounding the scan lines as true neighbors. To address the possibility that this missing data skewed the results of our SAR analysis, we performed a test of bias by examining the distribution of cover type and topographic variables within these scan lines versus areas with RdNBR data. To rule out bias due to scan line errors our examination on points within and outside the scan lines showed that the distribution of canopy cover, elevation, slope, solar radiation and topographic wetness index were nearly identical for both the Cascade and East Zone Complex fires (see Figure 4.5 for example).

RESULTS

SAR models show that past burn severity, topography, and weather are important predictors

For burn severity, significant predictors in the final models, based on lowest AIC values, varied between study areas. All three study fires had distance to edge, past burn severity, valley bottom, maximum temperature on the day of burning, and cover type as predictors in the final model (Table 4.2 and 4.3). In addition to these five common variables, the Tripod best model included canopy cover, elevation, and slope. The East Zone best model included elevation, TWI, and maximum wind gusts on day of burning. The Cascade best model also included slope, time since fire, maximum wind gusts on day of burning, and canopy cover. All other predictor variables were significant predictors of RdNBR, in part due to the large sample size, but were not included in the final selected models, based on lowest AIC values.

Previous wildfires reduced burn severity of subsequent wildfires

Past fire burn severity had a negative effect on subsequent burn severity on all three large fires we studied. Thus compared to unburned areas, previously burned pixels had reduced burn severity (Figure 4.6a). On the Tripod and East Zone fire areas that burned at high severity in the first fire demonstrated the largest reduction in burn severity in the second fire. On the Tripod the next largest reduction in burn severity was seen on areas previously burned with moderate burn severity, followed by low burn severity. However, on the East Zone, the reduction in reburn severity compared to previously unburned areas was the same on previous low and moderate burn severity. On the Cascade, low burn severity areas had the largest reduction in reburn severity, followed by moderate and then high. Thus, unlike the East Zone and Tripod, areas that burned initially at high burn severity had a higher likelihood of burning again at high severity compared to other previously burned areas.

The other two wildfire metrics we tested were distance to edge and time since fire. Distance to edge was a significant predictor and had a positive effect on burn severity, reflecting that interior regions of these large fires had higher burn severity than the perimeters. This applied to all three fires we studied. Time since past fire had mixed effects in the various models. We excluded this variable in the final model for the Tripod and East Zone due to only small decreases in the best model AIC values, but included it in the final model for the Cascade.

Fire weather, vegetation, and topography influenced burn severity

Of the weather variables we analyzed, the most important predictors of RdNBR were maximum temperature and minimum RH on the Tripod, and maximum temperature and maximum wind speed on the East Zone and on the Cascade. Because temperature and relative humidity are highly and inversely correlated, only maximum temperature, the stronger of the two predictors based on lower AIC values, was included in the Tripod study area final model. Both maximum temperature and maximum wind speed were included in the final model for the East Zone and Cascade.

Of all the LANDFIRE variables, vegetation canopy cover and cover type were the most important predictors of burn severity (Figure 4.6b). Forest canopy bulk density was also a significant predictor. However, because of the high correlation between canopy cover and canopy bulk density, only canopy cover was used in the final model. Canopy height and existing vegetation height were significant predictors, but we excluded these variables from the final model due to the presence of some non-forested cover types in all study areas. Specifically, the grass and shrub cover types had low canopy height measurements, which skewed the relationship between canopy height and burn severity.

Valley bottom, ridge top, and TPI metrics were all significant predictors. Valley bottom and ridge top were inversely correlated thus only valley bottom was included in final model for all three study areas, as this was a better predictor based on the lowest AIC values. Valley bottom had a negative relationship to burn severity, thus the closer to a valley or valley like setting the lower the burn severity. TPI was highly correlated with both of these metrics, and was excluded in the final model.

Elevation was a significant predictor of burn severity on the Tripod and East Zone fires. Burn severity was positively correlated with elevation on all three fires, with increasing burn severity at higher elevations up to 2150 m on the Tripod, 2450 m on the Cascade and 2550 m on the East Zone. After these elevations, burn severity decreased at the highest elevations of each fire (Figure 4.6c).

As slope and TWI were highly correlated, we only included one of these predictors in each model (Figure 4.6d). For the Tripod and Cascade, slope was a slightly better predictor and was used in the final model as this had a lower AIC value in the individual model and in the multifactor model. However, for the East Zone TWI was a stronger predictor and was used in the final model.

DISCUSSION

Both 2006 in the northern Cascades and 2007 in the Northern Rockies were years of widespread fire, also referred to as regional fire years (Cansler 2011, Hudak et al. 2011). Regional fire years are often characterized by higher spring and summer temperatures and drier than average summers (Gedalof et al. 2005, Morgan et al. 2008, Littell et al. 2009). In some previous studies, top-down weather factors were shown to be more dominant drivers of fire size, and override bottom-up factors such as local fuel loading (Bessie and Johnson 1995, Gedalof et al. 2005). However, in this study we see that even though fire weather conditions were extreme and weather factors influenced burn severity, bottom-up factors also influenced burn severity as found by Dillon et al. (2011) and Birch et al. (2015). Cover type, valley bottom, and TWI or slope were significant predictors of burn severity across all study areas. Additionally, several other topographic variables, such as canopy cover and elevation, were strong predictors in one or more fires.

Previous fires reduced severity of subsequent fires

In all three large wildfires we studied, burn severity and therefore ecological effects were moderated by previous burn severity (Figure 4.6a). Surface fuels and tree density, critical to how fires burn, were likely reduced on these previously burned areas (Stevens-Rumann and Morgan *in review*). This reduction in fuels is important for reducing subsequent crown scorch and fuel connectivity (Alexander and Cruz 2012) and explains the degree to which others have found that previous fires may serve as fuel breaks, limiting extent and reducing burn severity of subsequent fires (Peterson et al. 2005, Boer et al. 2009, Teske et al. 2012, Parks et al. 2015). On the Tripod, we found that areas previously burned at high severity exhibited the largest drop in subsequent burn severity. This result is in contrast to previous studies that found only low to moderate previous burn severity resulted in a reduction in subsequent burn severity (Collins et al. 2009, Holden et al. 2010, Parks et al. 2014a). These differences may be a product of rapid vegetation recovery in the Sierra

Nevada, where large shrubs grow rapidly following high burn severity fires (Collins et al. 2009, van Wagtendonk et al. 2012).

Our contrasting results may also be because we studied areas outside of wilderness, and thus in differing forest types and experienced different fire suppression actions than previously studied areas that were all within wilderness areas. Many of the wilderness areas in this region are dominated by higher-elevation, colder mixed-conifer forests (Haire et al. 2013, Parks et al. 2014a). These areas had some of the highest burn severities on our study areas (e.g. subalpine fir and lodgepole pine forests; Figure 4.6b), thus we would expect differences based on cover types even if these studies were in the same region (Parks et al. 2014a). Further, some of the observed difference may be a result of the interaction of suppression tactics or past management in these previously burned areas. Fire suppression methods and/or previous land management on the edge of these large wildfires may have interacted with past fire burn severity to decrease subsequent burn severity, regardless of initial burn severity.

Time since fire was not an important predictor in two of the three study areas, and on the Cascade it had the opposite effect than we hypothesized. We expected burn severity to be positively correlated with time since previous fire. However, we found the opposite: the longer the time since fire the lower the burn severity, which contrasted with previously published research (Holden et al. 2010; Parks et al. 2014a). This is likely a product of two interacting variables. First, many of the more recent wildfires, especially those that burned in 2006, were relatively small. As in fuel treatments (Allen et al. 2002, Graham et al. 2003), smaller past wildfire size may not be as effective in reducing the burn severity in subsequent large wildfires, especially during extreme fire weather as quickly moving fires may easily overrun proportionally small areas of fuels reduction. Second, cover type and topographic variables may vary on the different past wildfires, thus influencing burn severity more than a simple time-since fire metric indicates. For example, cover type in the more recent fires were predominantly higher elevation lodgepole pine and subalpine forests, thus the reburn severity was higher than older fires reburning in lower elevation Douglas-fir and ponderosa pine forests, similar to Parks et al. (2014a). Additional study areas should be added to assess if the observed differences between our study areas and those previously published were anomalies of these landscapes, if additional variables should be considered and to determine if certain relationships are dominant across multiple areas.

There is some concern about the validity of RdNBR values in assessing burn severity (Cansler and McKenzie 2012, Parks et al. 2014b), especially in repeatedly burned areas (Holden et al. 2010). Holden et al. (2010) attributed potential errors in the repeated categorization of high burn severity from satellite imagery to the relative change in understory vegetation and tree seedling mortality. After a high burn severity fire, satellite imagery from a subsequent fire may be detecting changes in understory vegetation even if the second fire burned at low intensity and the magnitude of ecological change was not as substantial as the first high severity fire. Rather than detecting similar forest structural and ecological changes that were associated with the first high burn severity fire, similar infrared changes are detecting smaller ecological changes (Holden et al. 2010). We attempted to address some of this concern by also examining the influence of cover type and canopy cover at the nearest time period available prior to the 2006 and 2007 wildfires. Some field validation plots were established in prescribed burn areas that reburned in the Tripod Complex, but most were in low burn severity areas as a result of the treatment effect (Cansler and McKenzie 2012, Prichard and Kennedy 2014). On these sites, producers accuracy was around 40%, however much of the misclassification occurred when RdNBR values were close to the burn severity cut-off established by Miller and Thode (2007) but field validation did not differ from that inferred from satellite imagery by more than one category (e.g. low severity classification when field validation was moderate severity). Additional field validation, a sensitivity analysis, and increased CBI plot sampling immediately following a reburn, especially high severity reburns, is needed to fully understand the possible vegetation shifts and responses to these repeated disturbances (van Wagtendonk et al. 2012, Parks et al. 2014a).

More extreme fire weather resulted in higher burn severity

The predictive ability of our assigned weather variables, though broadly summarized from nearby weather stations, indicates that these nearby weather stations may be a decent proxy for smaller-scale, fire-weather relationships (Collins et al. 2007, Prichard and Kennedy 2014, Birch et al. 2015), however there are numerous limitations and assumptions made by using weather variables at this scale. Lower fuel moisture content indicated here with both minimum RH and maximum temperature indicate a possible increase in flaming combustion, which influences burn severity (Ferguson et al. 2002). Maximum wind gusts were also significant predictors, although their relationship to burn severity was mixed on the different study areas. On the East Zone Complex higher burn severity was correlated with higher maximum wind speeds, but a negative correlation was observed with burn severity on the Cascade Complex.

These inconsistent relationships may be explained by the accuracy of our assigned weather variables. First, the location and thus the accuracy of these nearby weather stations in summarizing weather across these large wildfires is unknown. The RAWS station used for the East Zone and Cascade was closer to the East Zone and thus may better represent that study fire. Second, fine-scale variability in weather patterns, especially in wind, may have played a more prominent or different role than we were able to detect using our data (Taylor et al. 2004, Hiers et al. 2009). Third, the use of progression maps to assign weather data is somewhat unique and allows us to examine fine-scale patterns of weather variability, but is not without limitations (Prichard and Kennedy 2014, Birch et al. 2015). The progression maps helped us to relate burn severity at a point to the weather at the general time of burning, as done by Birch et al. (2015). However, progression maps are imperfect and we don't know when and for how long within the progression period that a given point burned, nor how well the conditions at the RAWS station represent conditions at each point where we analyzed burn severity. The progression interval varied with some progression maps representing <24 hours and others representing up to four days of burning. Further, our approach does not take into account the degree to which the influence of weather is non-linear as suggested by Birch et al. (2015). Nonetheless, multiple weather variables were significant predictors of burn severity. We recognize that data from multiple weather stations that show finer-scale weather variability for the time when each point burned would greatly inform our understanding of environmental influences on burn severity.

Our weather variables do not reflect climatic conditions prior to or after the fire. Other predictor climatic variables such as energy release component or Palmer Drought Severity Index, both of which reflect cumulative effects of prior weather, may have influenced burning conditions (Abatzoglou and Kolden 2013). Although all three large fires burned in years of widespread fires, pre-fire conditions may have varied between the Tripod that burned in 2006 and both of the East Zone and Cascade fires that burned in 2007. Additionally the weather immediately following these fires varied and could have influenced fire effects as inferred using RdNBR.

Dense colder forests burned more severely than open warmer forests

Vegetation cover and type are both significant predictors of burn severity. Burn severity was higher with higher canopy cover values, meaning denser, closed-canopy forests burned at higher severity than open canopy forests, as expected (Schoennagel et al. 2004). Burn severity was highest in the “subalpine” and “lodgepole pine” forest types (Figure 4.6b). Thus colder, high elevation forests are more likely to burn with a higher proportion of high severity, stand-replacing fires, as reported by others (Bigler et al. 2005, Collins et al. 2007, Prichard and Kennedy et al. 2014). Our canopy cover to burn severity relationships, support the paradigm that multi-story, dense forests tend to burn more severely than open canopy, low density forests (Bigler et al. 2005, Lentile et al. 2006).

Burn severity in grasslands and shrublands was variable across study areas and our models generally were not significant predictors of burn severity here. These areas comprised a relatively small portion of the total landscape, with 8% on the Tripod, 15% on the East Zone, and 18% on the Cascade across all four categories (deciduous shrub, shrub, grass and subalpine grass or avalanche area). The lack of a relationship between burn severity and key predictor variables in these cover types may be due to the difficulty in inferring burn severity from satellite imagery in certain cover types (Hudak et al. 2007) or the variation among and within these cover types.

Topographic position, slope, and elevation were all strong predictors of burn severity

TWI and slope were important predictors of burn severity in each model. The positive correlation between burn severity and slope indicates that as slope steepness increases so does burn severity, similar to the findings of others (Chandler et al. 1991, Lentile et al. 2006, Collins et al. 2007). However, burn severity was only positively correlated with slope up to approximately 18% slope on all three fires, then burn severity declines at steeper slopes (Figure 4.6d). Burn severity decreased as TWI increased, similar to what others have found using this metric (Holden et al. 2009, Dillon et al. 2011). These relationships are generally associated with fire behavior. As wildfires burn up steep, drier slopes fire intensity have been shown it increase, transition from surface to crown fire is easier, rate of spread increases (Alexander 1985, Scott and Reinhardt 2001, Finney 2004).

The positive correlation between elevation and burn severity is likely a result of differences in cover type at these various elevations. Low elevation areas of all three fires were dominated by relatively fire-resistant species such as Douglas-fir and ponderosa pine. Conversely, mid- to high elevation areas were dominated by higher-density mixed-conifer forests. Such forests include many thin-barked species that historically burned at higher burn severity with longer fire return intervals, such as lodgepole pine and subalpine fir (Agee 1993). Across many regions of the western US, within forested ecosystems as elevation increases so does fire return intervals and proportion of high burn severity when fires do occur (Schoennagel et al 2004). The highest elevations had burn severities similar to the lowest elevations, likely due to both the reduction in burnable vegetation at or above tree line and the decrease in extreme fire weather conditions with lower temperatures and higher relative humidities, as seen in other locations (Fischer and Bradley 1987). In the Tripod Complex, subalpine meadows and other wet meadows generally did not burn or burned at low severity, supporting this hypothesis (personal observation, Susan Prichard).

The category “valley bottom” was also a strong negatively correlated predictor of burn severity. Burn severity was higher on areas that were on hillsides and uplands, whereas areas categorized as “valleys” or “valley-like” had lower burn severity. This may have been a result of drier fuels, represented here with solar radiation and TWI, and the interaction of vegetation, slope and wind speeds increasing burn severity in “non-valley like” areas. The strong relationship observed here between burn severity and topographic positioning highlights the importance of bottom-up factors as found by Dillon et al. (2011) and Birch et al. (2015).

Repeated fires demonstrate self-regulation even when large fires burn under extreme conditions

Self-regulation of repeated high-intensity or high-severity fires over a short time period may be possible due to reductions in fuels and shifts in stand structure (Peterson 2002, Parks et al. 2014). Patches of stand replacing fires or areas maintained by frequent surface fires create natural fuel breaks that may reduce subsequent fire spread or burn severity (McKenzie et al. 2011). The decrease in burn severity across all our previously burned areas supports this concept. We did not see evidence of positive feedback (such as the grass-fire cycle (Balch et al. 2012) with subsequent fires. Though other variables were also strongly predictive, large decreases in burn severity, especially in areas previously burned at high burn severity indicates that these altered landscapes are less likely to burn severely again within a short time period. However, the amount of area reburned in these landscapes is small (roughly 3% of our total study fires) and this could change with more extensive fires and changing environmental conditions. As previous wildfires altered burn severity under the extreme burning conditions experienced in all three study areas, we expect these past wildfires to be even more effective in the event of fires burning under less extreme conditions (Pollet and Omi 2002).

The current ability of forests within the western US to self-regulate is somewhat unknown given the long fire-free period in the past century (Hessburg et al. 2007). However, as demonstrated here and by others, previous wildfires can alter burn severity and even fire spread (Teske et al. 2012, Haire et al. 2013, Parks et al. 2014a, 2015), thus a single reintroduction of fire may be sufficient to reinitiate self-regulation. The longevity and effectiveness of this self-regulation is highly dependent of vegetation type and vegetation recovery, as seen by others (Peterson 2002, Holden et al. 2007, van Wagtendonk et al. 2012). Therefore additional analysis on these study areas contrasting reburn severity within vegetation types and other topographic variables may foster a better understanding of where and for how long previous fire burn severity impacts subsequent wildfires. While we observed decreases in reburn severity across all our study areas, this additional analysis will further demonstrate the conditions under which these interactions are compounding to reduce burn severity and where topographic or vegetation variables supersede any changes from past fires. Another aspect that should be explored on our study areas is when past wildfires were effectively stopped fire spread and when topographic and weather variables overrode the potential fuel breaks. This will further our understanding of self-regulation in terms of inhibiting the occurrence of repeated disturbances, not just reducing the severity of these disturbances. This will also aid managers to better understand of when to use past fire perimeters in their fire management decisions.

CONCLUSIONS

Past wildfires, topography, vegetation, and weather interacted to influence burn severity on subsequent large wildfires. Past wildfires reduced the burn severity of subsequent fires, especially in areas previously burned at high severity. Both topography and weather on the day of burning strongly influenced burn severity. Differences in predictor variables between study areas were likely a result of varying cover types and the location and size of past wildfires.

Our study supports others that found past wildfires influenced subsequent wildfire burn severity (e.g. van Wagtendonk et al. 2012, Parks et al. 2014a). While many factors influence burn severity, previous wildfires did reduce burn severity of all three subsequent large fires. As such, managers have the potential to use these past burn perimeters in fire management for up to 23 years following a previous fire. Similar to studies on fuel treatment effectiveness, forest types that historically had frequent fire return intervals may be better suited for reducing reburn severity (Fernandes and Botelho 2003, Peterson et al. 2005, Fulé et al. 2012) than lodgepole pine or subalpine forest types that historically burned infrequently at higher burn severity (Reinhardt et al. 2008). Additionally, past wildfires may be more effective at reducing subsequent burn severity when wildfires are burning under less extreme fire weather when fuels are the dominant limiting factors in fire spread (Littell et al. 2009, Parks et al. 2015). Even under extreme fire weather, vegetation, topography, and past fire burn severity influenced burn severity on large fires. This supports previous researchers, who predict burn severity will be less sensitive to future climate change than fire extent (Birch et al. 2015), which encourages the further consideration of allowing wildfires to burn into previously burned landscapes.

In the future, previously burned areas should be considered more readily in both fire suppression and in achieving land management goals. Given the rising cost of fire suppression (Houtman et al. 2013), knowing when and how to use past wildfires as effective barriers or reducers of burn severity could help to reduce these costs of future large wildfires. Additionally, altering management perspectives to consider the use of these large burned landscapes in achieving broader land management goals similar to fuels reduction treatments by reducing tree density and surface fuels, can further our compliance with the federal fire management policy (USDA and USDI 2009). Wildfires, even the large fire events studied here, possess some attributes of self-regulation. Limited suppression management will not only serve managers by potentially mitigating of future burn severity, but increases the ecological function of these landscapes as self-regulating entities that can withstand the impacts of repeated disturbances that will become ever more present with climate change.

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TABLES

Table 4.1 Predictor variables considered for spatial autoregression (SAR) modeling for all three study areas (Tripod, Cascade and East Zone).

Type of variable	Variable	
Wildfire data	Past burn severity	Categorical RdNBR (unburned, low, moderate, high)
	Distance to Edge (m)	Distance from study fire perimeter
	Time since previous fire	Number of years since each pixel burned, if not previously burned since 1984 (the record of MTBS data) we assigned each point "100 years"
Fire weather	Maximum temperature(°C)	Maximum temperature over each fire progression interval
	Average temperature(°C)	Average temperature over each fire progression interval
	Maximum wind speed(kph)	Maximum recorded wind gust over each fire progression interval
	Average wind speed(kph)	Average wind speed over each fire progression interval
	MinRH (%)	Minimum relative humidity each fire progression interval
Vegetation	Canopy height (m)	Average height of the top of the vegetated canopy (LANDFIRE)
	Canopy bulk density (kg m^{-3})	Density of available canopy fuel in a stand (LANDFIRE)
	Cover Type	Derived from existing vegetation type (LANDFIRE)
	Canopy Cover (%)	Percent canopy cover of vegetation (LANDFIRE)
	Existing veg height (m)	Average height of the dominant vegetation (LANDFIRE)
Topography	Elevation (m)	Elevation model from the National Elevation Dataset
	Hill Shade	The hillshade of the digital elevation model
	Slope(degrees)	Slope gradient
	Solar radiation (WH m^{-2})	The potential incoming solar radiation (no cloud cover) summarized over a one year preiod. Accounts for aspect, slope angle, and terrain shading for the latitude of the study area and is derived from the elevation model.
	Topographic wetness (TWI)	Steady state Topographic Wetness Index
	Topographic position index (TPI)	A discrete classified TPI raster using the class breaks as described in the 'Slope Position' portion of Weiss (2001).
	Valley	Fuzzy Valley Bottom or 'Valley-like' settings: A raster representing a fuzzy ramp of valley-like settings at scale of the neighborhood (see "dist" radius), where 0 = 'No support for the proposition "is a valley", and 100 is 'Full support for the proposition "is a valley".
	Ridgetop	Fuzzy Ridgetop or 'Ridge-like' settings: A raster representing a fuzzy ramp of ridge-like settings at scale of the neighborhood (see "dist" radius), where 0 = 'No support for the proposition "is a ridge", and 100 is 'Full support for the proposition "is a ridge".

Table 4.2. Regression models of relative differenced Normalized Burn Ratio (RdNBR) for the Tripod, Cascade and East Zone study areas. N is the number of points analyzed.

Model	Predictor variables	N	R^2	AIC
Tripod	edge + canopycover + pastSeverity + covertype + maxtemp +elevation + valley + slope	326,541	0.92	4,211,617
East Zone	elevation + valley + maxtemp + twi + maxgust + edge + covertype + pastSeverity	905,805	0.73	12,705,587
Cascade	slope + edge + valley + maxtemp + cc + maxgust + covertype + timesincefire + pastSeverity + canopycover	975,414	0.77	13,728,154

Table 4.3. Outputs for final SAR model for each variable.

	Tripod			East Zone			Cascade		
Variables	Estimate	SE	*P*	Estimate	SE	*P*	Estimate	SE	*P*
Intercept	-1.78E+02	3.54E+01	<0.0001	-71.4345	7.384062	<0.0001	7.66E+02	3.19E+01	<0.0001
Distance to edge	8.66E-02	4.49E-03	<0.0001	0.004026	0.000695	<0.0001	3.67E-02	1.34E-03	<0.0001
Valley bottom	-1.76E-01	2.46E-02	<0.0001	-0.51774	0.020128	<0.0001	-6.68E-01	2.49E-02	<0.0001
Maxtemp	1.11E+00	3.76E-01	0.003229	3.424787	0.130957	<0.0001	5.96E+00	2.27E-01	<0.0001
Past severity – High	-8.61E+01	5.84E+00	<0.0001	-25.6887	2.402644	<0.0001	-2.55E+02	2.76E+01	<0.0001
Past severity – Low	-1.64E+01	2.90E+00	<0.0001	-16.8467	1.29375	<0.0001	-2.94E+02	2.75E+01	<0.0001
Past severity– Moderate	-4.35E+01	4.38E+00	<0.0001	-16.9921	1.755713	<0.0001	-2.76E+02	2.75E+01	<0.0001
TWI *or* slope	1.71E+00	1.57E-01	<0.0001	-5.12705	0.183951	<0.0001	-4.91E-01	1.10E-01	<0.0001
covertypeDFHE	5.35E+00	9.23E+00	0.562113	7.476463	1.446297	<0.0001	-3.38E-02	2.70E+00	0.990028
covertypeDSHRUB	9.28E+00	2.79E+00	0.000879	6.392009	3.973786	0.1077	7.72E+00	3.16E+00	0.014372
covertypeGRASS	-4.23E+00	2.12E+00	0.045925	2.656381	2.37053	0.2625	1.32E+01	3.52E+00	0.000165
covertypeLP	2.48E+00	1.05E+00	0.018281	8.480408	1.855558	<0.0001	8.32E+00	2.85E+00	0.003502
covertypeNV	7.63E+00	4.27E+00	0.073813	-27.7242	4.846212	<0.0001	-1.19E+01	5.73E+00	0.03721
covertypePP	-9.32E+00	3.72E+00	0.012233	2.409217	2.079962	0.2467	-2.09E+00	4.40E+00	0.6352
covertypeRIP	-5.65E+01	3.66E+00	<0.0001	-1.13451	2.851916	0.6908	-9.09E+00	3.54E+00	0.010137
covertypeSHRUB	-6.05E+00	3.17E+00	0.055898	22.23183	2.064449	<0.0001	5.04E+00	3.12E+00	0.106447
covertypeSUBALP	3.27E+00	1.05E+00	0.001783	11.0589	1.661906	<0.0001	1.04E+01	2.80E+00	0.000204
covertypeSUBGRASS/AV	1.10E+01	2.42E+00	<0.0001	15.31953	2.022118	<0.0001	1.13E+01	3.20E+00	0.000441
Elevation	3.97E-01	1.83E-02	<0.0001	0.309098	0.003711	<0.0001	-	-	-
Canopy cover	7.71E-01	3.17E-02	<0.0001	-	-	-	6.38E-01	2.74E-02	<0.0001
Maxgust	-	-	-	1.251197	0.234168	<0.0001	-4.10E+00	1.99E-01	<0.0001
YearsSinceFire	-	-	-	-	-	-	-3.41E+00	3.07E-01	<0.0001

FIGURES

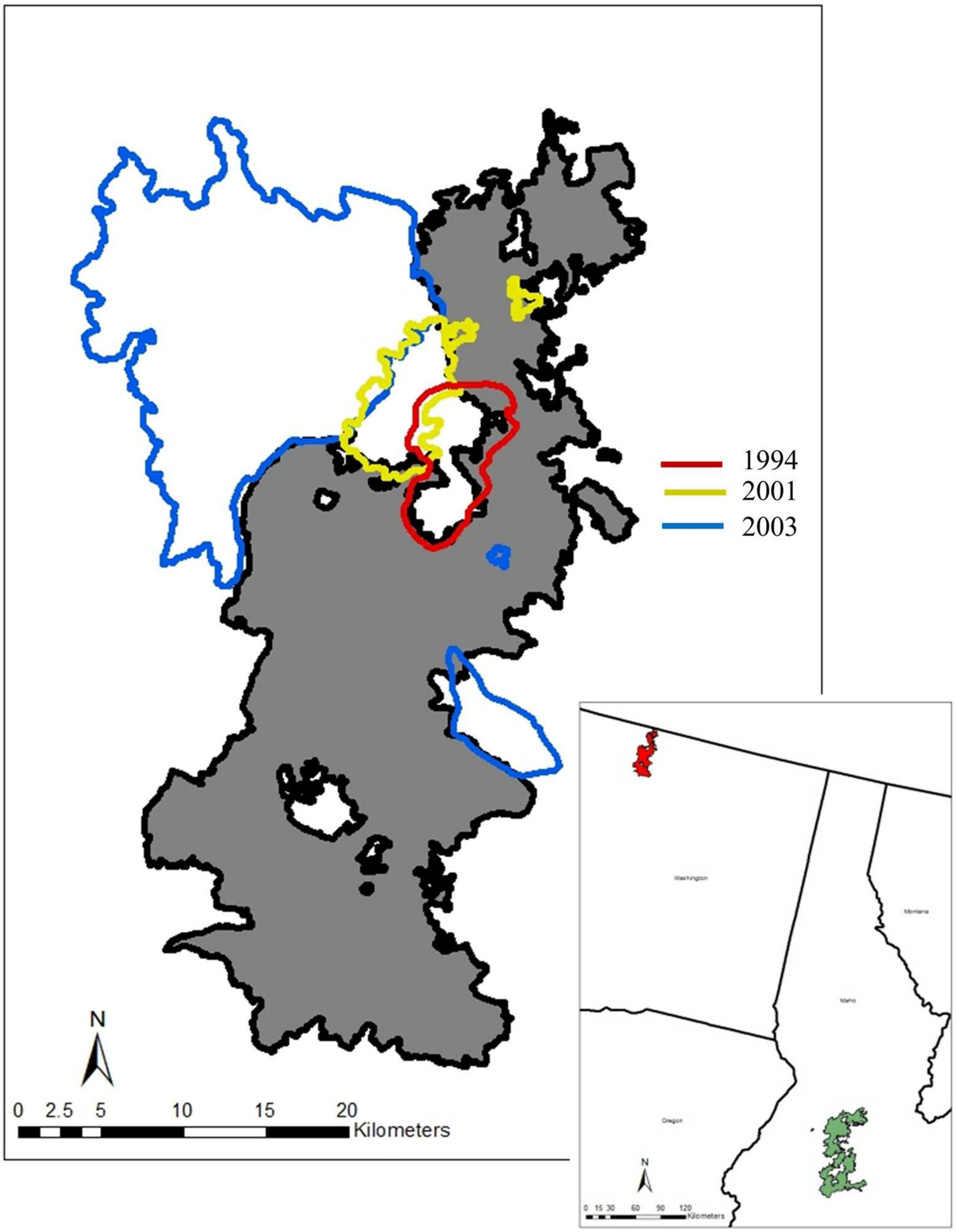

Figure 4.1. Tripod Complex (gray) with perimeters of previous wildfire. Older past fires are indicated with warmer colors (red), while more recent fires are indicated in cooler colors (blue). Inset: location relative to Washington, Oregon, Idaho and Montana, with the red area indicating the Tripod Complex.

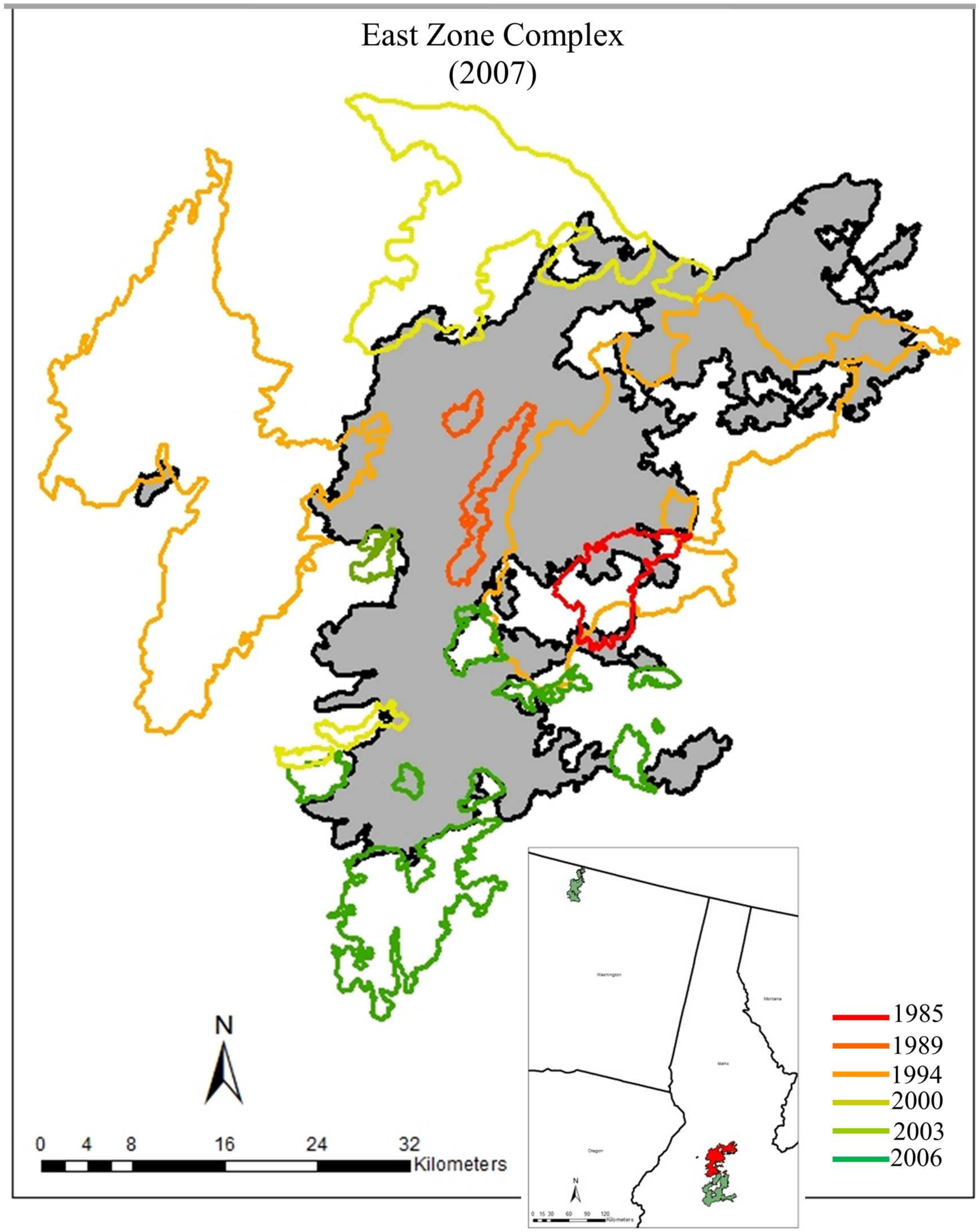

Figure 4.2. East Zone Complex Fire (gray) with perimeters of previous wildfires. Older past fires are indicated with warmer colors (red and orange), while more recent fires are indicated in cooler colors (green and blue). Inset: location relative to Washington, Oregon, Idaho and Montana, with the red area indicating the East Zone Complex.

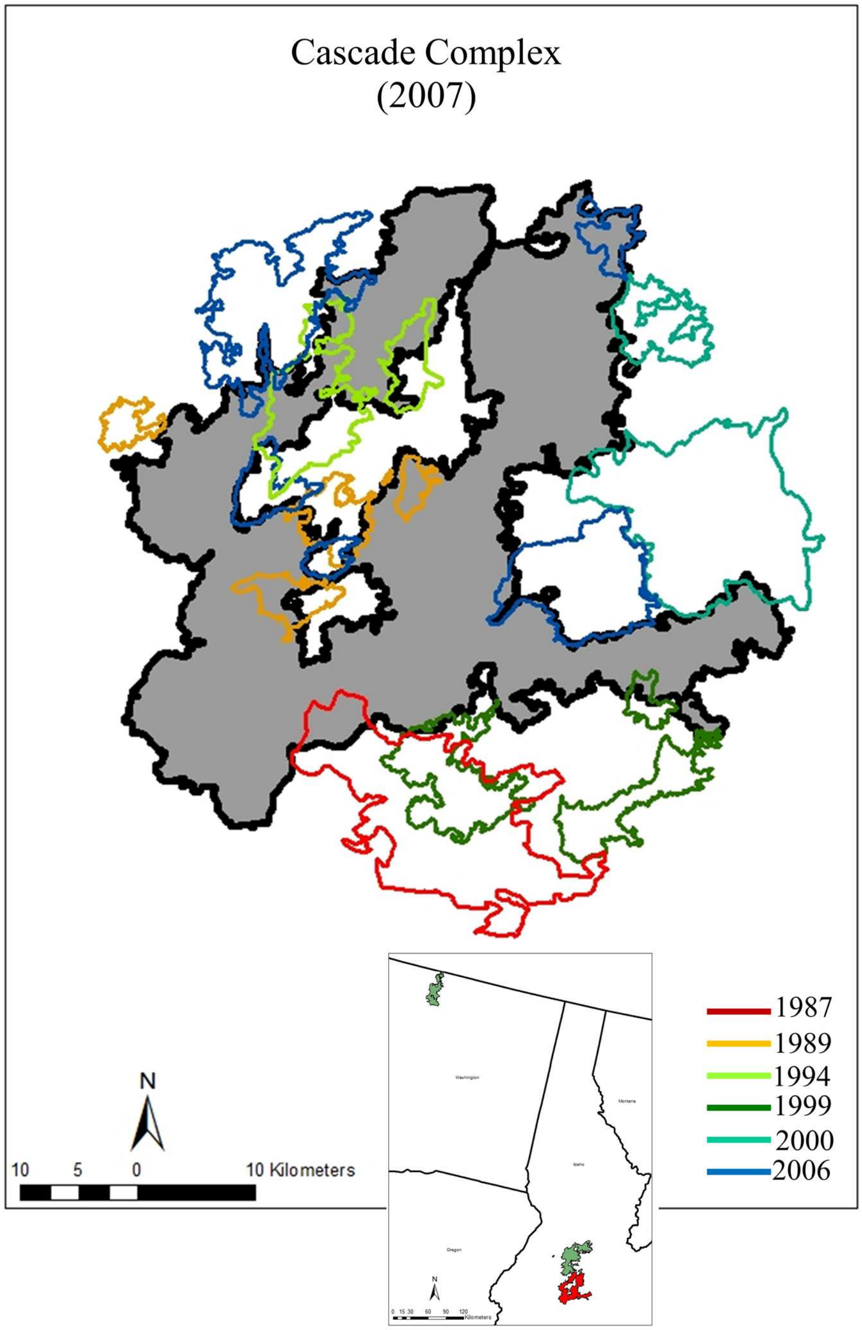

Figure 4.3. Cascade Complex Fire (gray) with perimeters of previous wildfires. Older past fires are indicated with warmer colors (red and orange), while more recent fires are indicated in cooler colors (green and blue). Inset: location relative to Washington, Oregon, Idaho and Montana, with the red area indicating the Cascade Complex.

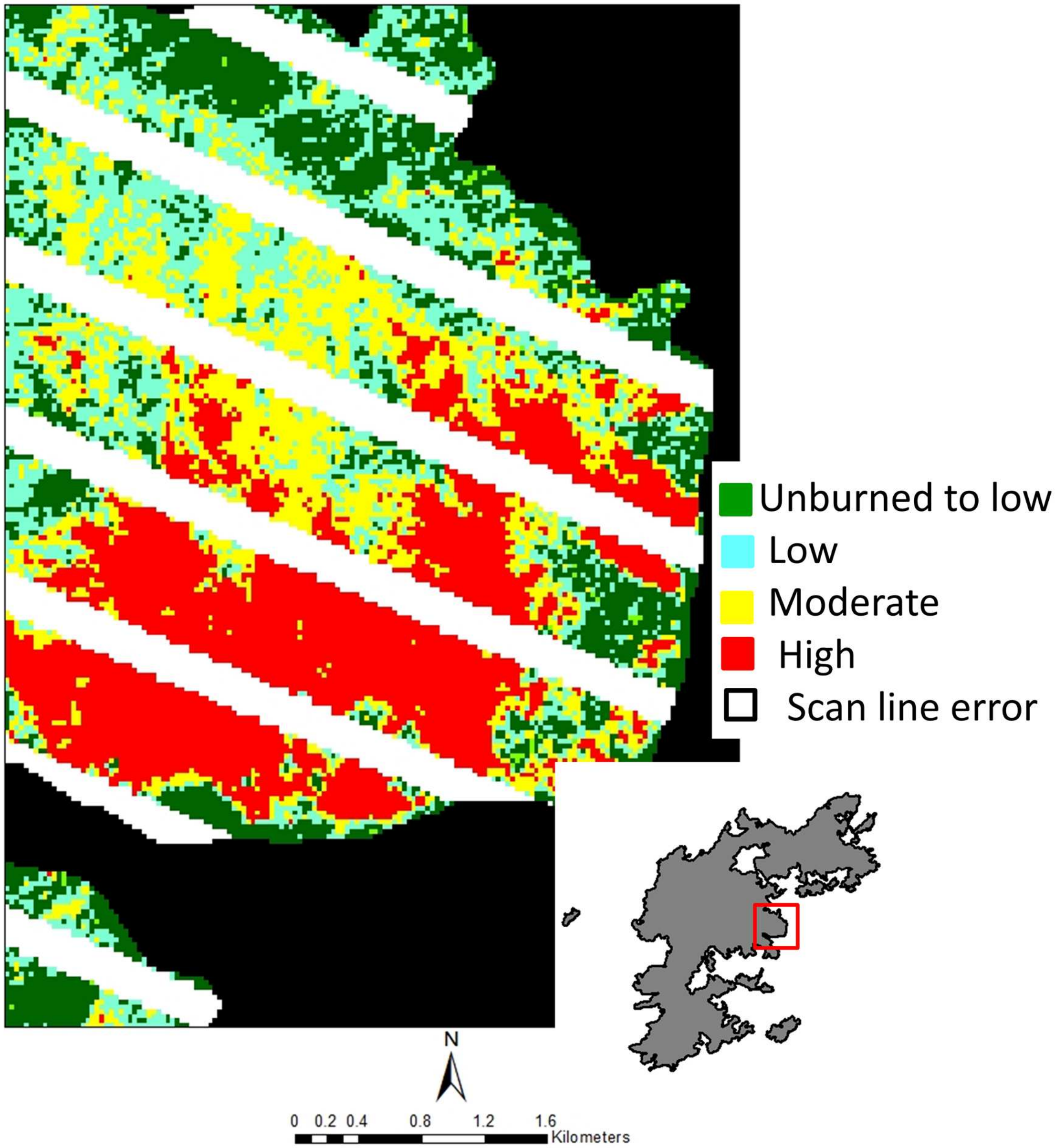

Figure 4.4. Example of scan line errors in the Landsat satellite data on the East Zone Complex Fire. White lines indicate missing data; lines are 150 m wide.

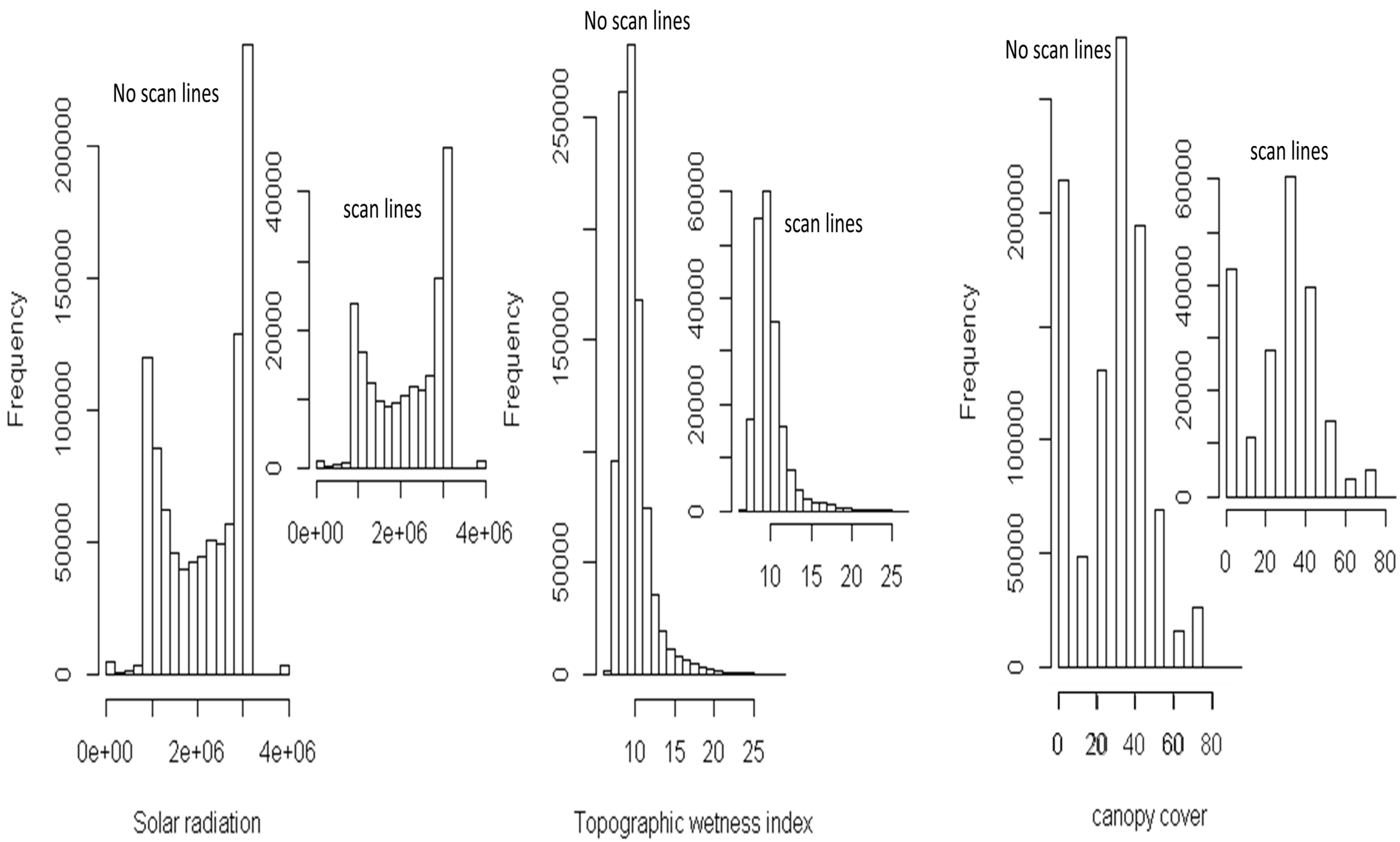

Figure 4.5. Distribution of topographic (solar radiation and topographic wetness index) and vegetation (canopy cover) variables in data set excluding scan lines and in a data set of only scan line error pixels. Distributions are very similar for both, reducing the possibility of bias with the missing data.

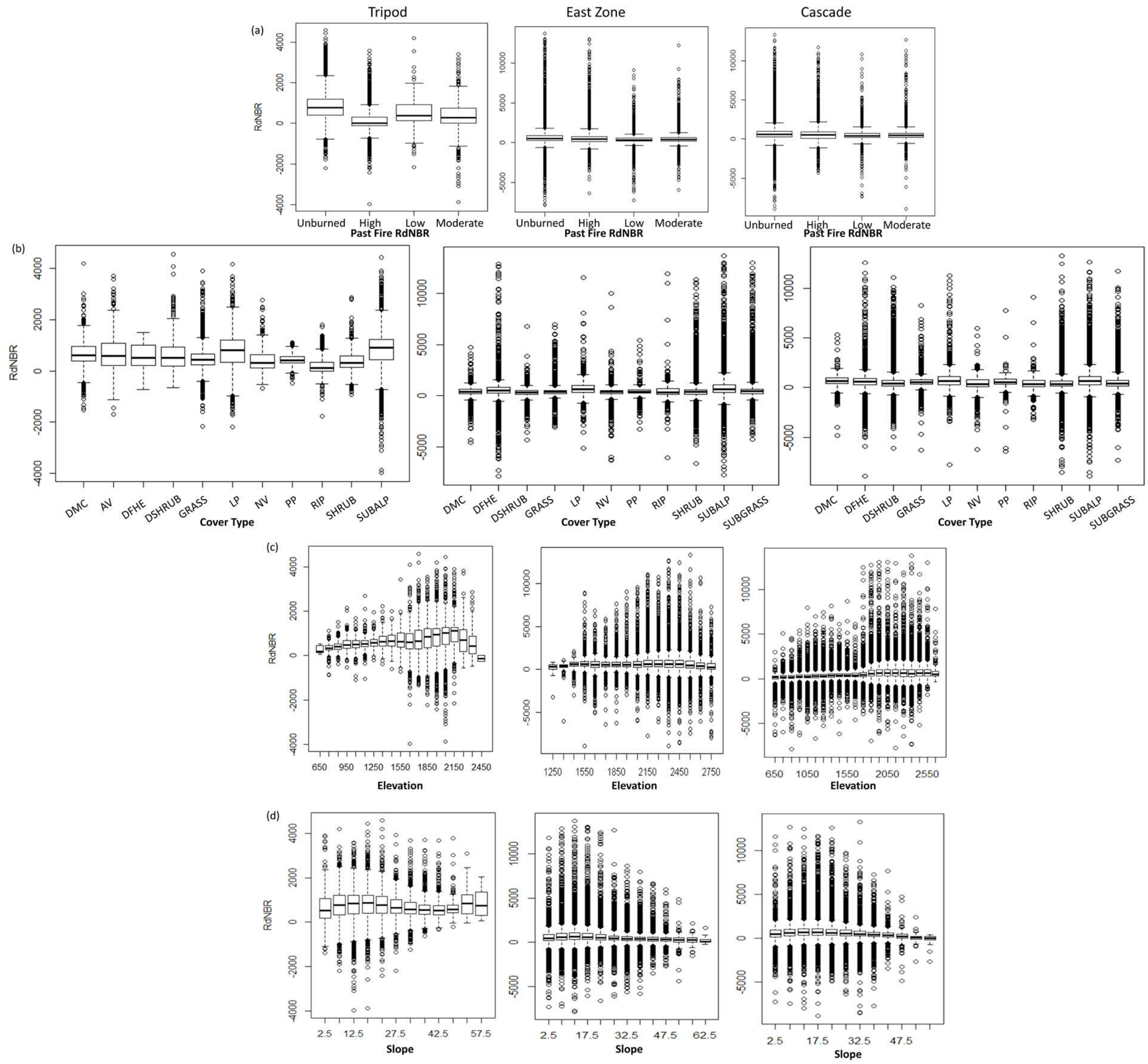

Figure 4.6. Box and whisker plots of RdNBR response by four predictor variables considered in the SAR analysis. The Tripod is on the left, East Zone in the middle, and Cascade on the right. (a) past fire RdNBR, (b) Cover Type, (c)Elevation in meters, (d) Slope in degrees.

CHAPTER 5: REPEATED DISTURBANCES AND RESILIENCE: IMPLICATIONS AND CONCLUSIONS

My dissertation covers the impacts of repeated disturbances, both bark beetles and wildfires, on forest structure and tree regeneration. I used various methods including field and remote sensing to examine questions of forest resilience and resistance following repeated disturbances at both stand and landscape scales. This project is one of the first to look at the ecological effects of repeated disturbances on mixed-conifer forest recovery and has enhanced our understanding of how previous disturbances influence the effects of subsequent disturbances and forest resilience. Here, I highlight the need for additional research to fully understand the nature of these interactions and the broader implications of my findings.

FUTURE RESEARCH NEEDS

First, long-term studies, manipulative experiments, and ecosystem process modeling to examine differences in forest trajectories through time are needed to fully understand the impact of large and repeated interacting disturbances. This project focused on relatively short time intervals (<23 years), and the scientific community lacks the understanding of forest dynamics at the full time-scale of forest recovery and succession (Morgan et al. 2014). Through the use of repeatedly monitored sites we can understand longer-term recovery trajectories in the changing climate. Process modeling would allow us to project possible changes into the future and under different climatic conditions to understand forest dynamics on broad temporal and spatial scales that I will be unable to observe during my lifetime as a scientist. Process models not only allow us to test assumptions we make about short-term and long-term forest recovery trajectories, but they also examine landscape level changes and ecosystem processes that will further our understanding of resilience to both climate change and repeated disturbances. Manipulation experiments that reintroduce fire to previously disturbed sites will also eliminate some of the inherent variability in natural experiments such as mine. These methods will help advance our understanding of forest resiliency to disturbances at different temporal and spatial scales and under different climatic conditions. Climate change is expected to have a large impact on these forests in the coming decades (Littell et al. 2009), thus current patterns of tree regeneration and relationships between interacting disturbances may be very different than we observe under future climate scenarios.

Second, studying the interactions of bark beetles and wildfires across a full range of severities (i.e. low, moderate, and high percent tree mortality) and in different forest types will enable a more comprehensive understanding of the interactions. I focused on area of high tree mortality (high severity) and did not include a full range of conditions, and thus some interactions may have been masked by the severity of the subsequent disturbance (see Chapter 2 for full details). I expect some compounding interaction effects of bark beetle activity followed by wildfire would be visible in areas that burned at lower severity. Additionally, by grouping some burn severities and only studying a single year of subsequent wildfire in chapter three, I may have missed some of the variability of fire effects present across the full range of conditions (see Chapter 3 for full details). I expect that the general patterns I observed in the various burn severities would be similar but other effects such as years between wildfires may show more significant patterns, such as those seen by Parks et al. (2014).

Third, analysis of additional fires as well as further analysis on other aspects of repeated wildfires would improve our understanding of forest resistance and the dominant influencing factors on burn severity that I discussed in Chapter 3. For example, examining the differences in burn severity within a cover type, instead of across all cover types may reveal different patterns than observed across the whole landscape. Similarly, analyzing the topographic and weather variables of importance within a burn severity class (i.e. low, moderate, high) may show some predictors are stronger than others.

SCIENTIFIC AND MANAGEMENT IMPLICATIONS

Wildfires and bark beetle outbreaks are often viewed as negative, especially when they alter forest structure and kill large trees over large areas. These disturbances can reduce potential timber value in the short-term and alter species composition. These disturbances can also alter landscapes that people recreate in and enjoy, especially when and where people highly value landscapes with little evidence of recent disturbance. However, disturbances, even repeated disturbances, do not need to be viewed negatively from an ecological perspective. These repeatedly disturbed landscapes can enhance wildlife habitat (Hutto 2006), increase landscape heterogeneity (Allen et al. 2002, Parks et al. 2015), and reduce future fire hazards (Larsen et al. 2013, Hoffman et al. 2013). I found that repeated disturbances can foster resilience of forest to future fires depending on the severity and frequency of these disturbances.

The forests of the Interior Northwest are resilient ecosystems that are adapted to multiple disturbances. Even after a century of fire suppression and other human-forest manipulations, tree regeneration, forest structure, and woody fuels demonstrate resilience following large, repeated disturbances. Managers can use these disturbances to meet management objectives. Repeatedly burned areas meet some vegetation management objectives by reducing fuel loads and increasing the density of large snags, and managers can use them to reduce burn severity and act as fuel breaks in the event of a future wildfire (Prichard and Kennedy 2014, Parks et al. 2015). However, the severity and timing of the subsequent wildfires does influence fuels reduction and tree seedling regeneration and may not meet vegetation management objectives in all areas. Areas of bark beetle mortality and wildfire do not show signs of a compounding negative impact on forest recovery and tree seedling establishment and growth. Thus, if resilience is the goal, these areas do not require additional management actions beyond those deemed necessary for areas of high burn severity.

CONCLUSIONS

This dissertation is one of the first studies to look at the impact of repeated disturbances on forest resilience using both field and remote sensing methods. My dissertation showed that mixed-conifer forests of the Interior Northwest are generally resilient to repeated disturbances. Short-term forest trajectories following both wildfires and bark beetle outbreaks illustrate that the majority of these forests will likely regenerate into forests. However, areas of repeated high severity disturbances may not be recovering to similar forest types and may be of particular concern given future fire-climate change relationships. Thus long-term (>25 years) forest trajectory studies are needed. My dissertation is not only the first to look at forest resilience following multiple types of disturbances, but it speaks to multiple aspects of forest resilience and resistance of these mixed-conifer landscapes. My dissertation applies ecological theory to collected data and furthers our scientific understanding of these complex relationships, however much additional work is needed to fully understand the nature of interacting disturbances.

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APPENDIX A

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