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Title: Patterns of Crown Damage within a Large Wildfire in the Klamath-Siskiyou Bioregion

Abstract approved:

Thomas A. Spies

The 2002 Biscuit Fire burned through more than 200,000 ha of mixed-conifer/evergreen hardwood forests in southwestern Oregon and northwestern California. The remarkable size of the fire and the diversity of conditions through which it burned provided an opportunity to analyze the correlates of burn severity across vegetation types and disturbance histories and to describe the weather, topographical, and fuel conditions that gave rise to the mosaic of crown damage.

In chapter two, I focused on a region that had burned previously by the 1987 Silver Fire then was subject, in part, to salvage-logging and conifer planting before being reburned by the Biscuit Fire. I used the Landsat-based differenced normalized burn ratio (dNBR) to quantify severity in both fires and took a hypothesis-testing approach to answering two questions: First, was severity in the Biscuit Fire associated with severity in the Silver Fire in unmanaged areas? And second, did areas that were salvaged-logged and planted with conifers after the Silver Fire burn more or less severely in the Biscuit Fire than comparable unmanaged areas? I found that areas that

burned severely in 1987 tended to re-burn at high severity in 2002, after controlling for the influence of several topographical and biophysical covariates. Areas unaffected by the initial fire tended to burn at the lowest severities in 2002. In addition, areas that were salvage-logged and planted after the initial fire burned more severely than comparable unmanaged areas, suggesting that post-fire logging and planting did not reduce future fire severity as had been suggested by some.

In chapter three, I again focused on the twice-burned landscape, but this time I used a temporal sequence of digital aerial photography plots (6.25 ha) to measure changes in shrub-stratum, hardwood, and conifer cover. I estimated the strength and nature of relationships between crown damage and several fuel, topographical, weather, and management variables. Median crown damage, including damage to the shrub-stratum, on unmanaged plots was 63% after the Biscuit Fire and was most strongly related to damage in the Silver Fire. Plots that burned severely in the Silver Fire and had succeeded to a mix of shrubs and tree regeneration experienced high levels of Biscuit Fire damage. Plots dominated by large conifer cover after the Silver Fire had the lowest levels of Biscuit Fire canopy damage. Median crown damage was 39% for conifer cover and 85% for hardwood cover, and was most strongly related to daily average temperature and "burn period," an index of fire weather and fire suppression effort. Damage in the tree-stratum was largely independent of Silver Fire severity. Plots that had experienced stand replacing fire in 1987 and then were logged and planted with conifers had median crown damage of 100%. Plots that experienced a stand replacing fire but were unmanaged had median crown damage of 95%. The managed areas were at higher topographical positions and had greater total pre-fire cover, which may explain the small difference. These results suggest that in productive, fire-prone landscapes, the patch mosaic of young regenerating forest created by mixed-severity fire can structure the severity pattern of future wildfires occurring at short intervals and support the previous studies findings that post-fire logging and planting did not reduce fire severity.

In Chapter four, I expanded my focus to include the entire region burned by the Biscuit Fire and again used digital aerial photos taken before and after the fire to interpret patterns of crown damage and relate them to several fuel, topographical, weather, and management variables. Ninety-eight percent of plots experienced some level of crown damage, but only 10% experienced complete crown damage. The median level of crown damage on unmanaged plots was 74%. Median conifer damage was 52%. The most important predictors of total crown damage were the percentage of pre-fire shrub-stratum vegetation cover and average daily temperature. The most important predictors of conifer damage were average daily temperature and burn period. Increasing levels of shrub-stratum cover were associated with increasing levels of conifer damage and hardwood damage. Large conifers had 32% median crown damage while small conifers had 62% median crown damage. Owing largely to widespread shrub-stratum cover, low-productivity ultramafic soils had 92% median crown damage compared to 59% on non-ultramafic sites. Patterns of damage were similar within the area that burned previously in the 1987 Silver Fire and edaphically comparable areas without a recently history of fire. Median crown damage in conifer plantations was 89% and plantation age was, by far, the most important predictor of the level of damage. Plantations under 20 years old experienced the highest rates of damage. I conclude that weather and vegetation conditions—not topography—were the primary determinants of Biscuit Fire crown damage. These findings suggest that in productive fire-prone ecosystems, fuel treatments that open tree canopies and stimulate shrub-stratum development may be counterproductive.

Patterns of Crown Damage within a Large Wildfire in the Klamath-Siskiyou Bioregion

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

Jonathan R. Thompson, Author

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CONTRIBUTION OF AUTHORS

Jonathan Thompson was primarily responsible for all aspects of this dissertation. Tom Spies assisted with the study design and writing of chapters 2-4. Lisa Ganio assisted with the study design and data analysis of chapter 2.

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CHAPTER 1: INTRODUCTION

Wildfire is the most influential natural disturbance within temperate forest ecosystems (Bond and van Wilgen 1996, Barnes et al. 1998). The frequency, intensity, extent, and seasonality of wildfires, over long time-frames (i.e. the fire regime) has a profound influence on forest composition, structure, and successional pathways (Johnson 1992, Agee 1993, Turner et al. 1994, Franklin et al. 2002) and is a significant evolutionary force (Bond and Keeley 2005). Despite the importance of fire regimes for structuring forest ecosystems, universal theories regarding their causes and consequences have been difficult to develop (Pyne et al. 1996).

In recent years, the occurrence of several large, intense wildfires has raised concerns that land use and fire suppression have fundamentally altered historical fire regimes (Agee and Skinner 2005, Jain and Graham 2007). In response, forest policy makers have directed managers to restore fire resilience and resistance to forest ecosystems, primarily through thinning small trees and prescribed burning (Stephens and Ruth 2005). While it is clear that forest structure has been altered throughout the West due to past management, including fire suppression, timber harvest, and tree planting, it is it not clear that these changes are the principle cause of increasing fire-size and severity. Warmer temperatures and an earlier spring snowmelt have been linked to increasing large-wildfire frequency, longer wildfire durations, and longer wildfire seasons (Westerling et al. 2006). In addition, mounting evidence suggests that the idealized high-frequency, low-severity regime thought to characterize many fire-prone forests (Covington 2000), was not as widespread as once thought (Baker et al. 2007). High and mixed severity fires were common historically, even in fire-prone ecosystems, yet the fuel conditions that gave rise to these events are poorly understood (Schoennagel et al. 2004, Hessburg et al. 2005, Baker et al. 2007).

While the chemical process of combustion and the pattern of fire spread through fuel are predictable in a laboratory (Rothermel 1972), stochasticty is the rule within actual wildfires. Fire behavior is a product of complex interactions between weather, topography, and fuels (Agee 1993, Finney 2005). Together these three factors make up the “fire environment” (Pyne et al. 1996) and they largely dictate the effect of fire on vegetation, (i.e. burn severity, sensu Agee (1993)). The relative importance of each component varies within and between fires but some generalities can be made. When fire weather is extreme (i.e. high temperatures, low humidity, and high wind speeds), the influence of fuel and topography on fire severity is reduced (Bessie and Johnson 1995, Pyne et al. 1996). However, some empirical evidence suggests that forest conditions and topography can still be important determinants of burn severity, even during periods of extreme drought and fire weather (Bigler et al. 2005, Finney et al. 2005). Topography can interact with weather to affect fire behavior by altering wind speed and direction (Rothermel 1972) and by differentially affecting fuel moisture on topographical positions and aspects with greater solar radiation (Albini 1976, Kushla and Ripple 1997). Forest structure and composition vary with climate and topography and influence fuel characteristics, such as crown bulk density, crown base height, and fuel moisture and continuity, which, in turn, affect fire behavior and severity (Rothermel 1972, Agee 1993, Sandberg et al. 2001, Agee et al. 2002, Graham et al. 2004). Understanding the relative contribution of the weather, topography and forest composition and structure (i.e. fuel) has been identified as a critical research priority for fire scientists (Schmoldt et al. 1999) and is of great interest to forest managers who are tasked with planning fuel treatments and suppressing future fires.

I used the Biscuit Fire as a case study to address this issue. The Biscuit Fire was ignited by lighting on July 13, 2002 and burned for more than 100 days. By the time rains finally extinguished the fire, it had encompassed more than 200,000 hectares of southwest Oregon and northwest California (Rogue Siskiyou National Forest 2004), making it the largest wildfire in modern Oregon history. It burned through wide range of conditions, including: > 65,000 ha of unproductive forests on ultramafic soils, >38000 ha that had burned 15 years prior during the 1987 Silver Fire, and > 8000 ha of conifer plantations ranging from < 10 to > 40 years old. Ever since autumn rains extinguished the fire, there has been considerable speculation and sometimes intense debate regarding the pattern of vegetation damage caused by the Biscuit Fire. Conflicting reports in the popular press have added to the confusion regarding the fire's effects. For example, an Associated Press headline on September 25th 2002 read: "For its size, Oregon Biscuit Fire did little severe burn damage." This stands in contrast a quote from Oregon Senator Gordon Smith who, two years after the burn, said: "The Biscuit Fire of 2002 was the largest in our state's recorded history and burned almost 500,000 acres in Southwestern Oregon. Today, nearly half of the

Siskiyou National Forest remains a charred moonscape." (Eugene Register Guard, Sept. 19, 2004). As a result, some believe the fire burned with uncharacteristic severity resulting from five decades of fire suppression and active management, while others see the Biscuit Fire as a characteristically mixed-severity burn occurring in an ecosystem well adapted to fire. While no one study can put this debate to rest, this dissertation gives a thorough accounting of the pattern of vegetation damage caused by the Biscuit Fire and analyzes the fire environment that gave rise to the burn mosaic. Throughout this dissertation, I progressively increased the level of detail and geographical scope.

In chapter two I analyzed burn severity patterns within a large region of the Biscuit Fire that burned fifteen years earlier during the 1987 Silver Fire. Both fires burned heterogeneously, creating mosaics of live and dead trees in variably sized patches. In the three years following the Silver Fire, more than 800 hectares were salvage-logged and planted with conifers. The arrangement of these disturbances presented a unique opportunity to address the potential for recent fire and post-fire management to influence future fire severity. Indeed, after noting that no previous research had examined this issue McIver and Ottmar (2007) wrote:

> "…probably our best alternative means of understanding how fuels generated by postfire logging influence short-term fire risk is to conduct retrospective studies in forests that have burned twice within a 25-year time period, in which we can measure fire severity in stands that were either logged or un-logged after the first burn. An excellent opportunity is the Biscuit Fire itself, where a portion of the fire burned over the Silver Fire of 1988 [sic]"

Chapter two heeds their call. I used the Landsat-based differenced normalized burn ratio (dNBR) as a metric of burn severity and a hypothesis-testing framework to address two research questions: First, was severity in the Biscuit Fire associated with severity in the Silver Fire in unmanaged areas? And second, did areas that were salvaged-logged and planted with conifers after the Silver Fire burn more or less severely in the Biscuit Fire than comparable unmanaged areas?

In chapter three I continued my focus on the Silver-Biscuit reburn. But, unlike chapter two, where I statistically controlled the effects of many covariates so I could test my hypotheses, in this analysis I wanted to describe how multiple factors interacted to dictate the pattern of severity across different vegetation types. I increased my ecological resolution beyond the dNBR metric of burn severity and used a temporal sequence of digital aerial photos, taken in 1987, 2000, and 2002, to document the layering of disturbances and the pattern of vegetation damage among the three dominant cover types: conifers, hardwoods, and shrub-stature vegetation (a mixture of shrubs and regenerating trees). My research objectives were: (1) To describe the relative importance of weather, topography, and the fuel legacy of the 1987 Silver Fire on patterns of crown damage created by the 2002 Biscuit Fire. (2) To compare patterns of damage between areas that were salvage-logged and planted after the Silver Fire to areas that experience stand-replacing fire but were unmanaged, with respect to weather, topography and fuel structure.

In chapter four, I expanded the geographic scope of my analysis to include the entire Biscuit Fire region. Again, I used a sequence of pre- and post-fire aerial photo plots to measure changes in vegetation composition and structure. The overarching goal of this chapter was to describe the relative importance of weather, topography, and fuel for predicting patterns of Biscuit Fire crown damage. In addition, the unique vegetation patterns along with diverse fire and management histories within the Biscuit region, led to several research questions regarding the affect of fuel conditions on crown damage: (1) What was the relative importance of weather, topography, and fuel for predicting patterns of crown damage? (2) Did patterns of crown damage differ between cover types and did the presence of some cover types affect the level of damage in other cover types in close proximity? (3) Did the pattern of crown damage differ between areas with and without a recent history of fire? (4) Did the pattern of crown damage differ between ultramafic and non-ultramafic soils? (5) Did the pattern of crown damage differ between plantations and unmanaged forests and how did plantation age affect the level of damage?

Eminent fire ecologist Jim Agee, upon whose research I have leaned heavily throughout the dissertation, once summarized the prevailing theme of this line of research in verse. He wrote:

Whether the weather is cold
Or whether the weather is hot
Topography and fuels
Are part of the rules
Whether we like it or not
(Agee 1997)

CHAPTER 2: REBURN SEVERITY IN MANAGED AND UNMANAGED VEGETATION IN A LARGE WILDFIRE.

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ABSTRACT

Debate over the influence of post-wildfire management on future fire severity is occurring in the absence of empirical studies. We used satellite data, government agency records, and aerial photography to examine a forest landscape in southwest Oregon, USA that burned in 1987 then was subject, in part, to salvage-logging and conifer planting before it re-burned during the 2002 Biscuit Fire. Areas that burned severely in 1987 tended to re-burn at high severity in 2002, after controlling for the influence of several topographical and biophysical covariates. Areas unaffected by the initial fire tended to burn at the lowest severities in 2002. Areas that were salvage-logged and planted after the initial fire burned more severely than comparable unmanaged areas, suggesting that fuel conditions in conifer plantations can increase fire severity despite removal of large woody fuels

INTRODUCTION

Large wildfires are increasingly common in western North America (Westerling et al. 2006). Changing climate patterns and the legacy of fire suppression within fire-prone forests suggest that this trend will continue. Post-fire management is, therefore, a growing concern for public land managers. Although it has been customary to salvage-log fire-killed trees and plant seedlings after large wildfires, there is a mounting debate regarding the practice (McIver and Starr 2001, Gorte 2006, McIver and Ottmar 2007). There are several reasons one might choose this management system, including recouping economic losses through timber sales and ensuring the reestablishment desirable tree species. Another common justification for this approach has been a perceived reduction in future fire risk associated with the removal of dead wood (Brown et al. 2003, Rogue Siskiyou National Forest 2004, Sessions et al. 2004, Gorte 2006). The threat of severe re-burns is real but not well understood (McIver and Starr 2001). For example, Oregon's Tillamook burns of the 1930's, 1940's and 1950's consisted of one large fire followed by three re-burns, six, twelve, and eighteen years later—in sum these fires burned more than 135,000 hectares. The threat of re-burns motivates public land managers to construct fuel-breaks and to salvage-log in order to hedge against the risks of future fire (Rogue Siskiyou National Forest 2004). Recent studies have found, however, that salvage logging can increase surface fuels available to fires above pre-logging levels by transferring unmerchantable material to the forest floor, suggesting that this post-fire management practice might actually increase fire risk for a time (Donato et al. 2006, McIver and Ottmar 2007). Until now, no study has quantified how recent fire history and post-fire management actually affects the severity of a large wildfire (McIver and Starr 2001).

The 2002 Biscuit Fire was among the largest forest fires in modern U.S. history, encompassing more than 200,000 hectares primarily within the Rogue-Siskiyou National Forest (RSNF) in southwest Oregon. In the years following, the Biscuit Fire has been a catalyst for a national debate regarding forest management in the aftermath of wildfires on public land. This debate is taking place in the absence of empirical research on how future wildfire severity is associated with past wildfires and how post-fire forest management alters future fire severity (McIver and Starr 2001). We analyzed burn severity patterns within 18,000 hectares of the Biscuit Fire that burned fifteen years earlier during the 1987 Silver Fire. Both fires burned heterogeneously, creating mosaics of live and dead trees in variably sized patches. In the three years following the Silver Fire, more than 800 hectares were salvage-logged and planted with conifers. The arrangement of these disturbances presented a unique opportunity to address two important research questions: First, was severity in the Biscuit Fire associated with severity in the Silver Fire in unmanaged areas? And second, did areas that were salvaged-logged and planted with conifers after the Silver Fire burn more or less severely in the Biscuit Fire than comparable unmanaged areas?

With regard to the first question, hereafter referred to as the re-burn question, a negative correlation between Biscuit and Silver Fire severity is plausible, if the forests that burned severely in 1987 had less remaining fuel to support the Biscuit Fire in 2002, or if regenerating young forests did not effectively carry fire. This relationship has been observed in lodge pole pine ecosystems (Despain and Sellers 1977, Romme 1982, Turner et al. 1999). An alternate hypothesis is that Biscuit Fire severity would be positively correlated with Silver Fire severity. This would occur if areas of higher Silver Fire severity had greater accumulations of fire-killed trees and vegetative growth available as fuel to the Biscuit Fire. This scenario is assumed to have influenced forest dynamics in more mesic forests of the Pacific Northwest (Agee 1993). Finally, there may be no discernable association between the severity patterns of the two fires. Many independent factors influence fire severity, including weather, topography, fuel, landscape structure, and fire suppression. Any of these could overwhelm the signal from the legacy of the Silver Fire.

The second question, hereafter referred to as the salvage-plant question, also has several plausible outcomes. The hypothesis that salvage-logging followed by planting conifers can reduce future fire severity is widely held and rests on the assumption that removing dead trees reduces fuel loads, and planting conifers and controlling competing vegetation hastens the return of fire-resistant forests (Brown et al. 2003, Rogue Siskiyou National Forest 2004, Sessions et al. 2004, Gorte 2006). An alternative hypothesis is that salvage-logging plus plantation creation exacerbates future fire severity. No studies have measured fire severity following salvage-logging, but it is known that it can increase available fine and coarse fuel loads if no fuel treatments are conducted (Donato et al. 2006, McIver and Ottmar 2007). In addition, several studies have documented high severity fire within young conifer plantations, where surface fuels can be fine, homogeneous, and continuous (Weatherspoon and Skinner 1995, Odion et al. 2004, Stephens and Moghaddas 2005).

Our study area is within the Siskiyou Mountains in southwest Oregon’s mixed-conifer and mixed-evergreen hardwood zones (Fig. 2.1 (Franklin and Dyrness 1988)). To estimate fire severity, we calculated the differenced normalized burn ratio (dNBR (Lutes et al. 2004); Fig. 2.1) from Landsat Thematic Mapper data acquired before and immediately after each fire (Table 2.1). dNBR is a unitless index that corresponds strongly to decreasing aboveground green biomass, as well as scorched and blackened vegetation; to lesser degree, dNBR corresponds to changes in soil moisture and color and to consumption of down fuels (Lutes et al. 2004). dNBR is an effective measure of burn severity within forested landscapes (Miller and Yool 2002, Brewer et al. 2005). We reconstructed post-Silver fire management history with the help of RSNF personnel, agency documents, and aerial photography. For our analysis, logging followed by planting was considered a single management system.

RESULTS

To address the re-burn question we randomly sampled the non-managed portion of the study area. We controlled for factors known to influence fire severity by constructing the best possible geostatistical regression models of covariates (Tables 2.2 and 2.3), before adding the variable of interest—Silver Fire severity. Akaike information criteria (AIC) identified two "best" regression models of covariates. The first model included elevation, slope, plant association group (PAG; (Hobbs et al. 1992)), day-of-burn, and 1986 greenness (a satellite-based metric associated with vegetation density (Crist and Cicone 1984, Cohen et al. 1995)). The second model contained all the previous variables plus a measure of topographic position (Table 2.2). We selected the second model because topographic position is known to influence severity patterns elsewhere (Agee 1993, Taylor and Skinner 2003). Using this as our full covariate model, we then added Silver Fire severity as an independent variable and found that it was significantly and positively correlated with Biscuit Fire severity ($p < 0.0001$, d.f. = 381; Fig. 2.2). An increase of 100 dNBR within the Silver Fire was associated with an increase of 84 dNBR with in the Biscuit Fire after controlling for the covariates (95% C.I. = 69 to 99 dNBR points). We confirmed that this relationship holds even for the low to moderate range of Silver Fire severity by re-analyzing the data after excluding the samples that burned at high Silver Fire severity. Overall, unburned areas or those that burned at lower severities in the Silver Fire tended to burn at lower severities in the Biscuit Fire while areas that burned at higher severities in Silver Fire tended to re-burn at higher severities in the Biscuit Fire.

To address the salvage-plant question, (i.e. did logged and planted areas burn more or less severely than comparable unmanaged areas?) we restricted our second random sampling to areas that burned in the upper 20 percent of the Silver Fire severity range and to the logged then planted areas. In addition, we only sampled within those plant associations that contained managed stands. These constraints ensured that we were comparing the logged and planted sites only to similar areas which also experienced a stand-replacing disturbance. Again, there were two best covariate models. One included: Silver Fire severity, elevation, slope, PAG, day-of-burn, and 1986 greenness. The other included all these covariates plus a measure of topographic position; and again we chose the model that included topographic position (Table 2.2). When added to this model, the indicator for the salvage-logged and planted sites was associated with a 182.3 point increase in Biscuit Fire dNBR (Fig. 2.3; $p < 0.0001$; 95% C.I. = 120.32 to 243.68; D.F. = 282). Biscuit fire severity in the logged and planted areas was 16 to 61 percent higher than comparable unmanaged areas depending on the values of the covariates. The particular ecological effects of this difference are unknown; nonetheless, the hypothesis that salvage-logging then planting reduces re-burn severity is not supported by these data.

DISCUSSION AND CONCLUSION

No previous study has compared fire severity in plantations and naturally regenerated vegetation of similar ages. Our findings are consistent with studies that show site history influences fire severity (Weatherspoon and Skinner 1995, Finney et al. 2005, Raymond and Peterson 2005), and with studies that have found an association of high severity fire with conifer plantations (Weatherspoon and Skinner 1995, Odion et al. 2004, Stephens and Moghaddas 2005). Our limited knowledge of the fuel characteristics at the time of the Biscuit Fire prevents us from separating the effects of logging and planting. The relative influence of these management actions on burn severity would vary over time: the influence of dead fuels and harvest debris would diminish as they decayed (McIver and Ottmar 2007) and the influence of live vegetation would increase as it developed. The patterns we observed apply to the particular conditions and history of post-Silver Fire management; they could change with shorter or longer intervals between fires.

The Biscuit Fire tended to burn at relatively high severity in young naturally regenerated stands, and even more severely in young conifer plantations of comparable age and fire history. This suggests that young forests, whether naturally or artificially regenerated, may be vulnerable to positive feedback cycles of high severity fire, creating more early-successional vegetation and delaying or precluding the return of historical mature forest composition and structure. Although patches of high severity fire and re-burns are a normal part of the mixed-severity fire regime within this forest type (Agee 1993, Taylor and Skinner 1998), increasing occurrence of wildfire driven by climate warming in this region (Westerling et al. 2006) may lead to increases in the prevalence of sclerophyllous species, which are adapted to frequent severe fires (Franklin and Dyrness 1988, Hobbs et al. 1992, Agee 1993).

Our findings are inconsistent with the hypothesis that this particular post-fire management system reduces the risk of high-severity fire in a re-burn occurring 15 years after the original fire. The logging component of this system is often considered a fuel-reduction treatment (Brown et al. 2003, Rogue Siskiyou National Forest 2004, Sessions et al. 2004, Gorte 2006), however, the large diameter fuels removed during harvest do not readily carry wildland fire (Anderson 1982, Agee 1993). Thus, logging may not reduce available fuels. In fact, harvesting fire-killed trees may increase available surface fuels by transferring unmerchantable material, such as tops, branches, and broken boles to the ground immediately after harvest (Donato et al. 2006, McIver and Ottmar 2007). This effect may be mitigated as logging slash decays, or through fuel reduction methods, such as broadcast-burning (Weatherspoon and Skinner 1995). Records of site preparation and their effectiveness in reducing fuels in the plantations are incomplete; however, at least 17 of the 44 plantations are reported as “broadcast-burned.” In a separate analysis, we found that these 17 plantations also burned with higher severities than comparable unmanaged stands. The planting component of the system is intended to promote long-term regrowth of conifer trees, but it also creates dense or continuous fuels that are at elevated risk of high severity fire (Stephens and Moghaddas 2005). It should be noted, however, that many of the plantations examined in this analysis had lower conifer densities and a larger component of shrubs and hardwoods than would be found in typical intensively managed plantations of the same age (11 to 14 yrs). Our analysis could not measure any details regarding differences in pre-burn structure and composition between the natural and artificially regenerated stands. Nonetheless, the naturally regenerated areas received no site preparation or planting; therefore, they likely contained a more diverse arrangement of young vegetation and open gaps (Hobbs et al. 1992, Shatford et al. 2007). Although these naturally regenerated areas also supported relatively high severity fire, abrupt changes in fuel profiles, which can slow fire spread (Agee 1993), may have reduced the average burn severity.

We currently lack general conceptual models or simulation models that can help us understand the effects of salvage logging on fire severity over large landscapes and long time frames. As our work indicates, research needs to consider all the components of post-fire management systems, individually and together. Thus far, the few studies that have examined re-burn potential in salvage-logged sites have emphasized the dead woody fuel. This is only part of the fire risk story—and it may not be the most important after a few years. On public land, salvage-logging is almost always followed by conifer planting, even when the objective is ecological recovery, such as expediting the return of old-growth forests. We are currently unable to examine the short and long-term tradeoffs associated with different post-fire management systems. For example, we do not know how the apparent difference in fire hazard between plantations and natural stands that we observed at 15 years varies over time and if this short-term risk is balanced in any way by longer term benefits in terms of stand development and reduced fire risk. However, the available evidence suggests that the combined influence from a pulse input of surface fuels resulting from salvage-logging (Donato et al. 2006, McIver and Ottmar 2007) followed by the establishment of uniform young plantations may increase susceptibility to severe re-burns in the early stages of forest development.

Managers may have few options to reduce the risk of high severity fire within areas that have recently burned severely. Typical fuel treatments, such as thinning, do not have much effect on fire risk in young forests (Stephens and Moghaddas 2005). Reducing connectivity of surface fuels at landscape scales is likely the only way to decrease the size and severity of re-burns until vertical diversification and fire resistance is achieved (Agee et al. 2000). The decision to salvage-log and plant, or not, after fire depends on a number of management considerations including risk of future high-severity fire, reducing hazards to fire fighters, timber revenue, and conservation of biodiversity. Further research, especially controlled experiments, is clearly needed to help managers understand tradeoffs. Given the difficulty of conducting experiments with large wildfires, it is important that good records of management actions are kept, so that more can be learned from future wildfires.

METHODS

Study Area

We limited our samples to the northern half of the Silver Fire, outside of the Kalmiopsis Wilderness Area, where an aerial photo record exists to gauge the accuracy of our characterization of fire severity and Landsat data were of sufficient atmospheric quality to map fire severity. The study area is 18,050 ha centered at -123º89' W latitude, 42º49' N longitude. The vegetation is characterized by mixed conifer and mixed-evergreen hardwoods (Franklin and Dyrness 1988), dominated by *Pseudotsuga menziesii, Lithocarpus densiflorus, Pinus lambertiana Abies concolor, Chrysolepis chrysophylla, Ceanothus velutinus, and Quercus chrysolepis*. The region has steep climatic, edaphic, and topographical gradients and is renowned for floristic diversity (Whittaker 1960). Much of the landscape has high forest productivity compared to other fire-prone ecosystems in the western U.S. (Waring et al. 2006). Topography is steep and complex; elevations range from 100 to over 1,500 meters. Soil parent materials include igneous, meta-sedimentary, and metamorphic types. The climate is Mediterranean, with dry, warm summers and wet, mild winters. Mean January temperature is 6ºC. Mean July temperature is 16ºC. Mean annual precipitation is 210 cm. Approximately 80 percent of the area falls within relatively dry Douglas-fir and tanoak plant association groups (PAG), which historically burned at ten to 50 year intervals at low and mixed severities (Rogue Siskiyou National Forest 2004). Most of the remaining 20 percent falls within moist tanoak PAGs, where the fire regime is characterized as mixed severity with 50 to 100 year return intervals. Effective fire suppression began in 1940, and the dry PAGs are thought to have missed one or more fire cycles. The Silver Fire was ignited by lightning on August 30, 1987, and burned generally from the northeast to the southwest (Siskiyou National Forest 1988). The Biscuit Fire re-burned the region of the Silver Fire beginning July 17, 2002 and continuing through August 18, 2002, burning generally from east to west (Rogue Siskiyou National Forest 2004).

Image processing

Our procedures for image rectification, and atmospheric correction and normalization were as follows: A Landsat Thematic Mapper scene acquired immediately after the Biscuit Fire (Table 2.1) was rectified to a 2003 USGS Digital Orthoquad (DOQ), using more than 150 tie points, and a first-order polynomial transformation, which produced a 14.0 meter RMSE. Three corresponding TM scenes, one acquired a year before the Biscuit Fire, one acquired immediately after the Silver Fire, and one acquired a year prior to the Silver Fire (Table 2.1), were then all co-registered to the 2002 Landsat image using the Landsat Orthorectification tool in ERDAS Imagine version 8.7. Each rectification utilized a ten-meter digital elevation model and more than 1200 tie points, which were located using an automated tie-point finder (Kennedy and Cohen 2003); RMSE errors were less than one-half pixel in all cases. Our Landsat images have a 29 meter resolution and were registered in 1927 North American Datum, UTM Zone 10. The two pre-fire images were then converted to reflectance and atmospherically corrected using the COST method (Chavez 1996). We then used an automated ordination algorithm called multivariate alteration detection (Canty et al. 2004, Schroeder et al. 2006) that statistically located pseudoinvariant pixels, which were subsequently used in a reduced major axis regression to radiometricly normalize post-fire to pre-fire images. We selected October imagery, despite the low sun angle in autumn, because we wanted to pair our imagery to the dates of historical aerial photos to aid an assessment of accuracy and because we wanted to capture the fire's effects without the confounding influence of spring green-up that may have occurred had we used imagery from the following summer. Moreover, the use of near anniversary dates within a change-detection of normalized vegetation indices greatly minimizes any negative effects of late-season imagery.

Burn severity and initial vegetation condition

Our measure of fire severity was the differenced normalized burn ratio (dNBR; (Lutes et al. 2004)). It is a measure of pre- to post-fire change in the ratio of near-infrared (B4, 0.76 - 0.90μm) to shortwave infrared (B7, 2.08-2.35μm) spectral reflectance. B4 is associated with foliage on green trees and understory, while B7 is associated with dry and blackened soil (Lutes et al. 2004). dNBR compares well to ground data (Miller and Yool 2002, Brewer et al. 2005) and has outperformed other satellite derived measures of burn severity (Brewer et al. 2005). Using the processed images described above, we calculated dNBR as described in: (Lutes et al. 2004). Normalization of pre- and post-fire imagery centers the unchanged (i.e. unburned) pixels within each fire to a dNBR value of zero. A comparison of dNBR values to aerial photography suggests that dNBR values are roughly equivalent in terms of their correspondence to vegetation damage; however, the severity maps were constructed independently to maximize the accuracy for each fire. Because our images were acquired immediately after the fire, our estimate of severity does not capture any vegetation that may have experienced delayed mortality or re-greening in the years subsequent to the fire.

We excluded areas with ultra-mafic soils from this analysis because they had an anomalous spectral response and are ecologically distinct from the rest of the landscape (<4% percent of the study area). The fire weather indices Burn Index and Energy Release Component were calculated from Quail Prairie remote weather station data (~ 25km south of the study area) by the Oregon Department of Forestry. A digital map of plant association groups was provided by the RSNF and included in the model selection procedure to control for differences in biophysical characteristics (e.g. productivity and plant composition). In addition, tasseled cap wetness, greenness and brightness indexes (Crist and Cicone 1984) were derived from the processed 1986 Landsat TM data and were used during model selection to control for differences in pre-Silver Fire in vegetation condition. Tasseled cap indexes are closely related to forest composition and structure in Pacific Northwest forests (Cohen et al. 1995).

We used the continuous dNBR data for all statistical analysis. We also constructed categorical burn severity maps to assess the accuracy of the satellite data in relation to characterizations of tree crown damage from high resolution aerial photos. The categorical Silver Fire severity map was also used, in part, to define the sampling universe for the salvage-plant question. Three levels of severity were classified: 1. Unburned/Low Severity, where less than ten percent of the crown was scorched or consumed by the fire; 2. Moderate Severity, where ten to fifty percent of the crown was scorched or consumed; and 3. High Severity, where greater than fifty percent of the of the crown was scorched or consumed. Using a set of 109, randomly placed, high resolution, digital aerial photo plots, we developed thresholds in dNBR values that best classified the data for each fire. An independent sample of 141 aerial photo plots was used in an accuracy assessment, which resulted in overall estimate of user's accuracy of 83 percent for unburned/low severity pixels, 75 percent for moderate severity pixels, and 85 percent for high severity pixels.

Forest Management Data

We identified 44 management units (approximately 850 hectares; Fig. 2.1) that were logged in the three years following the Silver Fire, then planted with conifers (primarily Douglas-fir), and later certified as "successful plantations." The salvage logging guidelines set by the Forest Service required that, within harvest units, 12 to 18 standing snags > 60cm diameter and >12m tall, along with 2.8m3 of down wood be retained per hectare (Siskiyou National Forest 1988). Plantations were deemed successful if, three to five years after planting, conifers exceeded 370 stems per hectare and were considered healthy enough to survive competition with shrubs and hardwood trees. Though post Silver Fire records from the RSNF are not complete, they indicate that some certified plantations had undergone mechanical treatment to suppress competing vegetation and that conifer stocking typically ranged from approximately 600 to 1100 trees per hectare. In addition to the logged and planted areas used in the analysis, we distinguished approximately 250 hectares that were harvested in part or in full after the Silver Fire but were either not planted or the planted conifers were not certified ‘free to grow’. We excluded these later areas from all our analyses because their history was too uncertain and variable to accurately characterize. All management polygons were provided by the RSNF and were edited in a geographic information system (GIS) using one meter resolution USGS DOQ acquired in 1994 to better fit the perimeters of harvest units. Areas logged and planted prior to the Silver Fire were excluded from all analyses. The bulk of the remaining 17,000 hectares in the northern Silver Fire perimeter were not harvested. However, some small unknown proportion was selectively logged but not planted; no records describing management actions are known to exist and we could not see evidence of them in careful analysis of the DOQs. It is likely that only the largest most valuable trees were extracted from these sites via helicopter. The unrecorded, lightly salvaged sites make up no more than ten percent of the study area, and are included in the population of unmanaged sites.

Sampling, variable selection, model fitting, & hypothesis testing

All sampling and data extraction was done in a GIS, in which all the variables were converted to 29 meter raster maps. The sample universe for the re-burn question was the region within the northern Silver Fire but outside any management areas. All known management units dating back over fifty years were excluded from re-burn analysis. Sample locations were determined by randomly selecting locations for 381 points with the constraint that they be separated by at least 300 meters to reduce spatial dependence. Sample values were calculated as the mean value of the closest nine contiguous pixels to the sample location (sample unit area ~ 0.75 hectares). When sampling for the salvage-plant question, we constrained the sample universe to the areas within the northern Silver Fire perimeter, within PAGs that contained management units and either burned at high severity in 1987 and received no or minimal post-fire management, or areas that burned at high severity and were clearcut during 1988, 1989, or 1990 then planted in the years following and certified by the RSNF as an established conifer plantation. All other management units were excluded. We sampled 292 random locations (225 unmanaged and 67 managed), separated from each other by at least 300 meters. Any plantation that did not include at least one sample during the initial sample selection had a sample randomly located within it. Data were extracted from all the raster maps as described above.

Statistical analysis was completed in the computing software, R (R Development Core Team 2006). Empirical variogram models indicated spatial autocorrelation of Biscuit dNBR data; a spherical theoretical variogram model best described the autocorrelation. This spatial dependence precluded the use of ordinary least squares regression. Instead, for model selection and hypothesis testing, we used generalized least squares (GLS) regression to fit linear models of predictor variables to Biscuit dNBR data. GLS models allow residuals to have a nonstandard covariance structure (Venables and Ripley 1997); we used a spherical spatial correlation structure. The modeled variogram from the reburn data had a range of 2314 meters and the nugget:sill was 0.387. The modeled variogram from the salvage-plant data had a range of 1096 meters and the nugget:sill was 0.399. Akaike information criterion (AIC) was used for model selection (Akaike 1973, Burnham and Anderson 2002). For both questions, we evaluated AIC scores from approximately 100 a priori candidate covariate models, including global models containing all non-correlated predictor variables and null models that contained none. All candidate models contained plausible combinations of predictor variables based on what is known about fire behavior and from previous studies of burn severity patterns on other fires (Table 2.3).

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Table 2.1. Acquisition dates of satellite imagery used to estimate fire severity

Landsat TM (Path 46 Row 31)	**Date Acquired**
Pre-Silver Fire	10/13/1986
Post-Silver Fire	10/16/1987
Pre-Biscuit Fire	10/10/2001
Post-Biscuit Fire	10/6/2002

Table 2.2. Variables included and retained during regression model selection for the Re-burn and salvage-plant questions. † = retained in full covariate regression model after AIC model selection; ♦ = The variable being tested after fitting the best covariate regression model; NBR = Normalized Burn Ratio; DEM = Digital Elevation Model; RSNF = Rogue Siskiyou National Forest; ODF = Oregon Department of Forestry; RAWS = Remote Automated Weather Station

Variables	Reburn	Salvage -plant	Definition
Disturbance History			
Silver Severity	♦	†	Differenced NBR from the Silver Fire, calculated from pre- and post-fire Landsat TM
Managed site		♦	Salvage-logged in 1988, 1989 or 1990, planted with conifers, then later certified as a successful plantation
Topography			
Elevation	†		In meters from a 10m DEM
Aspect			Aspect folded around south facing slopes (folded aspect = |180-|aspect-180||), from 10m DEM
Slope	†	†	In percent, from 10m DEM
Topo-position (Fine)	†	†	Difference between sample elevation and mean elevation of an annulus spanning 150 to 300m from the sample
Topo-position (Mid)			Difference between sample elevation and mean elevation of an annulus spanning 850 to 1000m from the sample
Topo-position (Coarse)			Difference between sample elevation and mean elevation of an annulus spanning 1850 to 2000m from the sample
Biophysical			
Plant Association Group	†	†	Potential vegetation in the absence of disturbance, based on climatic, biogeographical and physiographic factors; obtained from RSNF
1986 Brightness			Brightness axis from tasseled cap transformation of 1986 Landsat data - Associated with the pre-Silver Fire soil and litter color, vegetation cover, and hardwood cover
1986 Wetness			Wetness axis from tasseled cap transformation of 1986 Landsat data - Associated with pre-Silver Fire forest structure
1986 Greenness	†	†	Greenness axis from tasseled cap transformation of 1986 Landsat TM data - Associated with the pre-Silver Fire vegetation cover and dense conifer or hardwood canopies
Soil			Soil data from Curry and Josephine counties (4 soil types)
Weather & other temporal change			
Burn Index			Daily fire behavior index measured using current and past weather data, heavily influenced by wind speed, Calculated by the ODF using data from the Quail Prairie, Oregon RAWS
Energy Release Component			Daily fuel moisture index that reflects the contribution of live and dead fuels to potential fire intensity, Calculated by the ODF using data from the Quail Prairie, Oregon RAWS
Day	†	†	The day on which the Biscuit Fire flaming front passed (a count from the first to the last day); obtained from RSNF

Table 2.3. Ten best generalized least squares regression models for each research question as determined by differences in AIC statistics

The Reburn Question

Rank	Δ AIC		Model
1	0.00		Biscuit Severity ~ 1986 Greenness + Day + Elevation + PAG + Slope
2	1.61	*	Biscuit Severity ~ 1986 Greenness + Day + Elevation + PAG + Slope + TP-F
3	2.66		Biscuit Severity ~ 1986 Brightness + Day + Elevation + PAG + TP-F
4	2.73		Biscuit Severity ~ 1986 Brightness + Day + Elevation + PAG + Slope + TP-F
5	2.91		Biscuit Severity ~ 1986 Greenness + Day + Elevation + PAG + TP-F
6	3.82		Biscuit Severity ~ 1986 Greenness + Day + PAG + TP-I
7	6.30		Biscuit Severity ~ 1986 Brightness + Day + PAG + TP-I
8	18.23		Biscuit Severity ~ 1986 Greenness + Burn Index + PAG + TP-I
9	24.61		Biscuit Severity ~ Aspect + Day + Elevation + PAG
10	26.57		Biscuit Severity ~ 1986 Greenness + PAG + TP-I

The Salvage-Plant Question

Rank	Δ AIC		Model
1	0.00	*	Biscuit Severity ~ 1986 Greenness + Day + Elev + PAG + Silver Severity + Slope + TP-F
2	1.77		Biscuit Severity ~ 1986 Greenness + Day + Elev + PAG + Silver Severity + Slope
3	19.36		Biscuit Severity ~ 1986 Greenness + Day + PAG + Silver Severity + TP-I
4	20.77		Biscuit Severity ~ 1986 Brightness + Day + Elev + PAG + Silver Severity + Slope + TP-F
5	21.52		Biscuit Severity ~ 1986 Greenness + Day + Elev + PAG + Silver Severity + TP-I
6	21.71		Biscuit Severity ~ 1986 Brightness + Day + Elev + PAG + Silver Severity+ Slope
7	23.71		Biscuit Severity ~ 1986 Brightness + Burn Index + PAG + Silver Severity + Slope + TP-I
8	26.67		Biscuit Severity ~ 1986 Greenness + PAG + Silver Severity + TP-I
9	43.70		Biscuit Severity ~ 1986 Greenness + Silver Severity + TP-I
10	45.62		Biscuit Severity ~ 1986 Greenness + Day + Silver Severity + TP-I

Note: All models included a spherical spatial correlation structure. Models with AIC statistics less than two units greater than the lowest AIC statistic are considered equal "best" models (Burnham and Anderson 2002). * = Selected covariate model; PAG = Plant Association Group; TP-I = Topographic position-intermediate resolution; TP-C = Topographic position-coarse resolution; TP-F = Topographic position-fine resolution; See Table 2.2 for variable descriptions.

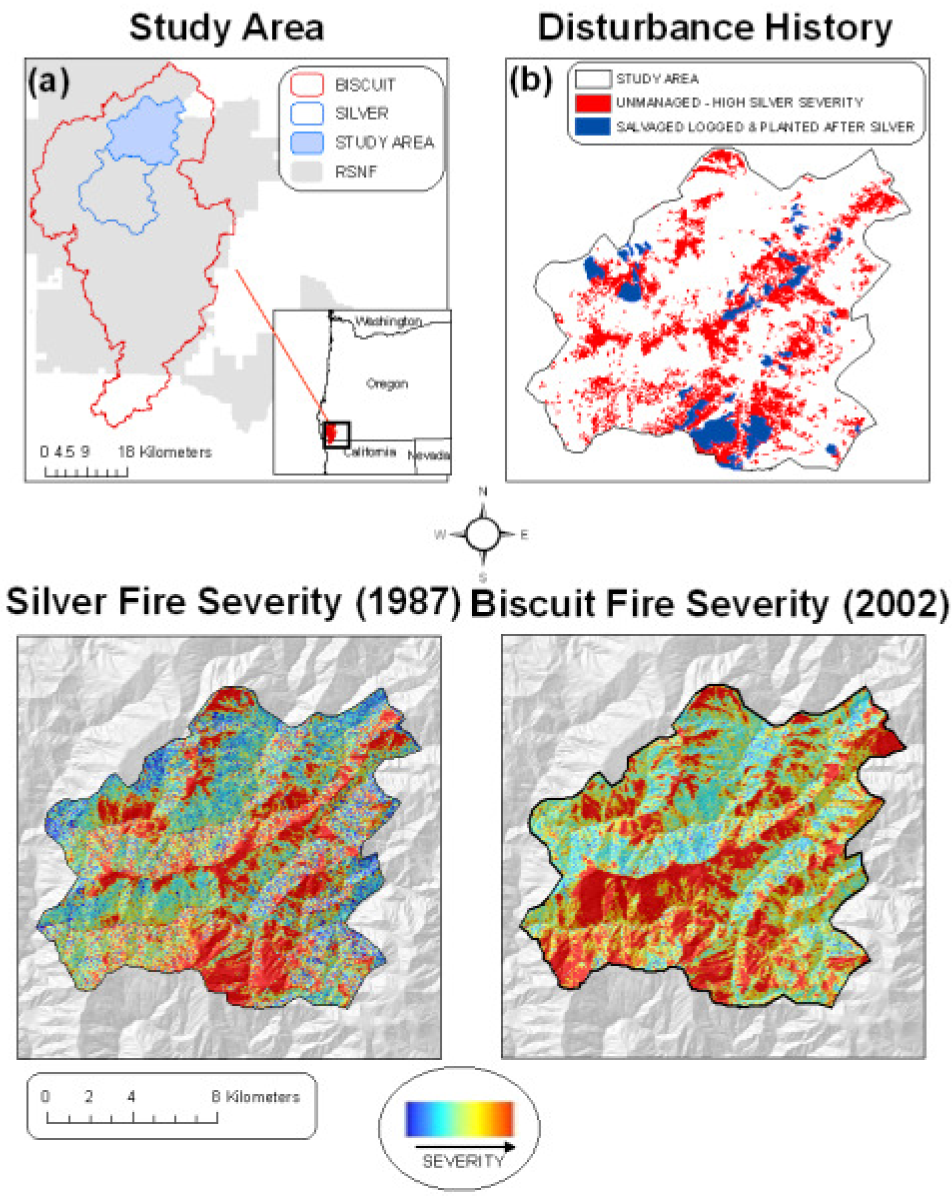

Figure 2.1. Maps of study area and fire severity (a) The study area in context of recent fires. (b) Sampling universe for the salvage-plant question (c) Burn severity of the 1987 Silver Fire (d) Burn severity of the 2002 Biscuit Fire

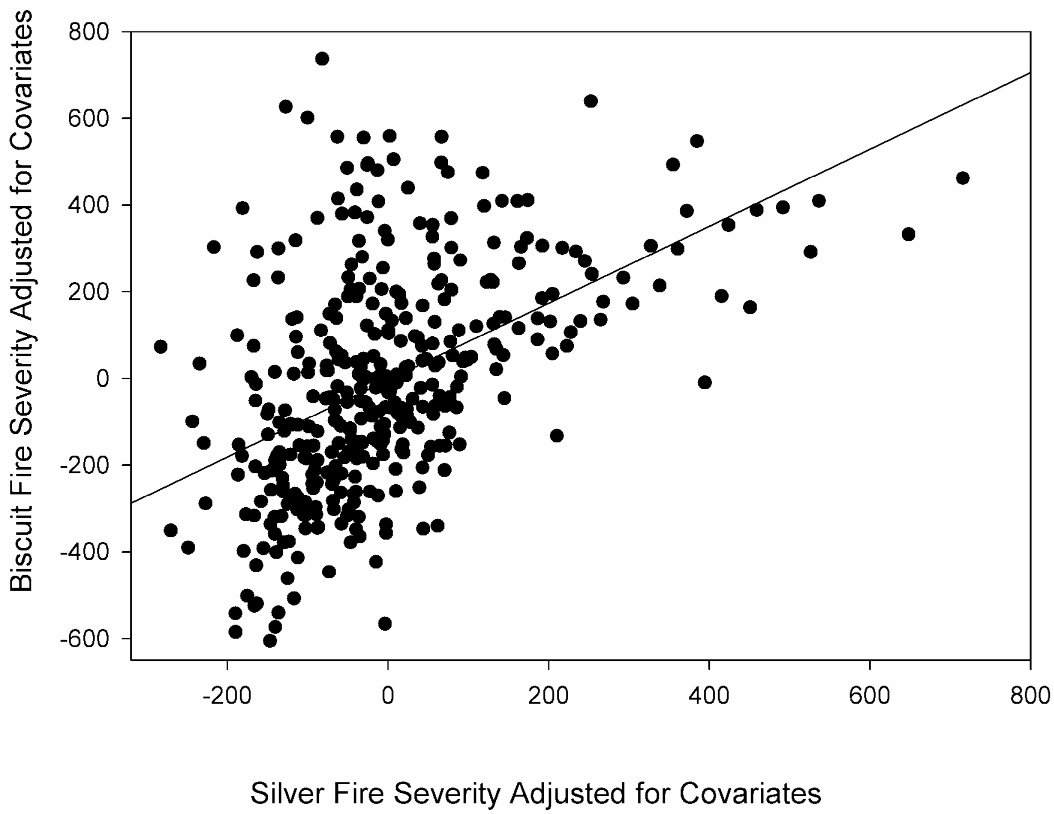

Figure 2.2. Added variable plot displaying the relationship between Silver and Biscuit Fire severity as estimated through a Landsat derived burn metric, dNBR. The effect of elevation, slope, plant association group, day-of-burn, 1986 greenness, topographic position on Biscuit Fire severity has been removed to illustrate the association between fire severities from both fires. See Table 2.2 for descriptions of the covariates.

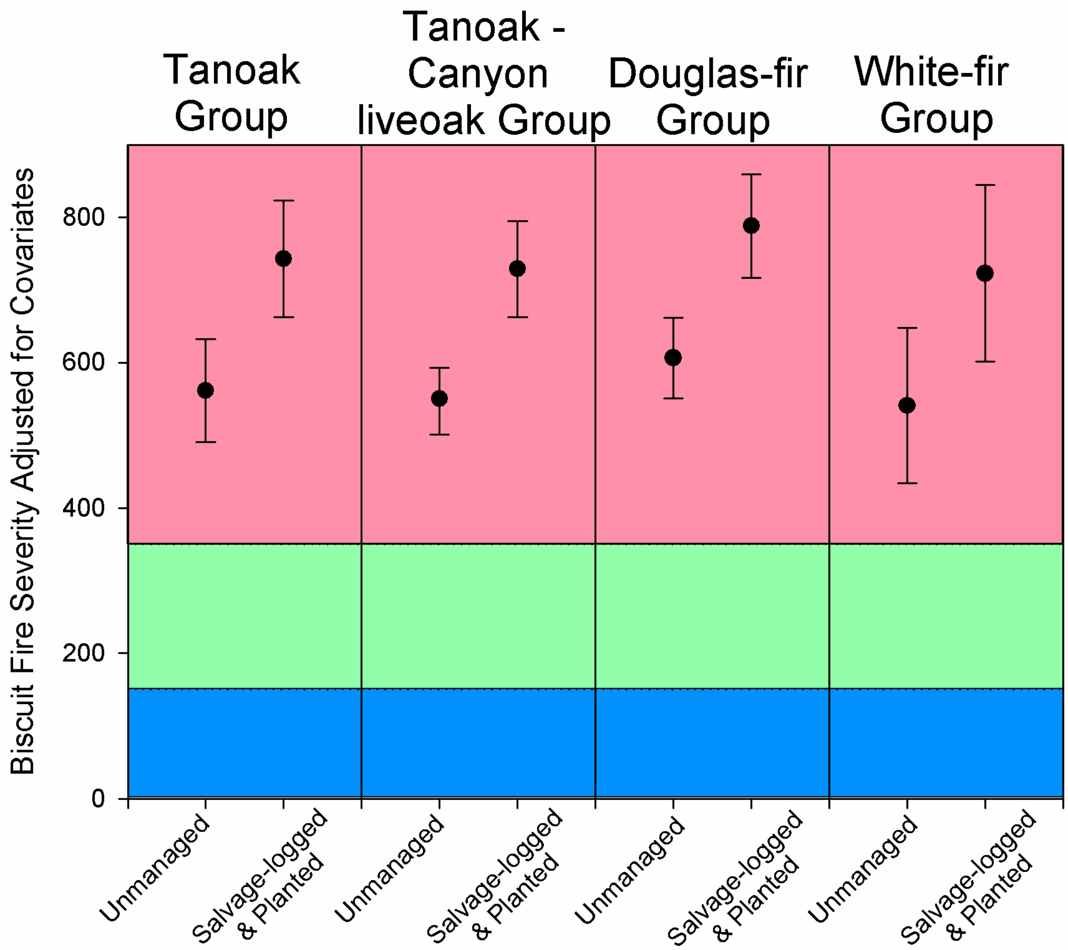

Figure 2.3. Estimates and confidence intervals comparing regions burned at high severity in the 1987 Silver Fire that were unmanaged to areas that were salvage-logged and planted following the Silver Fire. 95% confidence intervals for Biscuit Fire dNBR, a Landsat derived burn severity metric. Means were calculated at the multivariate centroid of the covariates: slope, elevation, topographic position, Silver Fire severity, day-of-burn, and 1986 greenness (Table 2.2). Means were similar across the four plant association groups. In this landscape, the Tanoak group is found on wetter sites in the western portion of the study area while the Tanoak-Canyon live oak group is found on dryer, inland sites; the Douglas-fir group is found on relatively dry sites and the White-fir group is found at somewhat higher elevation wetter sites (Hobbs et al. 1992). Colors correspond to Biscuit Fire severity calibrated through comparison with aerial photography: Blue = < 10% canopy scorch; Green = 10-50% canopy scorch; Red = >50% canopy scorch.

CHAPTER 3: FACTORS ASSOCIATED WITH CROWN DAMAGE FOLLOWING RECURRING MIXED-SEVERITY WILDFIRES AND POST-FIRE MANAGEMENT

ABSTRACT

Wildfires have a persistent influence on the quantity and arrangement of live and dead fuels, which can, in turn, affect the behavior of future fires. Similarly, post-fire logging and planting influences fuel characteristics and can also affect future fire behavior. In 2002, the Biscuit Fire reburned 38,000 hectares of mixed conifer/evergreen hardwood forest that had burned heterogeneously in the 1987 Silver Fire and then was subject, in part, to logging and planting. Using a temporal sequence of digital aerial photo-plots (6.25 ha), I estimated the strength and nature of relationships between crown damage and several fuel, topographical, weather, and management variables. Median crown damage, including damage to the shrub-stratum, on unmanaged plots was 63% after the Biscuit Fire and was most strongly related to damage in the Silver Fire. Plots that burned severely in the Silver Fire and had succeeded to a mix of shrubs and tree regeneration experienced high levels of Biscuit Fire damage. Plots dominated by large conifer cover after the Silver Fire had the lowest levels of Biscuit Fire canopy damage. Median tree crown damage was 39% for conifer cover and 85% for hardwood cover, and, for both tree cover types, was most strongly related to daily average temperature and "Burn Period," an index of fire weather fire and fire suppression effort. Damage in the tree-stratum was largely independent of Silver Fire severity. Plots that experienced stand replacing fire in the Silver Fire, and then were logged and planted with conifers had median crown damage

of 100%. Plots that experienced stand replacing fire in 1987 but were unmanaged, had median crown damage of 95%. The managed areas were at higher topographical positions and had greater total pre-fire cover, which may explain the small difference. These results suggest that in productive, fire-prone landscapes, the patch mosaic of young regenerating forest created by mixed-severity fire can structure the severity pattern of future wildfires and that post-fire salvage logging and planting does not reduce future fire severity, at least in the short term.

INTRODUCTION:

In the western United States, wildfire has been the dominant disturbance shaping forest ecosystems (Agee 1993, Pyne et al. 1996, Barnes et al. 1998). The frequency, intensity, extent, and seasonality of fires, over long time-frames, (i.e. the fire regime) has a profound influence on forest composition, structure, and successional pathways (Johnson 1992, Agee 1993, Turner et al. 1994, Paine et al. 1998). Individual wildfires have variable effects on vegetation (i.e. fire severity) and tend to increase the spatial and structural heterogeneity of live and dead fuels (Turner et al. 2003, Baker et al. 2007), which can, in turn, influence the behavior of subsequent wildfires (Agee 1993, Peterson 2002, Agee 2004). This pathway may be affected further by post-fire forest management (McIver and Ottmar 2007, Thompson et al. 2007). Although the effects of compounding disturbances remain relatively unstudied, severe forest disturbances, recurring over short time periods relative to their rate of recovery, can have qualitatively different ecological consequences than do isolated disturbances (Gray and Franklin 1997, Paine et al. 1998). Given that climate change is expected to result in more frequent and severe wildfires (Fried et al. 2004, Westerling et al. 2006), it is important to develop a better understanding of the ecological conditions that arise from compounding disturbances. I examined patterns of crown damage following recurring mixed-severity wildfires fifteen years apart: the 1987 Silver Fire and the 2002 Biscuit Fire.

The legacy of past wildfires has widely varying effects on future fire behavior. In some ecosystems, wildfire may have the short term effect of reducing future fire severity. For example, in low severity regimes, such as were found historically in dry, low elevation ponderosa pine (*Pinus ponderosa*) forests, frequent surface fires reduced fuels and the risk of crown fires (Covington and Moore 1994, Fule et al. 1997, Baker et al. 2007). Similarly, in some high severity regimes, fires can reduce short term fire hazard if regenerating vegetation is less flammable than older vegetation, such as in Rocky Mountain lodgepole pine (*P. contorta*) forests (Despain and Sellers 1977, Romme 1982). (Note that fire hazard is defined here as the potential influence of fuel on fire behavior independent of weather, sensu Hardy (2005)). In contrast, in other high-severity regimes forests, such as wet Douglas-fir (Pseudotsuga menziesii) forests in Washington, elevated fuel-loads from recent stand replacing fires (Agee and Huff 1987) can lead to repeated high severity fires in rapid succession (Agee 1993, Gray and Franklin 1997). For example, the Tillamook burns in northwestern Oregon during the 1930's, 1940's and 1950's consisted of one large fire followed by three re-burns, six, twelve, and eighteen years later. Overall, the "ecological memory" of past wildfires (sensu, Peterson 2002) ranges from quite strong (Minnich 1983) to non-existent (Bessie and Johnson 1995).

In mixed-severity regimes, characterized by variable fire frequencies and heterogeneous effects within and between fires, the post-fire legacy of live and dead fuels is variable over time and space and is comparatively not well understood (Agee 2004, Schoennagel et al. 2004). In the productive mixed-conifer/mixed-evergreen hardwood forests of southern Oregon and northern California, where the Silver and Biscuit Fires occurred, the post-fire landscape is typically a mosaic of high and low severity patches, which vary widely in size (Agee 1991, Skinner 1995, Taylor and Skinner 2003, Odion et al. 2004, Alexander et al. 2006). Within severely burned patches, most biomass remains on site but is converted from live to dead, while fine surface fuels and the forest floor are largely consumed (Campbell et al. 2007, Donato 2008). Dead aerial fuels gradually fall to the surface and decompose over time. Within a few years, live surface fuels increase dramatically, as shrubs and hardwoods resprout and conifer trees regenerate from seed, often at high densities (Stuart et al. 1993, Lopez-Ortiz 2007, Shatford et al. 2007, Donato 2008). Although there has been far less empirical research, patches that burn at low severity are thought to have reduced surface fuels while sustaining the thick-barked, high-crown "resistor" species typified by many conifer species such as Douglas-fir (Agee 1993, Odion et al. 2004). The patch mosaic created by a mixed-severity fire can structure and reinforce the severity pattern within future fires (Peterson 2002, Wimberly and Kennedy 2008).

Of course, wildfire is not controlled by fuels alone; topography and weather also affect fire behavior (Pyne et al. 1996). Wind, humidity, and temperature can override the influence of fuels by supplying oxygen and reducing the energy needed to sustain combustion (Rothermel 1972). Weather also interacts with topography, differentially affecting fuel moisture on aspects with greater solar radiation and steeper slopes. In some cases, particularly in high severity regimes, weather can be the dominant control of fire behavior (Turner et al. 1994, Bessie and Johnson 1995); that notwithstanding, even in times of drought and extreme fire weather, pre-fire fuel conditions can influence fire severity (Graham 2003, Finney et al. 2005, Kulakowski and Veblen 2007). In mixed severity regimes, when fire weather is extreme, the fuel environment can have a lesser influence on severity patterns (Agee 1997); therefore the feedbacks created through previous severity mosaics may be less important.

The risk of recurring high severity fires (often called the "reburn effect") is just one of many competing concerns that managers must consider in the aftermath of a fire. Post-fire logging (i.e. salvage logging) has long been a management choice, motivated primarily by interest in economic returns and a perceived reduction in the risk of future severe fires resulting from lower fuel loads (Poff 1989, Brown et al. 2003, Sessions et al. 2004, Gorte 2006). Some recent studies have found, however, that post-fire logging can increase short-term fire hazard by increasing the availability of fine fuels (Donato et al. 2006, McIver and Ottmar 2007). Planting conifers has also been widely employed in the aftermath of wildfires to expedite the return of desired tree species and hasten the return of fire resistant forests (Sessions et al. 2004). This practice, too, may elevate short-term fire hazard if planting increases the availability and continuity of fine fuels (Stephens and Moghaddas 2005). Several observational and modeling studies have documented the high severity fire within plantations (Weatherspoon and Skinner 1995, Odion et al. 2004, Stephens and Moghaddas 2005, Thompson et al. 2007), even when conifers are planted at low densities (Roloff et al. 2004).

I capitalized on a unique arrangement of disturbances to address questions of reburn severity and post-fire management. I examined a landscape in southwest Oregon's Siskiyou Mountains that burned heterogeneously during the 1987 Silver Fire, then was subject, in part, to salvage logging and planting, before reburning in the 2002 Biscuit Fire. In an earlier analysis of the same landscape, Thompson et al. (2007) used the Landsat-based differenced normalized burn ratio (dNBR; Key and Benson 2004), and found that areas that burned at high severity in 1987 tended to re-burn severely in 2002. Conversely, areas that burned at low severity in 1987 tended to reburn at the lowest severities. Further, they showed that areas that were salvage-logged and planted after the Silver Fire burned somewhat more severely in the Biscuit Fire than did areas that burned severely in the Silver but were left unmanaged. dNBR correlates well with overall vegetation damage (Appendix A; Miller and Yool 2002, Lutes et al. 2004, van Wagtendonk et al. 2004), and it, along with a relativeized version, RdNBR (Miller and Thode 2007), are commonly used for quantifying landscape-scale burn effects (e.g. Odion et al. 2004, Bigler et al. 2005, Finney et al. 2005, Collins et al. 2006, Kulakowski and Veblen 2007, Safford et al. 2007, Wimberly and Reilly 2007). However, dNBR and RdNBR can not effectively distinguish between the type or structure of burned vegetation. At high levels of dNBR, changes in the index may be more associated with surface soil features (e.g. ash, soil color) than with canopy mortality, which reaches 100% before the maximum level of dNBR is reached.

In this analysis, I increased ecological resolution beyond the Thompson et al. (2007) analysis by using a temporal sequence of digital aerial photography to document the layering of disturbances and the pattern of vegetation damage among the three dominant cover types: conifers, hardwoods, and low stature vegetation (a mix of shrubs and small trees, hereafter called the shrub-stratum, sensu Sandberg et al. (2001)). Like Thompson et al. (2007), I examined the relationship between 1987 Silver Fire severity and post-Silver management with Biscuit Fire severity; additionally, I estimated the relative importance and the nature of relationships between Biscuit Fire crown damage and several aspects of its fire environment and management history. My specific research objectives were:

1. To describe the relative importance of weather, topography, and the fuel legacy of the 1987 Silver Fire on patterns of crown damage created by the 2002 Biscuit Fire.
2. To compare patterns of damage between areas that were salvage-logged and planted after the Silver Fire to areas that experience stand-replacing fire but were unmanaged, with respect to weather, topography, and fuel structure.

METHODS

Study Area:

The analysis was limited to the 21,000 hectares that make up the northern half of the 1987 Silver Fire, centered at 123°89'W latitude 42°49'N longitude (Fig. 3.1). I excluded the southern half due to the absence of aerial photo record for that region in 1987 and 2002. The study area is managed by the Rouge-Siskiyou National Forest (RSNF), and has high floristic diversity (Whittaker 1960). The landscape is within the mixed evergreen zone (Franklin and Dyrness 1988), and is dominated by conifer species such as Douglas-fir, sugar pine (*P. lambertiana*), white fir (*Abies concolor*), and knobcone pine (*P. attenuata*). Dominant evergreen hardwoods include tanoak (*Lithocarpus densiflorus*), Pacific madrone (*Arbutus menziesii*), chinquapin (*Chrysolepis chrysophylla*), and canyon live-oak (*Quercus chrysolepis*) and shrubs such as manzanita (*Arctostaphylos sp.*) and ceanothus (*Ceanothus sp*.). In older stands, the sclerophyllous broad-leaved trees often form lower strata under the conifer overstory (Franklin and Dyrness 1988). Soil parent materials include igneous, meta-sedimentary, and metamorphic types. Less than 5% of the study area has ultramafic soils; these areas are floristically distinct and were excluded from this analysis. Topography is steep and complex; elevations range from 100 to 1500 m. The climate is Mediterranean, with dry, warm summers and wet, mild winters. Mean January temperature is 6ºC. Mean July temperature is 16ºC. Mean annual precipitation is 270 cm, with greater than 90% occurring in winter (Daly et al. 2002).

Fires were common throughout pre-European history with a large portion of them burning with low severity (i.e. much the conifer overstory survived most fires; Atzet and Martin 1991, Sensenig 2002). Fires remained a common disturbance through the mining era of the latter nineteenth and early twentieth centuries. In 1943, a smoke jumper base was established in Cave Junction, Oregon, on the eastern edge of the RSNF; since then, fire has been effectively suppressed, with a few punctuated exceptions in the years of 1987 and 2002 (Rogue Siskiyou National Forest 2004). The Silver Fire was the largest of more than 1600 fires ignited by a lightning in northwest California and southwestern Oregon on August 30, 1987 (Reider 1988, Hardy et al. 1992). By the time rains finally extinguished the fire, it had burned at mixed severities across more than 38,000 hectares (Fig. 3.1). The Biscuit Fire burned through and completely encompassed the region of the Silver Fire beginning on July 17, 2002 and continuing through August 18, 2002.

Image processing and interpretation:

I scanned print aerial photos taken on October 15, 1987, immediately after the Silver Fire, at high resolution (1200 dpi (20 microns)). The best available pre-Biscuit photos (August 2000) were digital orthoquads (DOQs) taken as part of the USDA National Agriculture Imagery Program (http://165.221.201.14/NAIP.html). These were my lowest quality images (1 m grain size, panchromatic) and they dictated the resolution of vegetation data that I could reliably interpret. The post-Biscuit Fire photos were taken on September 24, 2002; they were scanned directly from diapositives (1200 dpi (20 microns)).

I interpreted vegetation condition and fire effects within 181 randomly located photo-plots and 35 management units randomly selected from a database acquired from the RSNF (management data are described below). The unmanaged plots were constructed as polygons in a GIS that consisted of square a five-by-five grid of 50 m cells, totaling 6.25 ha (Fig. 3.2); plots were discarded if they contained any portion of a road, management unit or a large streams or river (stream order >3). To construct the management plots, a polygon grid of 50 m cells was overlain onto the variably shaped management units. If the unit was larger than 6.25 ha, then 25 cells were randomly selected to be used in the plot. If management units were smaller than 6.25 ha, then all cells were used. Management units smaller than 1.25 ha were excluded.

A temporal photo sequence for each plot was spatially co-registered using approximately fifteen ground control points (GCP) to link 1987 and 2002 photo-plots to the 2000 DOQs within Erdas Imagine 8.7. GCPs were concentrated in and around the plot, while the remainder of the photo was ignored and later clipped out of the image. Individual trees were the most common tie-points, but rock outcroppings and other topographic features were also utilized. After the GCPs were placed, a first-order polynomial transformation was used to geo-rectify the photo. The resulting grain size for the 1987 and 2002 photos was ~ 0.30 meters.

To conduct photo interpretation, I overlaid the plots onto the geo-referenced aerial photos in ESRI ArcMap 9.2. I interpreted vegetation structure at all three points in time and fire damage (scorch or consume) in the 1987 and 2002 photo-plots. Crown damage, in addition to being a direct measure of fire effects on forest structure, is also closely tied to fireline intensity and is a useful measure of a fire's effect on an ecosystem (Van Wagner 1973, Weatherspoon and Skinner 1995). Percent cover of conifer, hardwood, shrub-stature vegetation, bare ground/grass, and fire-related damage were estimated for every cell in a plot. (Note that shrub-stature vegetation is a measure of all low stature vegetation including young conifers and hardwood, not simply shrub species.) Areas of shadow were subtracted from the effective area of the plot. Pre-fire conifer cover in each cell was further assigned a size class: small, large, or mixed. The small class roughly corresponded to pole or small saw timber (< 20 cm DBH) and the large class corresponded to large saw timber (> 20cm DBH; Paine and Kiser 2003). (DBH estimates were verified with a post hoc comparison of conifers in photo photos to 70 co-located Forest Service inventory plots. Inner quartile range (IQR) for DBH on small plots = 7 to 19 cm; on large plots IQR = 18 to 42 cm; IQR on mixed plots = 10 to 23cm). Cover estimates summed to 100% in each cell. Cell-level cover estimates were averaged to obtain plot-level values. To establish a metric of structural complexity for each plot, I calculated Shannon's diversity (Hill 1973) at the plot level using the seven cover classes just described. Because it was impossible to reliably discern grass from bare ground in the 2000 photos, and because bare ground doesn't burn, I included a constraint that forced the percent bare/grass to be equal in 2002 and 2000. All photo interpretation was conducted by a single researcher (Thompson) to ensure consistency and limit error. At the onset of the research, I created a catalog of paired oblique-to-aerial photos for use as a training manual. Later, I ground truthed a subset of the photo-plots, which revealed excellent correspondence between field and photo measurements.

Topographic and weather variables

Using a 10-meter digital elevation model (DEM), I calculated average elevation, percent slope, aspect, and topographic position for each photo-plot. Beer's transformed aspect (Beers et al. 1966) was calculated for all plots and varies from -1 on NE facing slopes with little incident sunlight to 1 on SW facing plots receiving the abundant incident light. An index of topographic position (TP) was calculated at two scales. TP-Fine is the absolute difference in elevation between each pixel in the DEM and the average elevation in an annulus 150 to 300-meters from the pixel, while TP-Coarse uses an annulus 850 to 1000m from the focal pixel. The RSNF provided a daily fire progression map showing the area burned each day by the Biscuit Fire. I assigned weather data to each plot based on data from the Quail Prairie RAWS station, which is approximately 25 km south of the study area (RAWS data accessed from http://www.wrcc.dri.edu/fpa/). Average temperature, relative humidity, wind speed, and wind direction between 10:00 and 17:00 were assigned to each plot based on the day on which it burned. This technique, though temporally and spatially coarse, has been successfully used in other reconstructions of fire effects (Collins et al. 2007). Before averaging, wind direction was cosine transformed such that a value of -1 corresponded to winds out of the southwest and a value of +1 corresponded to winds out of the northeast, which are typically drier and associated with severe fire weather in this region (USDA Forest Service 2002).

I also created a variable that divided the burn area into three “Burn Periods,” which correspond to the spread of the Biscuit Fire and fire suppression effort during each Period (USDA Forest Service 2002, GAO 2004). Period A, represents 5% of the total Biscuit Fire area (7% of the study area) and includes the region that burned from July 13 to July 26 with comparatively little suppression effort and mild weather conditions (Table 3.1). Period B includes the region that burn from July 27 to Aug 04; 50%of the Biscuit Fire burned in this nine day period (46% of the study area), which was characterized by strong north-northeastern winds and low relative humidity. Flame lengths > 30 m and spread rates > 2500 m per hour were frequently reported during this period. Suppression resources increased during this period but were largely unsuccessful in preventing fire spread. Period C represents the remaining 45% of the Biscuit Fire (47% of study area) that burned through the study area from August 5 to 18. Fire suppression activities were extensive throughout Period C. The fire continued to spread during extreme weather but had a higher potential to be influenced by fire fighters, including extensive but undocumented controlled burns.

Management Data

All the management units were logged in the three years following the Silver Fire, then planted with conifers (primarily Douglas-fir) and later certified as “successful plantations.” The Silver Fire salvage logging guidelines set by the Forest Service required that, within harvest units, 12 to 18 standing snags > 60cm diameter and >12m tall, along with 2.8m^3 of down wood be retained per hectare. Plantations were deemed successful if, three to five years after planting, conifers exceeded 370 stems per hectare and were considered healthy enough to survive competition with shrubs and hardwood trees. Though post-Silver Fire records from the RSNF are not complete, they indicate that some certified plantations had undergone mechanical treatment to suppress competing vegetation and that conifer stocking typically ranged from approximately 600 to 1100 trees per hectare. All areas logged and planted prior to the Silver Fire were excluded from analyses. The bulk of the remaining ~20,000 hectares in the northern Silver Fire were not harvested. However, some small unknown proportion was selectively logged but not planted; no records describing management actions are known to exist and I could not see evidence of them in careful analysis of the photos. It is likely that only the largest most valuable trees were extracted from these sites via helicopter. The unrecorded, lightly salvaged sites make up no more than 10% of the study area, and are included in the unmanaged portion of the study area.

Data Analysis

I plotted empirical cumulative distributions for the percentage of plots with evidence of crown damage after each fire. (Note that this is a slightly different metric of fire damage than is used throughout the rest of the analysis. Typically, Biscuit Fire crown damage is a percentage of the pre-fire available crown cover; however, because I didn't have pre-Silver Fire cover data, I used the percentage of the plot area with evidence of fire. It is a minor distinction that only changes the estimates when there is substantial cover of bare/grass). I then summarized the proportion of conifer, hardwood, and shrubs burned in the Biscuit Fire at five-percent increments of Silver Fire canopy mortality.

Within the unmanaged plots, I structured three response variables that describe different aspects of canopy damage during the Biscuit Fire: total canopy damage, relative conifer damage (i.e. (2000 Conifer Cover – 2002 Conifer Cover) / 2000 Conifer Cover), and relative hardwood damage. I examined relative measures of conifer and hardwood damage because absolute measures of damage were roughly proportional to availability (e.g. hardwood damage was high where hardwood cover was high) and because relative damage isolates the influence of fuel, weather, and topographical setting on the individual vegetation components. When modeling relative hardwood damage, I subset the data to include only those plots with greater than 5% pre-fire hardwood cover (n = 107). I took a two-stage approach to analyzing the relationships between the response variables and the suite of predictor variables (Table 3.2). I first used random forest analysis (RFA; Breiman 2001) to estimate and rank the importance of predictors, and then used regression tree analysis (RTA) to illustrate the nature of relationships between the response and important predictor variables. These nonparametric methods are ideally suited for the analysis of high dimensional ecological data with hierarchical, complex, and non-linear relationships to response variables (De'ath and Fabricius 2000, Cutler et al. 2007). I am aware of no previous application of RFA to fire effects data, but RTA (and its analytical cousin, classification tree analysis) has been used extensively to model relationships between fire severity and predictor variables (e.g. Finney et al. 2005, Alexander et al. 2006, Collins et al. 2007, Lentile et al. 2006, Jain and Graham 2007, Kulakowski and Veblen 2007).

RFA is an ensemble learning algorithm (ELA) that averages predictions over multiple bootstrapped regression trees. (An ELA is any method that runs a base algorithm multiple times to construct a set of hypotheses, called an ensemble, which is then used to "vote" to predict the value of new data.) I used Liaw and Wiener's (2002) implementation of RFA within the R statistical language (R Development Core Team 2006); the algorithm, as applied to these data, was as follows: (1) Select 1500 bootstrap samples with replacement, each containing 63% of the data. (2) For each bootstrap sample, grow an un-pruned regression tree with the modification that at each node, rather than implementing the best split among all predictors, randomly select one-third of the predictor variables (six, in my case) and choose the best split from among those variables. (By selecting from a subset of predictors, RFA forces diversity among regression trees, which reduces bias in variable selection and reduces variance in the averaged prediction. Selecting among approximately one-third of the total number of predictors was suggested by Breiman (2001), and was found to be optimal for these data). (3) At each bootstrap iteration, predict the response value for data not included in the bootstrap sample—the so-called Out-Of-Bag or OOB data—and average those response values over all trees. (Because the OOB data are not used when building the trees, their estimates are essentially cross-validated accuracy estimates.) (4) Calculate importance values for each predictor by calculating the percent increase in mean squared error (MSE) when OOB data for each variable are permuted while all others are left unchanged. The MSE is computed as:

$$MSE_{OOB} = n^{-1}\sum_{1}^{n}\left\{y_i - \hat{y}_i^{OOB}\right\}^2$$

where $\hat{y}_i^{OOB}$ is the average of the OOB predictions for the *i*th observation. The "percent variance explained" is computed as:

$$1 - \frac{MSE_{OOB}}{\hat{\sigma}_y^2}$$

RFA is increasingly being used to analyze ecological data and, where it has been tested, has consistently out-performed other statistical methods, including RTA, for prediction accuracy (Garzon et al. 2006, Lawler et al. 2006, Prasad et al. 2006, Cutler et al. 2007, Peters et al. 2007). However, RFA has been termed a "grey box" model because it lacks full transparency (Prasad et al. 2006). There is no simple representation of the relationship between response and predictor variables; one cannot examine the individual regression trees, nor are there P-values, regression coefficients, or confidence intervals that accompany many traditional statistical techniques. However, RFA does produce several metrics that aid interpretation, including variable importance and partial dependence. I used variable importance measures to rank the predictors in terms of the strength of their relationship to the response. I also used partial dependence plots to show the effect of changing individual predictors while holding all other predictors at their average. Partial dependence plots are useful to visualize low-order interactions within the multivariate range of the predictor variables; however, they should not be used to interpret relationship outside the range of the data or where there are complex interactions between predictors and the response (Hastie et al. 2001, Cutler et al. 2007).

After identifying important predictor variables with RFA, I used RTA to better explain the nature of relationships between the six top-ranked predictor variables and the response variables. RTA is a non-parametric technique that recursively partitions a dataset into subsets that are increasingly homogeneous with regard to the response (Breiman et al. 1984, De'ath and Fabricius 2000). RTA produces a set of decision rules on predictor variables that can be easily interpreted as a dendrogram. Most implementations of RTA tend to over fit to a given dataset by creating splits that do not significantly reduce the variance. Trees are typically pruned back to include only those partitions assumed to be valuable beyond the sample data. I used an implementation of RTA called conditional inference trees, which forces statistical significance at each split (Hothorn et al. 2006). This technique prevents over-fitting and the need for pruning. The algorithm, as applied to these data, was as follows: (1) Test the null hypothesis of independence between any of the predictor variables and the response. Stop if this hypothesis cannot be rejected. Otherwise, select the input variable with strongest association to the response. This association is measured by a p-value estimated from a Monte Carlo randomization test of a single input variable and the response. Due to the tendency of spatially dependent data to inflate the significance of hypothesis tests (Dale 1999) I set the minimum criteria for applying a split conservatively at $p < 0.005$ (see below). (2) Implement a binary split in the selected input variable. (3) Recursively repeat steps one and two.

To evaluate the association between crown damage and management history, I pooled the management plots (n=35) with the portion of unmanaged plots (or contiguous portions of unmanaged plots greater than 1.25 hectares) that experienced complete overstory mortality during the Silver Fire (n=35). I compared the damage to the regenerated shrub-stratum vegetation between the managed and unmanaged plots with a Monte Carlo test (Gotelli and Ellison 2004) with 10,000 randomizations to compute a p-value describing the probability of encountering differences in median levels of crown damage that are at least as large as those observed, given the distribution of data. I then used RFA and RTA to examine the relationship between predictors and canopy damage while including management history as a potential predictor variable.

The potential for spatial autocorrelation to affect statistical tests should be considered in all analyses of spatial phenomena (Legendre 1993, Dale 1999), particularly fire effects (Bataineh et al. 2006). Fire severity has positive spatial autocorrelation, which results in increased occurrence of Type-I errors (incorrectly rejecting H0). Ideally, samples should be spaced beyond the range of autocorrelation. In the case of the Silver/Biscuit Reburn, however, significant spatial autocorrelation in crown damage existed at distances greater than 3000 m; therefore, it would have been impossible to simultaneously collect a sufficient sample and adequately disperse sample units. Given this constraint, I took several steps to examine the effects of autocorrelation and limit exposure to its negative consequences. To gaurd against Type-I errors within the RTA algorithm, which required a criterion for partitioning, I set α conservatively to 0.005 (sensu, Dale and Zbigniewicz 1997). I examined the degree of autocorrelation in the RFA and RTA model residuals using empirical semivariograms. With spatially partitioned predictor variables, regression trees are spatially heterogeneous in functional form; in other words, the decision rules can change based on their location on the landscape (McDonald and Urban 2006). As a result, residuals from the RTA and RFA models had low levels of autocorrelation compared to the raw severity data (see results). For the Monte Carlo randomization test, I reported an exact p-value to provide useful information, rather than predetermining an ecologically meaningful level for α, which can be unknowable in the face of autocorrelation (Fortin and Dale 2005).

RESULTS:

Overall pattern of crown damage

Within the unmanaged plots, median crown damage was 16% in the Silver Fire and 63% in the Biscuit Fire (Table 3.2; Fig. 3.3). The average coefficient of variation for within-plot crown damage was 1.05 after the Silver Fire and 0.60 after the Biscuit Fire. (Note that the estimates for Silver Fire damage do not account for the 850 hectares that burned severely in 1987 then were salvage-logged and planted—that region was not part of my sample for “unmanaged stands.” As such, as an estimate of median crown damage for the entire study area, it is biased downward by approximately 2%.) Plots with the highest levels of canopy damage in the Silver Fire also had the highest levels of canopy damage within the Biscuit Fire (Fig. 3.4). Plots that had been severely burned by the Silver Fire were dominated by shrub-stratum vegetation and contained low levels of tree-stratum cover at the time of the Biscuit Fire. The highest levels of damage in the tree-stratum (conifer and hardwood) during the Biscuit Fire were in areas of lowest Silver Fire damage (Fig. 3.4). The shrub-stratum cover that regenerated after the Silver Fire experienced the largest proportional damage (95%), of the cover types (Fig. 3.5). Between the two tree strata cover types, hardwoods experienced a greater proportional loss of canopy than conifers (85% versus 39%, respectively; Table 3.2, Fig.3.5).

Total crown damage models

The RFA model explained 46% of the variability in total crown damage. Crown damage during the Silver Fire and large conifer cover were identified as the most important predictor variables (Fig. 3.6). Increasing Silver Fire damage was associated with increasing Biscuit crown damage (Fig. 3.7). Increasing large conifer cover was associated with decreasing crown damage in the Biscuit (Fig. 3.7).

The RTA of total crown damage produced 5 terminal nodes (Fig. 3.8). The first partition was based on whether Silver Fire crown damage was > 39%; when it was Biscuit damage was generally > 90%. Plots that burned during period B, when the average temperature was greater than 31° C had similarly high levels of damage. Plots with >50% large conifer cover after the Silver Fire experienced the lowest levels of crown damage.

Semivariograms showed substantial spatial autocorrelation in the crown damage data to lag distances greater than 3000m, but that much of the spatial dependency was explained by the predictor variables in the RFA and RTA analyses (Fig. 3.9). This pattern of low spatial autocorrelation in the model residuals was consistent for all response variables that included spatially structured predictor variables (weather variables and burn period).

Conifer and Hardwood Damage

RFA models explained 32 and 18% of the variability in relative conifer and hardwood damage, respectively (Fig. 3.6). Weather variables and burn period were ranked as most important in both cases. The first split in the regression tree of relative conifer damage was on burn period and indicated lower levels of damage during burn periods A and C (Fig. 3.10). Conifer damage was highest during period B when the average temperature was above 31°C. When temperatures were cooler burn periods A, B, and had similar levels of damage. The first split in the RTA for hardwood damage was related to average daily temperature, but overall, patterns were similar to those in relative conifer damage (Fig. 3.11).

Management History

The median value of pre-fire shrub-stratum cover was higher in the salvage-logged and planted plots (95%) than in the plots that burn severely in the Silver Fire but were unmanaged (86%; Fig. 3.12). After the Biscuit Fire, the managed stands had lower median cover (0%) than the unmanaged (5%). The Monte Carlo randomization test for median differences in proportional damage produced a p-value of 0.0008 (Fig.

3.12). With the managed and unmanaged data pooled, RFA explained 37% of the variability in crown damage and identified two measures of topographic position and management history as the most important predictors (Fig. 3.13). Higher topographic position and management was associated with higher crown damage. Consistent with this finding, the first split in the regression tree was on TP-Fine; plots on lower topographic positions had median crown damage of 93% and included only unmanaged plots (Fig. 3.14). Among plots with higher topographic positions, an additional split was based on whether shrub cover was above 79%. To better examine the effect of management, I constructed separate boxplots for managed and unmanaged areas at the terminal nodes of the regression tree, even though the management variable was not identified as a split in the model.

DISCUSSION

The Biscuit Fire resulted in higher levels of canopy damage than did the Silver Fire. However, this was not necessarily a result of it being a reburn. In fact, Biscuit-related crown damage in edaphically similar (i.e. non-ultramafic) areas outside the reburn area also exceeded the level of Silver Fire damage (see Chapter 4). Differences in the overall severity of the two fires may largely be a result of differences in weather conditions at the time of burning. The Silver Fire burned late in the season, when weather conditions were comparatively mild (Table 3.1). No daily progression map exists for the Silver Fire so I was unable to link changes in Silver Fire behavior to daily weather conditions. However, average daily weather conditions favored greater fire spread and severity during the Biscuit Fire (Table 3.1).

Despite differences in weather, there was substantial evidence that the legacy of the Silver Fire had an influence on the severity of the Biscuit Fire. Consistent with Thompson et al. (2007), there was a trend of increasing Biscuit severity with increasing Silver severity (Fig. 3.4). Here, I have expanded on that analysis to show that areas that experienced repeated high severity fire were dominated by shrub-stratum vegetation. Shrub-stratum fuels are available to surface fires, have high surface-area-to-volume ratios, and are associated with flashy and sometimes intense fire (Albini 1976, Anderson 1982, Graham et al. 2004). Early successional pathways in this region are often dominated by sprouting hardwoods and shrubs (Hobbs et al. 1992, Stuart et al. 1993, Hanson and Stuart 2005, Lopez-Ortiz 2007), and although conifers will usually succeed (Shatford et al. 2007), the period in which most live biomass remains in the shrub-stratum and is vulnerable to repeated severe burning may be protracted over several decades (Donato 2008). Biscuit Fire severity within the high Silver severity areas may also have been influenced by the legacy of dead fuels, though I was unable to measure them or account for that influence. Dead biomass decays quickly and this region and fuel measures and fire modeling suggest that live fuels are the major driver of early seral fire hazard (Donato 2008).

Absolute tree canopy damage of hardwoods and conifers (as opposed to relative damage) was highest in areas that burned at low severity in the Silver Fire (Fig. 3.4). This finding may seem counterintuitive; however, it simply reflects the fact that areas that did not experience crown fire in the Silver Fire had trees canopies available to burn in the Biscuit. More in line with expectations is the finding that areas with high levels of large conifer cover after the Silver Fire experienced the lowest levels of Biscuit Crown damage. This is likely due to the elevated closed canopies and lower levels of surface fuels which characterize older conifer forests in this region. These areas may have experienced underburning in both the Silver and Biscuit Fires and, thus, may have become more resistant to canopy fire with each burn. Although almost every plot showed some evidence of fire, from aerial photos it is impossible to know definitively where a surface fire occurred under a tree canopy.

Within tree canopies, the hardwood overstory suffered greater proportional damage than the conifer overstory (Fig. 3.5). This pattern is consistent with their respective anatomies and life history strategies, and is also consistent with patterns of damage reported from a series of field plots on the west side of the fire (Raymond and Peterson 2005). Most hardwoods in this landscape are evergreen (primarily tanoak and pacific madrone), and are easily top-killed by fire due to their flammable leaves, relatively thin bark and low crowns (Brown and Smith 2000). However, they are aggressive basal sprouters in the wake of fire and can quickly dominate a site after disturbance (Tappeiner et al. 1984). Therefore, although hardwoods suffered higher levels of canopy damage, actual mortality of hardwood individuals was likely quite low. Douglas-fir is the most common conifer species in this landscape and it becomes more fire resistant with age due to thickening bark and increasing crown base height (Agee 1993). Foliar moisture, which can affect crown fire initiation and crown scorch

(Van Wagner 1973, 1977), is similar in first year tanoak and Douglas-fir foliage, while older tanoak foliage can be dryer than Douglas-fir (Raymond and Peterson 2005).

Damage to conifers and hardwoods relative to their pre-fire abundance was related primarily to weather and burn period. Period B, when spread rates were at their highest, was associated with significantly higher levels of crown damage. This finding is consistent with wide-spread accounts of torching, crowning and spotting during this period reported by Forest Service personnel (USDA Forest Service 2002). Relative damage in tree canopies was largely independent of the fuel or topographic setting, regardless of Burn Period. Relative hardwood damage did increase with the level of hardwood cover, but this predictor was comparatively weak; it ranked fourth in the RFA model (which itself only explained 18% of the variability) and did not produce a split in the regression tree. Relative conifer damage was lower in areas with greater large conifer cover, but again, this relationship was weak and did not result in a significant split in the RTA. Interestingly, Silver Fire severity was not an important predictor of relative tree-stratum damage, which suggests that the legacy of Silver Fire was primarily limited to damage in the shrub-stratum.

Topographic variables were less important predictors of tree canopy damage than were the vegetation and weather variables. This finding was surprising because topographic features such as slope, elevation, topographic position, and aspect have been associated with severity patterns elsewhere within the region (Taylor and Skinner 1998, Beaty and Taylor 2001, Odion et al. 2004). And, in a dNBR-based analysis of

Biscuit reburn severity, elevation, slope, and topographic position were included in an AIC-based model selection procedure (Thompson et al. 2007). In this case, it may be that vegetation composition and structure, which was determined by the Silver Fire and not included in the previous study, overrode, or was confounded with, the influence of topography, at least at levels detectable with this smaller sample.

Because the weather variables were tied to the day on which a plot burned and, thus, spatially partitioned, it is likely that some of the variability they explained was related the spatial autocorrelation present in the data. In an observational study, such as this, there is no way to obtain weather information that is specific to each point on the landscape at the time of burning. Nonetheless, temperature and wind speed were consistently among the most important weather variable considered, which is consistent with the findings of Collins et al. (2007), who used a similar approach for interpolating weather information across a post-fire landscape.

Fire has a persistent influence on the composition and structure of live and dead fuels that is dynamic over time and the length of the fire-free interval can have a strong influence on fire behavior. In this case, the fire-free interval was just fifteen years—well within the estimate range of the historical return interval (Atzet and Martin 1991, Sensenig 2002). Therefore, unless the effect of fire suppression on fuels was persistent and strong through the Silver Fire, the severity pattern witnessed in the Biscuit reburn should be within the “natural range of variability.” Heightened fire risk resulting from lengthening fire-free intervals (i.e. the “suppression effect”) relates most strongly to historically low-severity regimes, where frequent low severity surface fires persistently reduce fuel loads (Covington and Moore 1994, Fule et al. 1997). There is less information about the effect of the fire-free interval on severity patterns within mixed-severity regimes. In an examination of two fires in the Sierra Nevada Mountains, in a landscape that burned several times during the past 30-years, Collins et al. (2007) found the lowest levels of fire damage within areas that had experienced fire within 17 years or less. The dampening effect of previous fires existed only in forest-types typical of mixed- to low-severity regimes (Jeffery Pine, red fir, and white fir). Within the high-severity regime forests (lodge pole pine), the length of the fire free interval had no detectable effect on severity. In contrast, Odion et al. (2004) examined the severity patterns of several fires that burned in northern California during 1987 and concluded that fire severity was negatively correlated with the length of the fire free interval. These studies used fire perimeters as indicators of previous fires and did not consider the severity mosaic within each burn, which could have had a considerable influence on their findings. The different patterns witnessed between these two studies may be a result of different fire histories, namely the fact the Sierra Nevada landscapes were thought to be within their natural fire cycle while the northern California landscape had a history of fire suppression; however, conventional wisdom suggests that fire suppression would have had the opposite effect of severity patterns. Or, it may be due to the differing successional pathways, particularly the presence of sprouting hardwoods in northern California. My study area is more similar to the northern California landscape in both regards.

It may be that the mosaic of burn severity in these productive, fire-prone forests has an element of landscape inertia. Once an area burns, it can be caught within a positive feedback of repeated severe fires. Lightning strikes are ubiquitous in this region (Sensenig 2002), and can ignite wildfires that repeatedly reset succession resulting in semi-permanent shrub-fields (Agee 1993). After each burn, hardwoods conifers regenerate vigorously, setting the stage for the next severe burn (Donato 2008). Clearly, this cycle does not continue indefinitely, as is evidenced by the abundant old growth forests within the region. Periodically the fire-free interval must be sufficiently protracted to allow a tree stratum to develop. Then, once an area breaks out of the high severity feedback, it may begin to develop inertia toward low-severity burning, as forest canopies close and suppress understory vegetation and trees become larger and more resistant to fire. This feedback may strengthen until a severe fire, such as the Biscuit Fire, shifts the stand back to the early-seral, high-severity loop.

Relationship between crown damage and post-fire management

Shrub-stratum vegetation experienced high rates of crown mortality throughout the reburn. Median damage was five-percent higher in areas that had been logged and planted after burning severely in the Silver Fire. While this analysis was consistent with Thompson et al. (2007) in the respect that the managed stands burned more completely, it is reasonable to ask why the difference in median crown damage was 5% while the difference in dNBR was 16 to 61%. One possibility is that maximum dNBR is not reached at 100% scorch or consumption of vegetation canopy by fire

(Key and Benson 2004, Miller and Thode 2007). dNBR measures a synthesis of fire effects and though it is primarily related to canopy damage, it also corresponds to changes in soil moisture and color, ash color and content, and consumption of down wood (Key and Benson 2004). In this respect, to two estimates simply measure different aspects of "burn severity," a term that often means different things to different people (Jain 2004). An additional reason that the dNBR values may have been higher in the plantations is because it is a measure change from pre- to post-fire conditions. My results here show that shrub-stratum cover was higher in managed stands; as a result, they may have had a greater amount of change (higher dNBR), even when all the existing cover was consumed. Subsequent to the publication of Thompson et al. (2007), Miller and Thode (2007) published a new index, RdNBR, which is designed to help ameliorate this problem by relativizing the pre-fire estimate. I have since re-run the analysis using RdNBR, and found no changes in the results between the two indices. Finally, the lower magnitude of difference between plantations and unmanaged stands in this analysis may also be a product of having used different samples and different sample sizes.

Although the differences in canopy damage between managed and unmanaged stands with similar fire histories is small, it, nevertheless, indicates that post-fire management (salvage logging followed by planting) did not reduce fire severity, as has been hypothesized by some (Poff 1989, Brown et al. 2003, Sessions et al. 2004, Gorte 2006). The degree to which these small differences might affect subsequent ecological process is unknown. It may make no difference, whatsoever. It is also possible that small initial differences in heterogeneity of shrubs and trees and persistence of seed sources could affect the longer term structural diversity of these stands.

The RFA analysis ranked the indicator variable for management above the variable describing the cover of pre-fire shrub-stratum vegetation. In contrast, RTA included pre-fire vegetation as a split in the tree. This fact illustrates the strengths and weaknesses of the two approaches. The importance ranking from RFA indicates that, all other predictors being held constant, the fact that an area had been logged and planted, or not, explained more variability in the OOB data than did the amount of pre-fire shrub-stratum vegetation cover. In contrast, interpretation of each node in the regression tree is conditional on the nodes above it. Therefore, pre-fire vegetation cover is included as a split in the tree, only after moving down the branch related to higher topographic position. Furthermore, the importance values in RFA are not affected by collinear predictor variables, which guards against elimination of variables that are good predictors of the response (Cutler et al. 2007). RTA, in contrast, selects the best possible split and offers no information regarding other variables that may have reduced deviance almost as much as the chosen variable. In fact, when I re-ran the RTA without the pre-fire cover variable, the indicator for management was included as a split in the same location on the tree with $p = 0.026$.

The RTA analysis suggests that the difference in severity between managed and unmanaged stands was related to topographic position and pre-fire vegetation cover. Although Thompson et al. (2007) controlled for topographic position within a regression framework, they were not able to account for pre-fire cover. Shrub-stratum vegetation cover was higher within the logged and planted stands, which is consistent with some (Lopez-Ortiz 2007) but not all (Donato 2008) field studies in the region. Higher cover was presumably a result of planting; however, it may also be the case that the sites selected for harvest and planting were more productive, which may have also contributed to higher vegetation cover. Donato (2008) compared fuel structure between 5 managed stands measured 17 years after the Galice Fire (just northeast of the Biscuit Area), to 4 unmanaged stands measured 18 years after the Longwood Fire (just southeast of the Biscuit Fire) and found large compositional differences (more conifers in the managed stand) but no difference in fuel mass. Evenly spaced young conifers have been hypothesized to have fuel properties more conducive to fire spread than shrubs and young broadleaf hardwoods (Perry 1994), but no empirical research has been done. However, several studies have documented high burn severity within conifer plantations (Weatherspoon and Skinner 1995, Odion et al. 2004, Roloff et al. 2004), particularly when young (Graham 2003, Stephens and Moghaddas 2005).

CONCLUSIONS

My overall findings are consistent with Thompson et al. (2007), who used a different metric of burn severity (Landsat-based dNBR) and a different analytical technique (geostatistical regression). However, Thompson et al (2007) focused on testing two hypotheses regarding reburning and did not elaborate much beyond them. By interpreting changes in vegetation cover through a time series of aerial photos, I increased the ecological resolution far beyond what was reported in the first study. I found that areas that burned severely in the 1987 Silver Fire reburned severely in the 2002 Biscuit Fire, but that these areas contained primarily shrub-stratum vegetation. Relative damage within the tree stratum was largely independent of the legacy of the Silver Fire. Hardwoods experienced greater damage than conifers and large conifers experienced the lowest levels of damage. Areas that were salvage logged and planted after the Silver Fire experienced high rates of crown damage during the Biscuit Fire. Like all observational, retrospective analyses of wildfire, the patterns observed apply only to these specific fires and management history. According to a review of the available post-fire management literature, prior to the research on the Silver/Biscuit landscape (this work along with Thompson et al. (2007)) there had been no empirical analyses of reburn effects after post fire salvage logging planting (McIver and Starr 2001). And, to the best of my knowledge, there have been no previous studies of overlapping burn mosaics in unmanaged stands. Additional research is clearly needed to judge if these findings are generalizable to other mixed-severity reburns and to quantify the differences in reburn severity with longer and shorter intervals between fires.

The Biscuit Fire left almost 100,000 hectares of high severity patches across the RSNF. My findings suggest that the post-fire management practiced in this case does not reduce fire hazard at 15 years and may slightly increase it compared to early-seral unmanaged areas. In the short term, managers may not be able to reduce the likelihood of recurring high severity fire in these cover types through traditional silvicultural practices. Research done elsewhere within the Biscuit Fire has shown that thinning in green forests followed by prescribed fire can be an effective way to reduce fire severity in the short term, but that thinning without treating logging slash can increase severity compared with unmanaged stands (Raymond and Peterson 2005). It is unknown whether similar treatments would be effective within stands that were severely or partially burned. Managers may consider strategically placing thinning and burning treatments in configurations that might slow the spread of future fires enabling protection of key structures and habitat conditions (e.g. spotted owl habitat areas) with the landscape (Rouge Siskiyou National Forest 2004, Ager et al. 2007).

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Table 3.1. Dates, area, and weather information for burn periods distinguished by the spread of fire and the resources used for fire suppression. Weather variables are averages of the daily average between 10:00 and 17:00 within each Biscuit Fire burn period, or within the duration of Silver Fire. Wind direction has been cosine transformed such that a value of -1 is associated with southwesterly winds and a value of +1 is associated with northeasterly winds.

Burn Period	Start-Stop	No. Days	Hectares (% total)	Temperature (C°)	Relative Humidity (%)	Wind Speed (km/hr)	Wind Direction	Suppression Effort
A	7/13-7/26/2002	14	1485 (7)	23.3	43	13.0	0.08	Low
B	7/27-8/04/2002	9	9731 (46)	25.5	35	13.1	0.63	Moderate
C	8/05-8/18/2002	14	9761 (47)	27.2	22	9.8	0.66	High
Silver Fire	8/31-10-15/1987	46	21000 (100)	22.7	37	7.9	0.02	Unknown

Table 3.2. Summary statistics for response and predictor variables used in the random forest and regression tree analysis of unmanaged stands. Weather variables were determined using inverse distance weighting from weather stations surrounding the burn area (see text for details).

Response Variables	**Median**	**Mean**	**Min**	**Max**
All Crown Damage	62.6	58.4	0.0	100.0
Relative Conifer Damage	38.6	45.8	0.0	100.0
Relative Hardwood Damage	85.3	72.9	0.0	100.0
Predictor Variables				
1987 Crown Damage (Silver Fire)	16.4	28.3	0.0	100.0
1987 Large Conifer Cover	29.0	36.6	0.0	100.0
1987 Small Conifer Cover	1.2	6.1	0.0	82.8
1987 Mixed-size Conifer Cover	2.2	9.2	0.0	82.2
1987 Hardwood Cover	6.2	15.8	0.0	77.2
1987 Shrub Cover	0.0	2.1	0.0	52.0
1987 Bare/Grass Cover	0.0	1.9	0.0	44.0
1987 Shannon Diversity	0.6	0.5	0.0	1.0
Elevation (m)	700.0	701.9	136.8	1476.0
Topographic Position (Fine)	-0.9	-1.1	-58.5	53.8
Topographic Position (Coarse)	-6.1	-2.5	-271.0	275.6
Slope (%)	57.6	57.0	21.7	92.4
Beer's Aspect	0.1	0.1	-1.0	1.0
Temperature (C)	27.2	25.6	16.6	35.0
Relative Humidity (%)	28.3	29.3	11.4	53.6
Wind Speed (km/hr)	15.0	14.1	6.2	19.1
Wind Direction (cosine transformed)	0.6	0.5	-0.3	0.8
Burn Period	A(9%), B(49%), C(42%)			

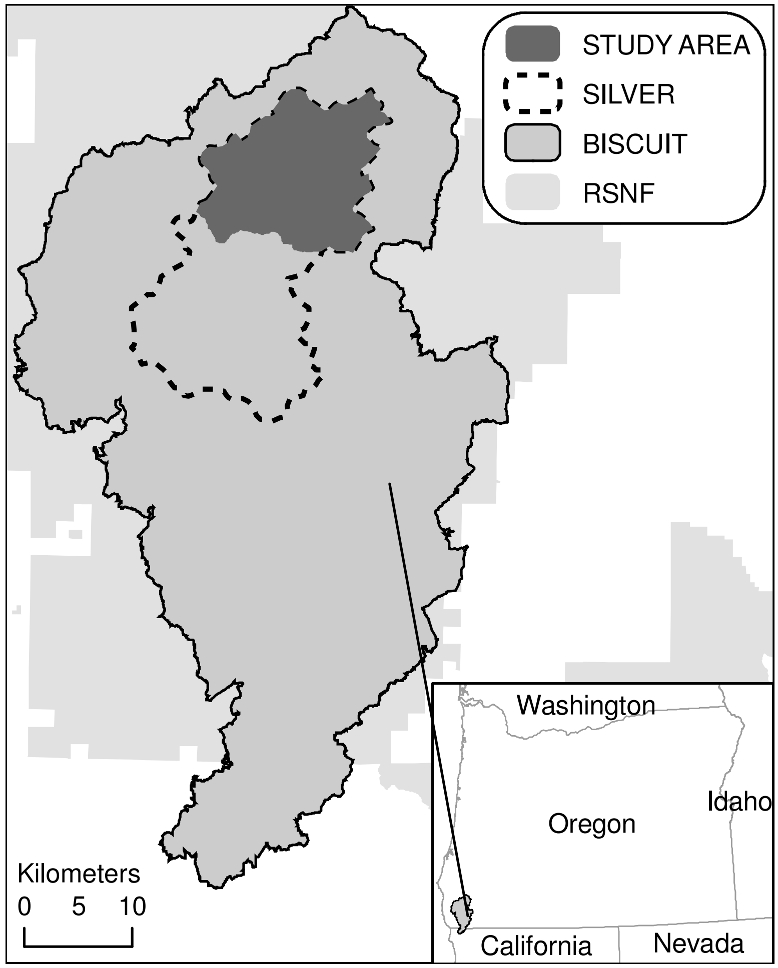

Figure 3.1. Map of the Study Area in context of the Silver and Biscuit Fires. (RSNF = Rouge Siskiyou National Forest)

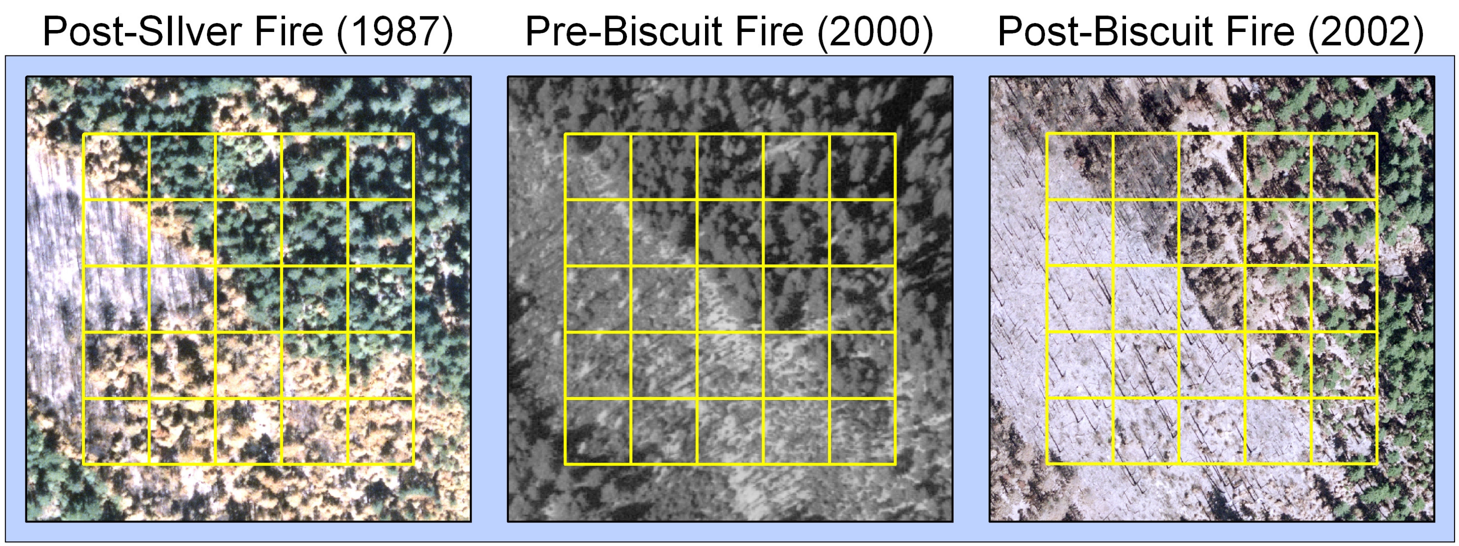

Figure 3.2. Example temporal sequence of aerial photo plots Plots are 6.25 ha; Cells are 0.25 ha (50m by 50m)

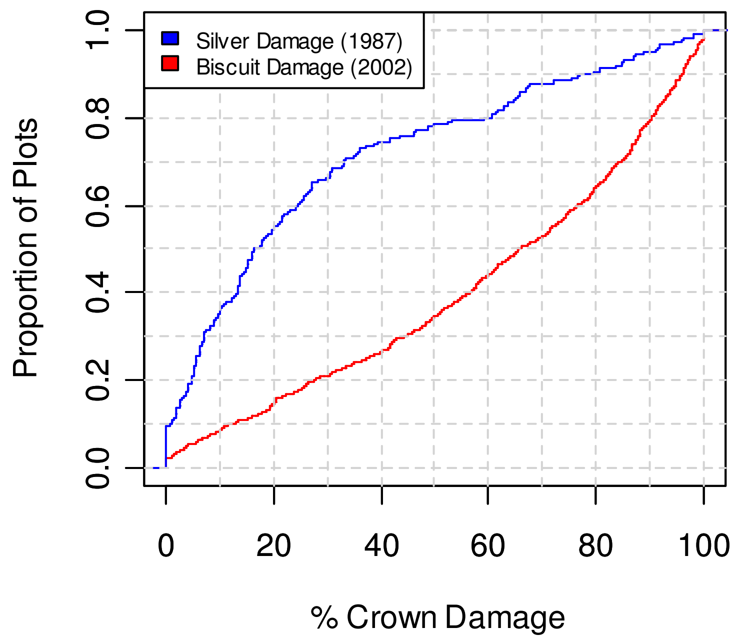

Figure 3.3. Empirical cumulative distribution of crown damage in the 1987 Silver Fire and 2002 Biscuit Fire.

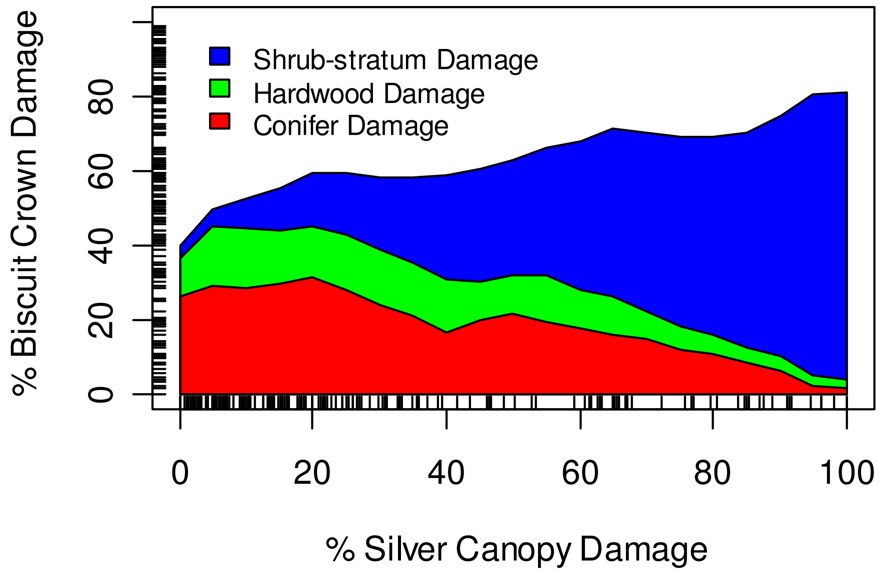

Figure 3.4. Stacked area graph showing absolute 2002 Biscuit Fire damage to shrub-stratum, hardwood, and conifer cover at five percent intervals of absolute Silver Fire crown damage. Rug plots along each axis correspond to the distribution of crown damage among photo plots for each fire. (Note that the crown damage depicted in this figure shows the absolute change in pre- to post fire cover and differs from the relative measures of crown damage used in the RFA and RTA models, see text for details)

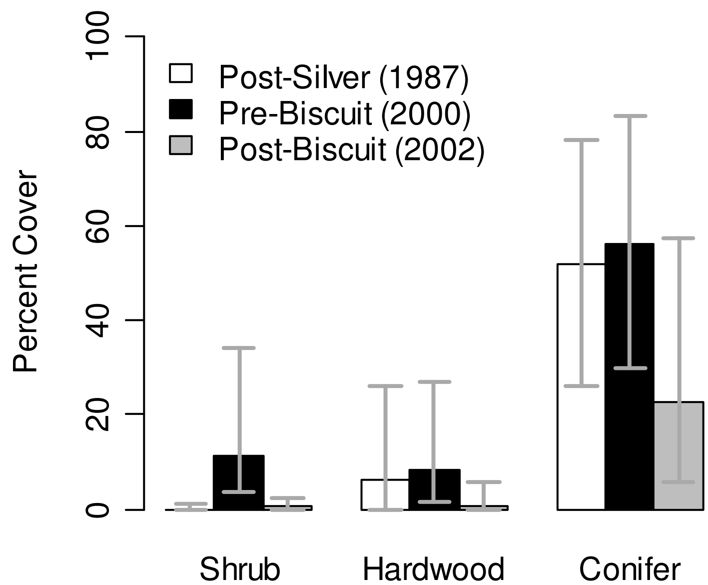

Figure 3.5. Median percent cover of shrub-stratum, hardwood, and conifer cover in 1987, immediately after the Silver Fire, in 2000, two years before the Biscuit Fire, and in 2002, immediately after the Biscuit Fire. Interval bars depict inner quartile range (25^{th} & 75^{th} percentiles).

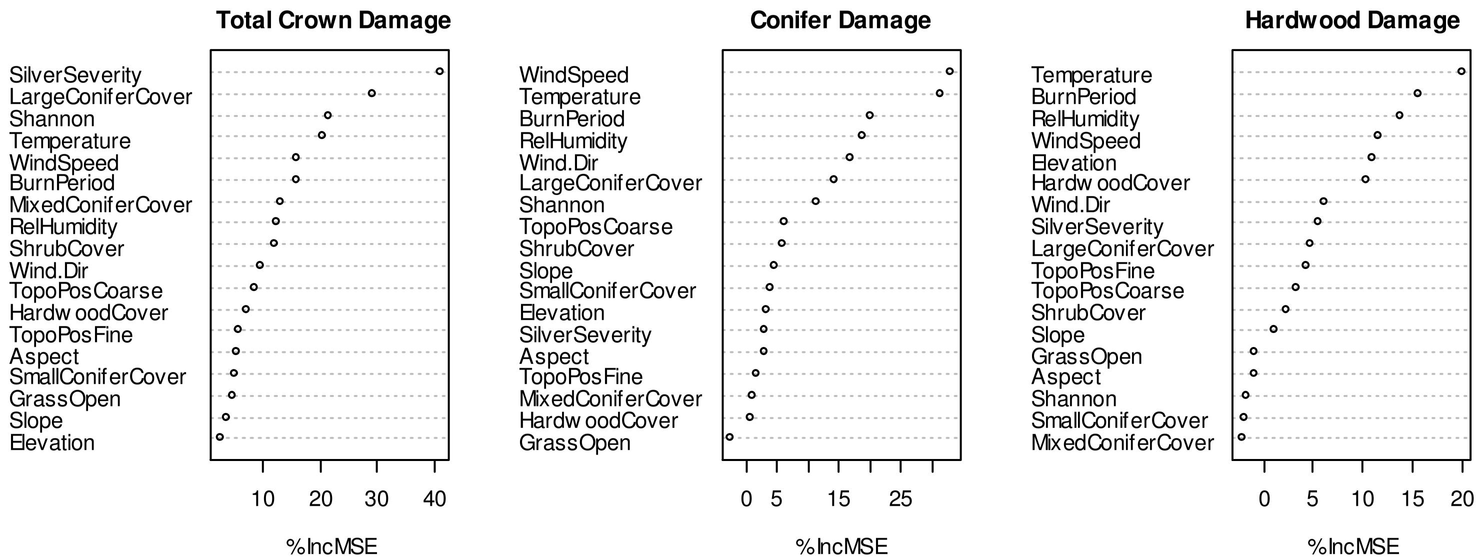

Figure 3.6. Variable importance plots for predictor variables from random forests models for total crown damage, conifer damage, and hardwood Damage. See text for details. Predictor variables are along the y-axis and the average increase in the mean square error when data for that variable are permuted and all other are left unchanged is on the x-axis.

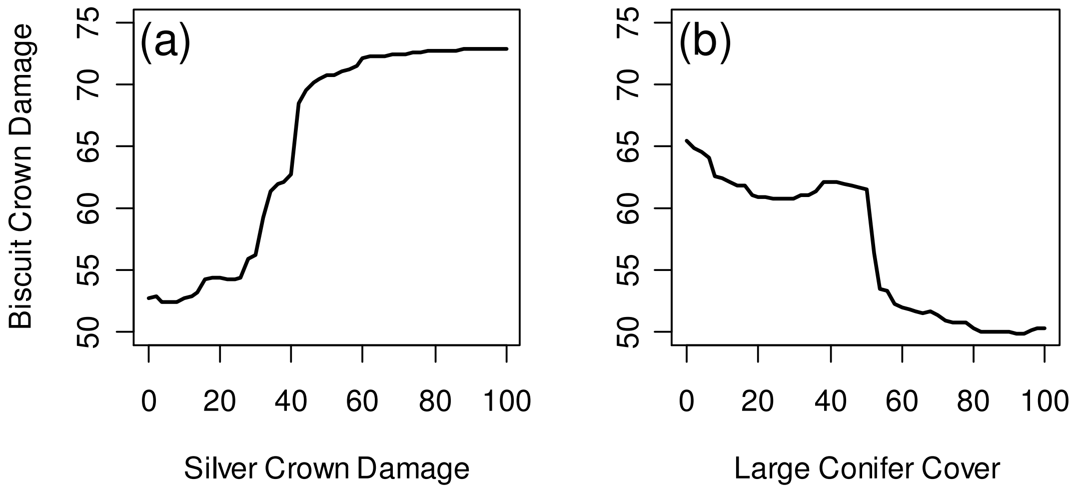

Figure 3.7. Partial dependence plots for random forests predictions of total damage on (a) silver fire severity and (b) large conifer cover based on random forest. Partial dependence is the predicted value of the response based on the value of one predictor variable after averaging out the effects of the other predictor variables in the model.

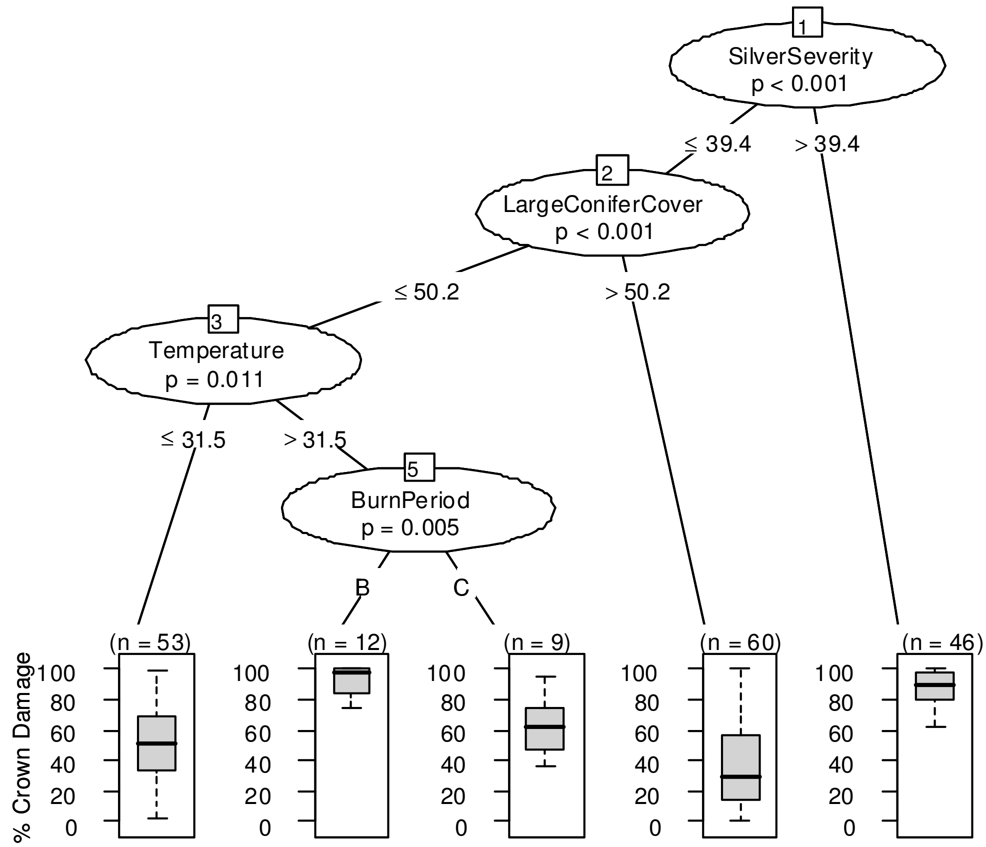

Figure 3.8. Regression tree for total crown damage based on the top six predictor variables from the random forest analysis (see figure 3.6). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range.

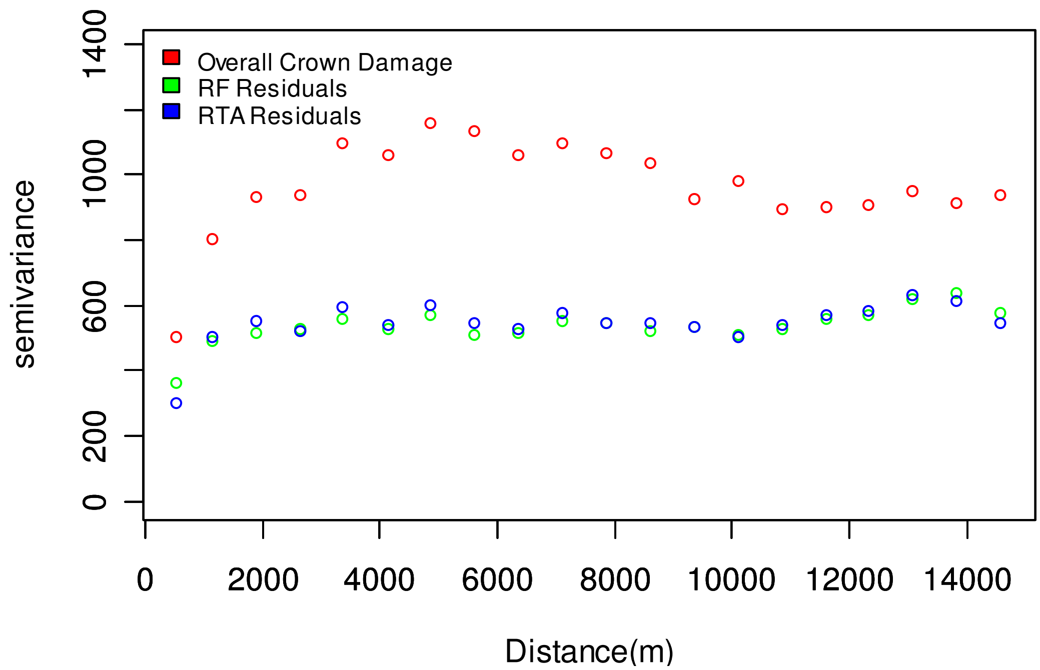

Figure 3.9. Semivariogram showing spatial autocorrelation in total crown damage data and the residuals from the regression tree analysis and random forest analysis for predicting total damage.

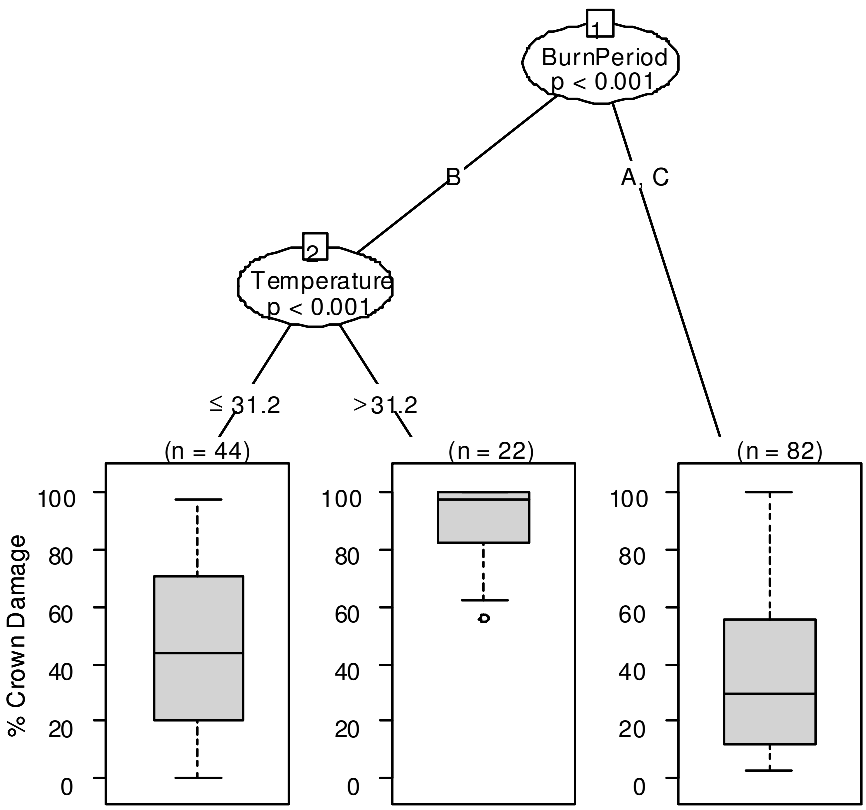

Figure 3.10. Regression tree of conifer damage using the top six predictor variables from the random forest analysis (see figure 3.6). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range. Dots represent more extreme data points

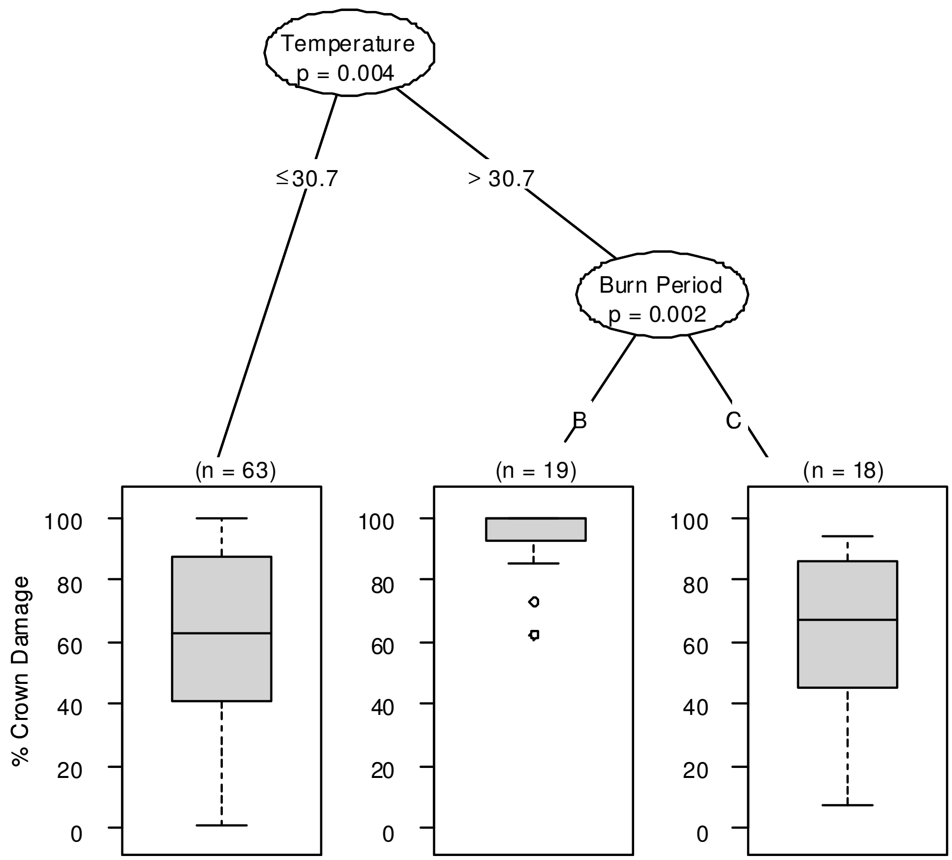

Figure 3.11. Regression tree of hardwood damage using the top six predictor variables from the random forest analysis (see figure 3.6). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range. Dots represent more extreme data points

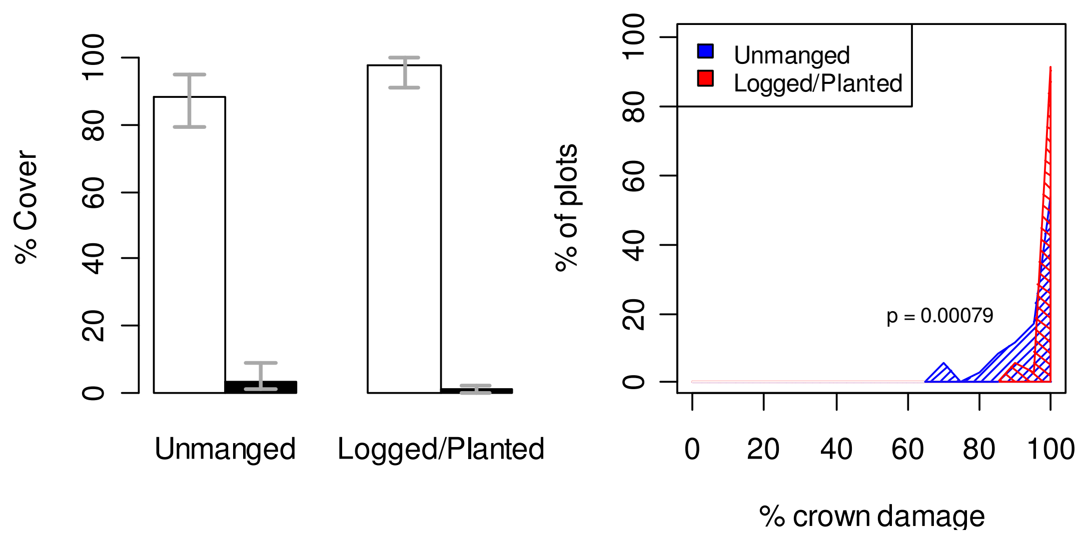

Figure 3.12. Comparison of damage in severely burned plots that were logging and planted and left unmanaged. Left: White bars are pre-fire cover and black bars are post fire cover; interval bars are inner quartile range (25th and 75th percentile). Right: distribution of relative damage as a percent of pre-fire cover. P-value is from a Monte Carlo randomization test for the difference in the median damage

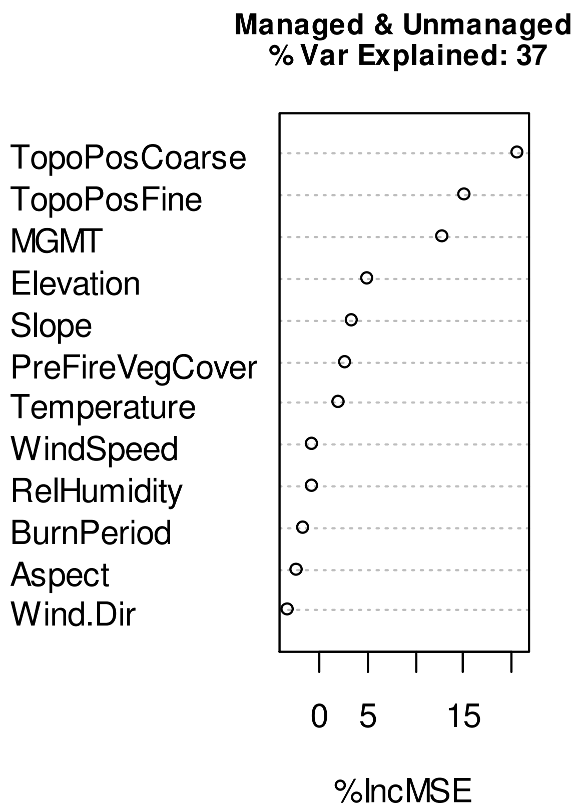

Figure 3.13. Variable importance plot from the random forests model of crown damage within areas that burned severely in the 1987 Silver Fire then were salvage logged and planted with conifers or were left unmanaged.

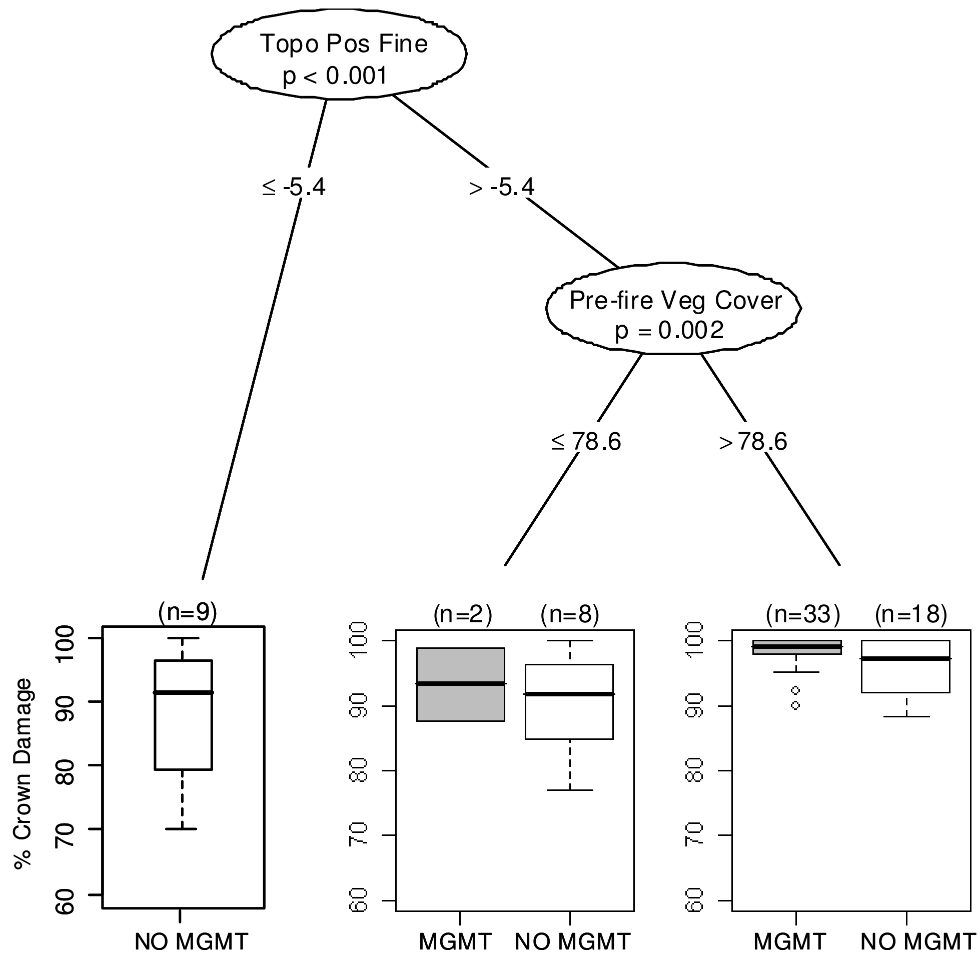

Figure 3.14. Regression tree of canopy damage within areas that burned severely in the 1987 Silver Fire then were salvage logged and planted with conifers or were left unmanaged. Boxplots were drawn independently for managed and unmanaged plots for illustrative purposes; they were not partitioned within the regression tree analysis.

CHAPTER 4: PATTERNS OF CROWN DAMAGE WITHIN A LARGE FIRE IN THE KLAMATH-SISKIYOU BIOREGION

ABSTRACT

The 2002 Biscuit Fire burned through more than 200,000 ha of mixed-conifer/ evergreen hardwood forests in southwestern Oregon and northwestern California. The remarkable size of the fire and the diversity of conditions through which it burned provided an opportunity to analyze the correlates of burn severity and describe the weather, topographical and vegetation conditions that produced the mosaic of crown damage. I measured pre- and post-fire vegetation cover on 761 digital aerial photo-plots (6.25 ha) within unmanaged forests and 198 photo-plots (1.25-6.25 ha) within conifer plantations. Ninety-seven percent of plots experienced some level of crown damage, but only 10% experienced complete crown damage. The median level of crown damage on unmanaged plots was 74%. The most important predictors of total crown damage were the percentage of pre-fire shrub-stratum vegetation cover and average daily temperature. The most important predictors of conifer damage were average daily temperature and "burn period," an index of fire weather fire and fire suppression effort. Increasing levels of shrub-stratum cover were associated with increasing levels of conifer damage and hardwood damage. Large conifers had 32% median crown damage while small conifers had 62% median crown damage. Patterns of damage were similar within the area that burned previously in the 1987 Silver Fire and edaphically similar areas without a recently history of fire. Owing largely to widespread shrub-stratum cover, low-productivity ultramafic soils had 92% median

crown damage compared to 59% on non-ultramafic sites. Median crown damage in conifer plantations was 89% and plantation age was, by far, the most important predictor of the level of damage. Plantations under 20 years old experienced the highest rates of damage.

INTRODUCTION

The 2002 Biscuit Fire was the largest contiguous wildfire in Oregon's recorded history. Encompassing more than 200,000 ha, it was the nation's largest forest fire in a year that experienced twice the national ten-year average annual area burned (NIFC 2008). Large infrequent disturbances (e.g. the Biscuit Fire, the 1980 eruption of Mount St. Helens, the 1988 Yellowstone Fires), offer rare opportunities to study events that will have a profound influence on ecosystem structure and composition for centuries (Foster et al. 1998), and potentially have ecological implications that are qualitatively different from those arising from smaller more frequent disturbances (Romme et al. 1998). In this respect, understanding the drivers and ecological effects of the Biscuit Fire are important research objectives in their own right. Additionally, although the Biscuit Fire was a singular event, it is also representative of a trend toward larger wildfires that is expected to continue coincident with climate warming (Westerling et al. 2006). The size of the Biscuit Fire and the diversity of conditions through which it burned presented an opportunity to address several questions regarding the pattern of vegetation damage (i.e. burn severity, sensu Agee (1993)) in relationship to its fire environment (weather, topography, and fuel) and management history.

The occurrence and behavior of large wildfires is a product of several interacting factors operating on multiple scales. Over broad spatial and temporal scales, the incidence of large fires is linked to climatic fluctuations (Hessl et al. 2004, Schoennagel et al. 2004, Westerling et al. 2006). At the scale of an individual event, large fires require contiguous and extensive fuels in addition to conducive fire weather, principally dry winds (Turner et al. 1994, Agee 1997, Turner et al. 1997). Wildfire behavior and severity, whether the fire is small or large, is highly stochastic and driven by complex interactions between weather, topography, and fuels (Agee 1993, Finney 2005). Fire modeling suggests that in some forest ecosystems extreme fire weather (i.e. high temperatures, low humidity, and high wind speeds) can override the influence of fuel and topography (Bessie and Johnson 1995). However, some empirical evidence suggests that forest conditions and topography can be important determinants of burn severity, even during periods of extreme drought and fire weather (Bigler et al. 2005, Finney et al. 2005). Topography can interact with weather to affect fire behavior by altering wind speed and direction (Rothermel 1972) and by differentially affecting fuel moisture on topographical positions and aspects with greater solar radiation (Albini 1976, Kushla and Ripple 1997). Forest structure and composition vary with climate and topography and influence fuel characteristics, such as crown bulk density, crown base height, and fuel moisture and continuity, which, in turn, affect fire behavior and severity (Rothermel 1972, Agee 1993, Sandberg et al. 2001, Agee et al. 2002, Graham et al. 2004). Greater heterogeneity of weather, topography, or fuels can disrupt fire spread and, over large spatial scales, is thought to be associated with lower overall severity (Minnich 1983).

The Biscuit Fire burned through diverse mixed-conifer/evergreen hardwood forests within the Klamath-Siskiyou bioregion. Before the era of effective fire suppression and timber harvest, circa 1940, fires were the most pervasive disturbance in the region. The composition and structure of the region's forest have been strongly influenced by a mixed-severity fire regime characterized by variable fire frequencies and heterogeneous effects within and between fires (Agee 1991, Wills and Stuart 1994, Taylor and Skinner 1998, Sensenig 2002, Taylor and Skinner 2003). The mixed-severity fire regime is an important component of temperate forests worldwide (Morgan et al. 2001, Fule et al. 2003, Schoennagel et al. 2004, Lentile et al. 2006b, Baker et al. 2007, Hessburg et al. 2007). Yet despite its prevalence, it is the least studied fire regime (Agee 2005), and the relative importance of weather, topography, and fuel for determining fire behavior in these systems is poorly understood (Schoennagel et al. 2004). Identifying the characteristics of the fire environment that were associated with patterns of Biscuit Fire severity is important for anticipating future fire effects and for prioritizing management in the post-fire landscape.

Of the three components of the fire environment, weather is the most difficult to relate to patterns of fire damage, post hoc. By consuming oxygen and releasing great quantities of heat, the chemical process of combustion circulates air and creates its own weather that is site-specific and impossible to precisely reconstruct (Pyne et al. 1996). Nonetheless, based on the anomalous size and rate of spread of the Biscuit Fire, I hypothesized that daily, regional weather conditions (temperature, humidity, wind speed, and wind direction) were strongly associated with patterns of fire severity. Other attempts to relate regional weather conditions to fire severity have found daily average temperature and wind speed to be among the most important correlates (Collins et al. 2007).

With regard to topography, I expected the Biscuit Fire to have followed similar patterns to those found elsewhere in the region. I hypothesized higher severity on southwesterly aspects due to lower fuel moisture and presence of smaller, more vulnerable vegetation (Weatherspoon and Skinner 1995, Taylor and Skinner 1998, Alexander et al. 2006). The relationship with elevation was more difficult to anticipate. Although higher elevations have been associated with lower severity in some fires, presumably due to lower temperature and higher humidity (Weatherspoon and Skinner 1995, Alexander et al. 2006), this pattern can be reversed when vegetation found at higher elevations is more susceptible to fire damage either due to species composition (Broncano and Retana 2004) or greater fuel accumulation associated with lower fire frequency (Agee 1991). Dendrochronological research in the Klamath Mountains suggests that upper topographical positions were historically associated with more severe fires, owing to lower fuel moisture and pre-heating of fuels (Taylor and Skinner 1998). I expected, therefore, that the Biscuit Fire had produced similar patterns.

Because fuel is the only component of the fire environment that can be managed, understanding its relationship to fire severity is of particular importance to land owners and forest managers throughout the West (Graham et al. 2004). The relationship between Biscuit Fire severity and pre-fire vegetation—particularly where large conifers are involved—is especially important to federal land managers, who have been charged with maintaining and promoting development of late-successional forest ecosystems (FEMAT 1993, USDA-USDI 1994). The Biscuit Fire burned through a tremendous diversity of vegetation conditions, and although it was just a single disturbance event, observations of fuel related differences in patterns of severity can reveal important information about fire behavior.

The Klamath-Siskiyou bioregion contains the most diverse flora in western North America (Whittaker 1960), much of which is characterized by a Douglas-fir (*Pseudotsuga menziesii*) dominated overstory and an evergreen hardwood subcanopy (primarily tanoak (*Lithocarpus densiflorus*) (Franklin and Dyrness 1988)). There has been little research that has explicitly considered the influence of evergreen hardwoods on fire severity in these mixed-evergreen forests. Raymond and Peterson (2005) reported high rates overstory mortality of tanoak within field plots in the Biscuit Fire, which is consistent with expectations given their low crown base heights, relatively thin bark, and flammable leaves (Brown and Smith 2000). In contrast, most conifers in the region (e.g. Douglas-fir and sugar pine (*Pinus lambertiana*)) have adaptations to fire such as thick bark and high canopies that make them increasingly resistant to fire as they age (Agee 1993). Several studies have found lower rates of damage in large conifers as opposed to small conifers (Alexander et al. 2006, Lentile et al. 2006). I therefore hypothesized that damage would have been lower in conifer cover than in hardwood cover and that large conifers had the lowest levels of damage. I further hypothesized that rates of conifer damage would have been higher in places with a hardwood sub-canopy, due to increased vertical fuel continuity (ladder fuels). This may have also occurred in places where large conifers were mixed with small conifers, a situation that is prevalent where fire suppression has led to increased conifer densities (Taylor and Skinner 1998). Alternatively, stands with multi-layered canopies may have cooler microclimates and contain large conifers that are especially resistant to damage.

The effect of fire suppression on fire hazard within the region's forests has not been studied directly (fire hazard is the influence of fuel on fire behavior independent of weather, sensu Hardy (2005)). In much of the western U.S., suppression is believed to have increased fuel loads and continuity, which has prompted calls for widespread landscape restoration (Stephens and Ruth 2005). The appropriateness of this strategy for the Klamath-Siskiyou is unclear. Higher forest productivity within region compared to most fire-prone forests (Waring et al. 2006) is incongruent with many of the assumptions underlying the idealized low-severity restoration model (e.g. Covington 2000). Nonetheless, several studies have shown that fire frequency has decreased from historical levels (Agee 1991, Taylor and Skinner 1998, Sensenig 2002, Taylor and Skinner 2003), which has coincided with increased forest density and decreased spatial complexity (Skinner 1995, Taylor and Skinner 1998). This suggests that fire hazard has increased due to suppression. However, in a study of several recent fires in region, Odion et al. (2004) found lower proportions of high severity fire where fires was previously absent since at least the 1920s, suggesting that fire suppression may have decrease fire hazard. A national effort to quantify the departure from historical conditions resulting from fire suppression (i.e. condition class) classified the Biscuit Fire region as a mix of low to high departure (LANDFIRE 2007).

The Biscuit Fire provided a unique opportunity to compare a large area that recently burned at mixed-severities, to a large area without with a recent history of fire. In 1987, the Silver Fire burned through > 38,000 ha, which was subsequently re-burned by the Biscuit Fire. Within the reburn area there was a positive correlation between severity in 1987 and 2002 (Thompson et al. 2007, and see Chapter 3). Places where overstory crown damage was severe during the Silver Fire succeeded to shrubs and regenerating trees in the intervening years, which then experienced severe crown damage during the Biscuit Fire. Places with low levels of overstory crown damage during the Silver Fire were the least likely to experience overstory damage in the Biscuit Fire. Thompson et al. (2007) made no comparisons between the once and twice burned regions of the Biscuit.

The fifteen year interval between the Silver and Biscuit Fires is well within the estimated historical fire frequency for the region, which lengthens along an east to west and low to high elevational gradients and typically ranges from 5 to 100 years (Agee 1991, Atzet and Martin 1991, Sensenig 2002, Taylor and Skinner 2003). The complexity of the region's fire regime and the potential impact of fire suppression on fire severity gave rise to several plausible hypotheses regarding the effect of the Silver Fire on Biscuit Fire severity. Based on Thompson et al. (2007), one reasonable hypothesis was that, as a result of more abundant shrubs and regenerating trees, the twice burned landscape had higher levels of total crown damage. This hypothesis could extend beyond the stand-replacement patches if under burning during the Silver Fire initiated basal sprouting of sub-canopy hardwoods (Tappeiner et al. 1984), thereby increasing surface fuels and vertical continuity under the tree canopy. Odion et al. (2004) found lower rates of fire severity in the Klamath region in areas without a recent history of fire and speculated that increasing overstory shade reduced surface fuel accumulation and crown fire initiation. However, the Silver Fire experienced fairly low levels of tree-stratum crown damage (Chapter 3). Therefore, an alternate hypothesis was that the Silver Fire decreased surface fuels and vertical continuity within the bulk of the landscape that burned as a surface fire, thereby reducing fire severity. This hypothesis is consistent with the conventional wisdom that fire suppression across the region has increased forest density and fuel hazard (Sensenig 2002, Taylor and Skinner 2003). Finally, there may be no difference in tree crown damage between the two landscapes. Tree-stratum damage inside the twice burned landscape was largely independent of the level of damage during the Silver Fire (Chapter 3). Therefore, it is also plausible that patterns of damage were similar inside and outside the twice burned region.

Vegetation patterns, and thus fuel patterns, within the bioregion are heavily influenced by distinctive geologic formations, including the largest exposure of ultramafic soils in North America (Kruckeberg 1984). Approximately one-third of the Biscuit Fire burned over peridotite and serpentinite parent materials, high in nickel and magnesium. Soils formed in these areas are highly erosive and have chemical and physical properties that result in low plant productivity. Sparse vegetation on these unproductive soils is commonly juxtaposed by dense stands of conifers and evergreen hardwoods on highly productive metamorphic and igneous parent material (Whittaker 1960, 1961, Atzet et al. 1996). Although there have been few empirical studies, conventional wisdom suggests that ultramafic sites do not readily carry fire owing to their low canopy and surface fuel loads, and that these areas can act to inhibit the spread of low- and moderate-intensity surface fires (Atzet and Wheeler 1982, Kruckeberg 1984, Taylor and Skinner 2003). When fires do occur on ultramafic sites, they are typically low intensity surface fires (Whittaker 1960, Atzet and Martin 1991). Accordingly, I hypothesized that crown damage would be less severe on ultramafic sites.

The fuel environment encountered by the Biscuit was further affected by past forest management. The region has a long history of even-aged silvicultural practices, which have been an important part of the economy since World War II (Walstad 1992). Even-aged silviculture tends to produce forests with continuous canopies and high stem densities, which can make them vulnerable to fire (Stephens and Moghaddas 2005). As plantations age, self-pruning results in higher crown base heights (Hanus et al. 2000), which may increase their resistance to fire. Although several studies have investigated the effect of partial harvests on fire severity, (Graham et al. 1999, Pollet and Omi 2002, Graham et al. 2004, Raymond and Peterson 2005), there have been far fewer examinations of burn severity within even-aged plantations. The available empirical evidence suggests that plantations following clearcutting are susceptible to high severity fire (Weatherspoon and Skinner 1995, Odion et al. 2004). I am aware of no previous empirical studies that explicitly relate burn severity to the age of plantations. However, Graham et al. (2003) noted that plantations older than 12 years experienced lower severity than younger plantations during the 2002 Hayman Fire in Colorado and Thompson et al. (2007) found universally high burn severity within 12 to 15 year old plantations that were salvage-logged and planted after the Silver Fire then reburned in the Biscuit Fire. Fire modeling also suggests that plantations are more vulnerable than unmanaged stands from the time they are young saplings (< 5yrs) at least until they reach >50cm diameter at breast height (DBH; Stephens and Moghaddas 2005). I hypothesized that conifer plantations experienced high rates of damage but that damage decreased coincident with age.

To address these hypotheses, I used digital aerial photography to measure changes in pre- to post-fire vegetation cover in relation to weather, topography, edaphic setting, recent fire history, and management history. I measured levels crown damage within three cover types—conifer, hardwood, and low-stature vegetation (a mix of shrubs and small trees, hereafter called the shrub-stratum, sensu Sandberg et al. (2001)). Digital aerial photography offers the highest possible resolution for measuring past vegetation conditions over large areas. Satellite-based remote sensing platforms are more commonly used to measure fire effects on vegetation, particularly the Landsat-based differenced normalized burn ratio (dNBR; Key and Benson 2004).

dNBR correlates well with total vegetation damage (Appendix A; Miller and Yool 2002, van Wagtendonk et al. 2004), and it is commonly used for quantifying landscape-scale burn effects (e.g. Odion et al. 2004, Bigler et al. 2005, Finney et al. 2005, Collins et al. 2006, Kulakowski and Veblen 2007, Safford et al. 2007, Thompson et al. 2007, Wimberly and Reilly 2007). However, dNBR can not effectively distinguish between the type or structure of burned vegetation. At high levels of dNBR, changes in the index may be more associated with surface soil features (e.g. ash, soil color) than with canopy mortality, which reaches 100% before the maximum level of dNBR is reached. By using aerial photography to measure the percent of crown damage on a continuous scale, I was able to provide a clear metric of "severity," which is a concept that is often ambiguous and can lead to miscommunication (Jain 2004).

Using data from the photos combined with several other existing data layers I addressed the following five questions regarding the 2002 Biscuit Fire:

1. *What was the relative importance of weather, topography, and fuel for predicting patterns of crown damage?*
2. *Did patterns of crown damage differ between cover types and did the presence of some cover types affect damage in other cover types in close proximity?*
3. *Did the pattern of crown damage differ between areas with and without a recent history of fire?*

4. *Did the pattern of crown damage differ between ultramafic and non-ultramafic soils?*
5. *Did the pattern of crown damage differ between plantations and unmanaged forests and how did plantation age affect the level of damage?*

METHODS

Study Area:

The study area was the area burned by the 2002 Biscuit Fire, centered at 123°91'W latitude 42°29'N longitude (Fig. 4.1). The fire began as five separate lightning ignitions between July 13th and 15th, 2002 (GAO 2004), which combined to become the Biscuit Fire over the following weeks. By September 6th, the fire had been contained at its final size of 202,168 ha. Ninety four percent of the fire was on land managed by the Rouge Siskiyou National Forest (RSNF) in southwestern Oregon, 5% was on the Six Rivers National Forest in northwestern California, and 1% was on Bureau of Land Management land (Rogue Siskiyou National Forest 2004). The area includes range of management histories and designations, including the 73,000 ha Kalmiopsis Wilderness Area as well as extensive roaded and logged areas. Topography is steep and complex with no consistent directional trends. Elevation ranges from 50 to 1550 m. Sixty-eight percent of the burn area is underlain by igneous, meta-sedimentary, and metamorphic soil parent materials. Forests on these soils are dominated by conifer species such as Douglas-fir, sugar pine, white fir (*Abies concolor*), and knobcone pine (*P. attenuata*). Dominant evergreen hardwoods include tanoak, Pacific madrone (*Arbutus menziesii*), chinquapin (*Chrysolepis chrysophylla*), and canyon live-oak (*Quercus chrysolepis*). Manzanita (*Arctostaphylos sp*.), ceanothus (*Ceanothus sp*.), and Sadler oak (*Q. sadleriana*) are common shrubs. In older stands, sclerophyllous broad-leaved trees often form lower strata under the conifer overstory (Franklin and Dyrness 1988). The remaining 32% of the study area is underlain by ultramafic parent material with low plant productivity and a distinct flora dominated by conifer species such as Jeffery pine (*P. jeffreyi*), Douglas-fir, sugar pine, and incense-cedar (*Calocedrus decurrens*). Common shrubs include huckleberry oak (*Q. vaccinifolia*), manzanita, and ceanothus. The climate is Mediterranean, with dry, warm summers and wet, mild winters. Mean January temperature is 6ºC. Mean July temperature is 16ºC. Mean annual precipitation is 290 cm, with greater than 90% occurring in winter (Daly et al. 2002).

Fires were common throughout pre-Euro American history with a large portion of them burning with low to mixed severity (Atzet and Martin 1991). Estimated historical return intervals lengthen along an east to west and low to high elevational gradient and typically range between 5 and 100 years (Agee 1991, Atzet and Martin 1991, Sensenig 2002). Though there has been little empirical research, return intervals are assumed to be longer on ultramafic sites where fuel can be a limiting factor (Atzet and Wheeler 1982, Atzet and Martin 1991). Fire was common throughout mining era of the latter nineteenth and early twentieth centuries. In 1943, a smoke jumper base was established on the eastern edge of the RSNF; since then, fire has been effectively suppressed, with punctuated exceptions in 1987 and 2002 (Atzet and Martin 1991, Rogue Siskiyou National Forest 2004). The 1987 Silver Fire encompassed > 38,000 ha of forests on largely non-ultramafic soil, which was entirely reburned by the Biscuit Fire (Fig. 4.1). Crown damage during the Silver Fire was below 50% within 75% of the burn area (see chapters 2 and 3 for more details on the Silver/Biscuit reburn area).

Image processing and interpretation:

I interpreted pre- and post-fire aerial photos distributed across approximately 85% of the Biscuit Fire (Fig. 4.1)—the remaining 15% was not included in the post-fire photo coverage (the reason for this omission has never been adequately explained). The best available pre-Biscuit photos (August 2000) were digital orthoquads (DOQs) taken as part of the USDA National Agriculture Imagery Program (http://165.221.201.14/NAIP.html). They were panchromatic with a 1 m grain size, and they dictated the resolution of vegetation data that I could reliably interpret. The post-Biscuit Fire photos were taken on September 24, 2002; they were scanned directly from diapositives (20 microns (1200 dpi)). I interpreted vegetation condition and fire effects within 761 randomly located photo-plots and 198 management units randomly selected from a database acquired from the RSNF (management data are described below). I constructed the unmanaged photo-plots as five-by-five polygon grids of 50 m cells within a geographic information system (GIS). Each plot covered 6.25 ha (Fig. 4.2), which could be overlain onto georeferenced aerial photos and other GIS layers. Unmanaged plots were distributed such that no portion of a road, management unit or large stream or river was included. To construct the management plots, a polygon grid of 50 m cells was overlain onto the variably shaped management units (Fig. 4.2). If the unit was larger than 6.25 ha, then 25 of the cells were randomly selected to be used in the plot. If management units were smaller than 6.25 ha, then all cells were used. Management units smaller than 1.25 ha were excluded.

Photo were spatially co-registered using approximately fifteen ground control points (GCP) to link the 2002 photos to the 2000 DOQs within ERDAS Imagine 8.7. GCPs were concentrated in and around the plot, while the remainder of the photo was ignored and later clipped out of the image. Individual trees were the most common tie-points, but rock outcroppings and other topographic features were also utilized. After the GCPs were placed, a first-order polynomial transformation was used to geo-rectify the photo. The resulting grain size for 2002 photos was ~ 0.30 m.

To conduct photo interpretation, I overlaid the GIS plot-grids onto the geo-referenced aerial photos in ESRI ArcMap 9.2. I interpreted vegetation conditions at both points in time. Percent cover of conifer, hardwood, shrub-stature vegetation, and bare ground/grass were estimated for every cell in every plot. (Note that shrub-stature vegetation is a measure of all low stature vegetation including young conifers and hardwood, not simply shrub species.) Areas of topographic shadow were subtracted from the effective area of the plot. Pre-fire conifer cover in each cell was further assigned a size class: small, large, or mixed. The small class roughly corresponded to pole or small saw timber (< 20 cm DBH) and the large class corresponded to large saw timber (> 20cm DBH; Paine and Kiser 2003). (DBH estimates were verified with a post hoc comparison of conifers in photo photos to 70 co-located Forest Service inventory plots. Inner quartile range (IQR) for DBH on small plots = 7 to 19 cm; on large plots IQR = 18 to 42 cm; on mixed plots IQR = 10 to 23cm). Percent cover of fire related crown damage (scorch or consume) was estimated from the 2002 photos. Crown damage, in addition to being a direct measure of fire effects on forest structure, is also closely tied to fireline intensity and is a useful measure of a fire's effect on an ecosystem (Van Wagner 1973, Weatherspoon and Skinner 1995). Cover estimates summed to 100% in each cell. Cell-level cover estimates were averaged to obtain plot-level values. To establish a metric of structural complexity for each plot, Shannon's diversity (Hill 1973) was calculated at the plot level using the seven cover classes just described. Because it was impossible to reliably discern grass from bare ground in the 2000 photos, and because bare ground doesn't burn, I included a constraint that forced the percent bare/grass in be equal in 2002 and 2000. All photo interpretation was conducted by a single researcher (Thompson) to ensure consistency and reduce error. At the onset of the research, I created a catalog of paired oblique-to-aerial photos for use as a training manual. Later, I ground truthed a subset of photo-plots, which revealed excellent correspondence between field and photo measurements.

Topographic, climate, weather, and management data

Using a 10-meter digital elevation model (DEM), I calculated average elevation, percent slope, aspect, and topographic position for each photo-plot. I used Beer's transformed aspect (Beers et al. 1966) for all plots, which varies from -1 on NE facing slopes with little incident sunlight to 1 on SW facing plots receiving abundant incident light. I calculated an index of topographic position (TP) at two spatial scales. TP-Fine is the absolute difference in elevation between each pixel in the DEM and the average elevation in an annulus 150 to 300 meters from the focal pixel. TP-Coarse uses an annulus 850 to 1000 meters from the focal pixel. I identified plots on ultramafic soil parent material using the USDA Soil Survey Geographic (SSURGO) Database (USDA 2008). To capture regional gradients in productivity associated with moisture availability, I assigned the local annual precipitation for the climatological period spanning 1971 to 2000 to each plot based on the PRISM model, which uses point data combined with elevation and climate averages to generate gridded estimates of annual precipitation at an 800 m grain size (Daly et al. 2002).

The RSNF provided a map showing the daily progression of the fire, which I used to assign weather data to each photo-plot at a daily resolution. This technique, though temporally and spatially coarse, has been successfully used in other reconstructions of fire effects (Collins et al. 2006). Average daily values for temperature, relative humidity, wind speed, and wind direction between 10:00 and 17:00 for each day of the burn were calculated from four remote automated weather stations (RAWS) that surround the burn (Table 4.1; Fig. 4.1; (RAWS data accessed from http://www.wrcc.dri.edu/fpa/)). Before averaging, I cosine transformed wind direction such that a value of -1 corresponded to winds out of the southwest and a value of +1 corresponded to winds out of the northeast, which are typically drier and associated with severe fire weather in this region (USDA Forest Service 2002). The RAWS span large geographical and elevational gradients and, although data were moderately correlated between stations ($r \approx 0.3$- 0.8), each provided some unique information (Table 4.1). I explored several methods for interpolating weather data to the plots, including assigning each station's data individually and more complicated methods based on elevational similarity and Euclidian proximity between RAWS and photo-plots. I found that calculating weather estimates for each plot using inverse distance weighting of the RAWS data explained the most variability in crown damage. Therefore, I assigned a weighted average of each weather variable to each plot, where the weights were based on the inverse of the squared Euclidian distance between each plot and each RAWS station.

I also created a variable that divided the burn area into three "burn periods," which correspond to the spread of the Biscuit Fire and fire suppression effort during each period (Fig. 4.1 USDA Forest Service 2002, GAO 2004). Period A, represents 5% of the total Biscuit Fire area and includes the region that burned from July 13 to July 26 with comparatively little suppression effort and mild weather conditions (Table 4.2). Period B includes the region that burn from July 27 to August 4. Fifty percent of the Biscuit Fire burned in this nine day period, which was characterized by strong north-northeasterly winds and comparatively low relative humidity. Suppression resources increased during this period but were largely unsuccessful in preventing fire spread. Period C represents the remaining 45% of the Biscuit Fire that burned from Aug 5 to Sept 06, when the fire was effectively contained. Fire suppression activities were extensive throughout Period C. The fire continued to spread during extreme weather but had a higher potential to be influenced by fire fighters, including extensive but undocumented controlled burns.

I used management data from a RSNF geo-database that described historical logging and planting on more than 650 stands (8300 ha) within the burn perimeter. I randomly selected 200 management units with the constraints that each unit had been clearcut between 1960 and 1996 and had some record of planting. Of these, 35 were salvage-harvests completed between 1988 and 1991 following the 1987 Silver Fire. Two units were later excluded because their positions were inaccurate within the GIS layer. Management records were often incomplete regarding species and volume removed, site preparation, and planting density and success. Existing records indicate that some live trees were left after harvests and that planting consisted primarily of Douglas-fir with a lesser component of ponderosa and sugar-pine. Site preparation after harvest was quite variable and records were incomplete. Multiple planting dates, all clustered within one to three years of harvest, were often associated with individual management units. I therefore used the date of harvest as a surrogate for the establishment date, unless there was evidence that original planting had failed and the site had been reforested at a later date. Harvest dates were seen as reliable by RSNF personnel (pers. comm. J. Hawkins, Gold Beach Ranger District, RSNF, September 2005).

Data Analysis

To summarize fire effects on vegetation, I calculated the median plot-scale percent cover of conifer, hardwood, and shrub-stratum vegetation before and after the Biscuit Fire (2000 and 2002). I plotted empirical cumulative distributions of total canopy damage, conifer damage, hardwood damage, and shrub-stratum damage at both the cell (0.25 ha) and plot (6.25 ha) scale to illustrate differences in fire effects at two spatial resolutions. I calculated summary statistics for differences in vegetation cover between ultra-mafic and non-ultramafic sites and used Monte Carlo tests (Gotelli and Ellison 2004) with 10,000 randomizations to compute a p-value describing the probability of encountering differences in median levels of crown damage that are at least as large as those observed, given the distribution of data. Similarly, I used Monte Carlo randomization tests to compare proportional crown damage in areas that had burned during the 1987 Silver Fire and edaphically similar areas (non-ultramafic) without any recent history of fire.

To examine the relationship between crown damage in the unmanaged plots and the suite of predictor variables (Table 4.3), I structured four response variables that describe different aspects of canopy damage: total canopy damage, relative conifer damage (i.e. ((2000 Conifer Cover – 2002 Conifer Cover) / 2000 Conifer Cover)*100))), relative hardwood damage, and burn variability (Table 4.3). I examined relative measures of conifer and hardwood damage because absolute measures of damage were roughly proportional to availability (e.g. hardwood damage was highest where hardwood cover was highest). Also, by modeling relative damage, I was able illustrate the relationship between fuel, weather, and topographical setting and damage done to the vegetation cover available to the fire. When modeling relative canopy damage of conifers and hardwoods, I subset the data to include only those plots with >5% pre-fire cover of that cover-type. I used the standard deviation of the percent crown damage among the 25 cells within each plot as my measure of plot-level burn variability. I included this response variable because I wanted to identify the conditions that led to high levels of burn heterogeneity (i.e. mixed-severity), which may, in turn, influence future succession and disturbance pathways. Only plots that had some evidence of fire were included in this portion of the analysis.

I analyzed the logged and planted stands (hereafter: plantations) independently from the unmanaged stands. Many plantations were young and contained only shrub-stratum vegetation. Rather than make a distinction between conifer cover in older plantations and shrub-stratum cover in younger plantations, I interpreted the percent cover of total woody vegetation versus cover of bare/grass, and used the age of the unit as a proxy for vegetation size and structure. Percent damage to available crowns was my response variable. There were no plantations on ultramafic sites and it was not included as a potential predictor variable, nor was Shannon's diversity index. Eighty-five percent of the managed stands burned during period C, so it was not included as a predictor variable. The size of each plantation was added pool of potential predictors (Table 4.4).

I took a two-stage approach to analyzing relationships between response and the suite predictor variables (Table 4.3; Table 4.4). I first used random forest analysis (RFA; Breiman 2001) to estimate and rank the importance of predictors and then used regression tree analysis (RTA; Breiman et al. 1984, De'ath and Fabricius 2000) to illustrate the nature of relationships between the response and important predictor variables. These nonparametric methods are ideally suited for the analysis of high dimensional ecological data with hierarchical, complex, and non-linear relationships to response variables (De'ath and Fabricius 2000, Cutler et al. 2007). I are aware of no previous application of RFA to fire effects data, but RTA (and its analytical cousin, classification tree analysis) has been used extensively to model relationships between fire severity and environmental predictor variables (e.g. Finney et al. 2005, Alexander et al. 2006, Collins et al. 2006, Lentile et al. 2006b, Jain and Graham 2007, Kulakowski and Veblen 2007).

RFA is an ensemble learning algorithm (ELA) that averages predictions over multiple bootstrapped regression trees. (An ELA is any method that runs a base algorithm multiple times to construct a set of hypotheses, called an ensemble, which is then used to "vote" to predict the value of new data.) I used Liaw and Wiener's (2002) implementation of RFA within the R statistical language (R Development Core Team 2006); the algorithm, as applied to my data, was as follows: (1) Select 1500 bootstrap samples with replacement, each containing 63% of the data. (2) For each bootstrap sample, grow an un-pruned regression tree with the modification that at each node, rather than implementing the best split among all predictors, randomly select one-third of the predictor variables (six, in my case) and choose the best split from among those variables. (By selecting from a subset of predictors, RFA forces diversity among regression trees, which reduces bias in variable selection and reduces variance in the averaged prediction. Selecting among approximately one-third of the total number of predictors was suggested by Breiman (2001), and was found to be optimal for these data). (3) At each bootstrap iteration, predict the response value for data not included in the bootstrap sample—the so-called Out-Of-Bag or OOB data—and average those response values over all trees. (Because the OOB data are not used when building the trees, their estimates are essentially cross-validated accuracy estimates.) (4) Calculate importance values for each predictor by calculating the percent increase in mean squared error (MSE) when OOB data for each variable are permuted while all others are left unchanged. The MSE is computed as:

$$MSE_{OOB} = n^{-1} \sum_{1}^{n} \left\{ y_i - \hat{y}_i^{OOB} \right\}^2$$

where $\hat{y}_i^{OOB}$ is the average of the OOB predictions for the *i*th observation. The "percent variance explained" is computed as:

$$1 - \frac{MSE_{OOB}}{\hat{\sigma}_y^2}$$

RFA is increasingly being used to analyze ecological data and to select important predictor variables where many possibilities exist. Where it has been tested, RFA has consistently out-performed other statistical methods, including RTA, for prediction accuracy (Garzon et al. 2006, Lawler et al. 2006, Prasad et al. 2006, Cutler et al. 2007, Peters et al. 2007). However, RFA has been termed a "grey box" model because it lacks full transparency (Prasad et al. 2006). There is no simple representation of the relationship between response and predictor variables; one cannot reasonably examine all the individual regression trees, nor are there p-values, regression coefficients, or confidence intervals that accompany many traditional statistical techniques. I used variable importance values derived from RFA to rank the predictors in terms of the strength of their relationship to the response. I also used partial dependence plots to show the effect of changing individual predictors while holding all other predictors at their average. Partial dependence plots are useful to visualize low-order interactions within the multivariate range of the predictor variables; however, they should not be used to interpret relationships outside the range of the data or where there are complex interactions between predictors and the response (Hastie et al. 2001, Cutler et al. 2007). I constrained the axes on partial dependence plots to avoid false extrapolation.

After identifying important predictor variables with RFA, I used RTA to better explain the nature of relationships between the six top-ranked predictor variables and the response variables. RTA is a non-parametric technique that recursively partitions a dataset into subsets that are increasingly homogeneous with regard to the response (Breiman et al. 1984, De'ath and Fabricius 2000). RTA produces a set of decision rules on predictor variables that are easily interpreted as a dendrogram. Most implementations of RTA over fit to a given dataset by creating splits that do not significantly reduce the variance (De'ath and Fabricius 2000). Trees are typically pruned back to include only partitions assumed to be valuable beyond the sample data. I used an implementation of RTA called conditional inference trees, which forces statistical significance at each split (Hothorn et al. 2006). This technique prevents over-fitting and the need for pruning. The algorithm, as applied to my data, was as follows: (1) Test the null hypothesis of independence between any of the predictor variables and the response. Stop if this hypothesis cannot be rejected. Otherwise, select the input variable with strongest association to the response. This association is measured by a p-value estimated from a Monte Carlo randomization test of a single input variable and the response. Due to the tendency of spatially dependent data to inflate the significance of hypothesis tests (Dale 1999), I set the minimum criteria for applying a split conservatively at $p < 0.005$ (see below). (2) Implement a binary split in the selected input variable. (3) Recursively repeat steps one and two.

To more fully understand the relationship between plantation age and crown damage, I further analyzed its relationship with crown damage using generalized least squares regression with a spherical spatial correlation structure to accommodate positive spatial autocorrelation within the data (see below). Based on information regarding the non-linear pattern of height growth and increasing crown base heights within southwest Oregon conifer plantations (Hann and Scrivani 1987, Hanus et al. 2000), I included a polynomial term in the regression equation, which allowed the relationship to vary over time.

The potential for spatial autocorrelation to affect statistical tests should be considered in all analyses of spatial phenomenon (Legendre 1993, Dale 1999), particularly fire effects (Bataineh et al. 2006). Fire severity has positive spatial autocorrelation, which results in increased occurrence of Type-I errors (incorrectly rejecting H_0). Ideally, samples should be spaced beyond the range of autocorrelation. In the case of the Biscuit Fire, however, significant spatial autocorrelation in crown damage existed at distances greater than 4800 m; therefore, it would have been impossible to simultaneously collect a sufficient sample and adequately disperse my sample units. Given this constraint, I took several steps to examine the effects of autocorrelation and limit my exposure to its negative consequences. For the Monte Carlo randomization tests, I reported exact p-values to provide information, rather than predetermining an ecologically meaningful level for α, which can be unknowable in the face of autocorrelation (Fortin and Dale 2005). To guard against Type-I errors within the RTA algorithm, which required a criterion for partitioning, I set α conservatively to 0.005 (sensu, Dale and Zbigniewicz 1997). I examined the degree of autocorrelation in the RFA and RTA model residuals using empirical semivariograms. With spatially partitioned predictor variables, regression trees are spatially heterogeneous in functional form; in other words, the decision rules can change based on their location on the landscape (McDonald and Urban 2006). As a result, residuals from RTA and RFA models often have low levels of autocorrelation when the predictors variables are themselves spatially structured (Cablk et al. 2002). Finally, I used generalized least squares (GLS) for the polynomial regression of plantation age on crown damage. GLS models allow residuals to have a nonstandard covariance structure (Venables and Ripley 1997); I used a spherical spatial correlation structure, which described the data well.

RESULTS

Overall patterns of crown damage

Median total crown damage in the Biscuit Fire was 74% (Table 4.5; Figs 4.3, 4.4). Median crown damage was highest in shrub-stratum cover (96%) intermediate in hardwood cover (88%), and lowest in conifer cover (53%) (Table 4.5; Fig. 4.3). The median damage in large conifer cover (32%) was lower than in mixed-size conifer cover (53%), which was lower than in small conifer cover (62%) (Table 4.5; Fig. 4.5).

Ninety-seven percent of plots (6.25 ha) and 88% of cells (0.25 ha) experienced >1% crown damage (Fig. 4.3). Ten percent of plots and 33% of cells experienced 100% crown damage. Seventy-five percent of plots and 62% of cells retained more than 10% of their live pre-fire conifer cover; 52% of plots and 39% of cells retained more than 10% of their pre-fire hardwood cover.

Plots that had burned previously, during in 1987 Silver Fire, had higher median pre-fire shrub-stratum cover (11 versus 4%) and large conifer cover (32 versus 10%), but lower median hardwood cover (8 versus 40%), small conifer cover (0.4 versus 7%) and mixed-size conifer cover (0 versus 10%), than did edaphically similar areas (non-ultramafic) outside the Silver Fire (Table 4.5). Median total crown damage was similar inside and outside the Silver Fire (62 and 57%, respectively). Median relative damage to the individual cover types was also similar (Fig. 4.6).

Ultramafic plots had higher pre-fire shrub-stratum (46 versus 6%), small conifer (13 versus 5%), and bare/grass cover (6 versus 1%), but lower hardwood (17 versus 30%), mixed-size conifer (4 versus 8%) and large conifer cover (0 versus 12%) than did non-ultramafic plots (Table 4.5; Fig. 4.4). Ultramafic plots experienced higher total crown damage (92 versus, 59%; Table 4.5; Fig. 4.6). Ultramafic areas had similarly high median damage to the shrub-stratum (95 versus 97%), but higher median damage to hardwood (95 versus 76%), and conifer cover (78 versus 37%; Fig.4.7).

Total crown damage models

RFA explained 45% of variation in total crown damage. Shrub-stratum cover was, by far, the most important predictor variable (Fig. 4.8); increasing shrub-stratum cover was associated with increasing crown damage (Fig. 4.9). Average temperature and Burn Period were similarly important and were ranked second and third, respectively. Large conifer cover was ranked fourth and was associated with decreasing total damage (Fig. 4.9). The RTA of total crown damage produced 7 terminal nodes (Fig. 4.10). The first partition was based on whether shrub-stratum cover was > 28%; when it was, median crown damage was > 85%. Lower levels of total damage were generally associated with lower levels of shrub-stratum cover, particularly during period C when shrub-stratum cover was below 2%. Low levels of total crown damage were also associated with large conifer cover above 80% during periods A and B.

Semivariograms showed substantial spatial autocorrelation in the crown damage data to lag distances greater than 4500 meters, but most, but not all, of the autocorrelation was explained by the predictor variables within the RFA and RTA models (Fig. 4.11). Model residuals had low levels of spatial autocorrelation across all response variables where spatially partitioned predictors (i.e. daily weather variables, Burn Period and precipitation) were included.

Conifer damage models

RFA explained 38% of variation in relative conifer damage. Average daily temperature, burn period and shrub-stratum cover were ranked as the first, second and third most important predictors but permuting the values of any resulted in similar increases in the MSE (Fig. 4.8). Increasing shrub-stratum cover, particularly between 0 and 10%, was associated with a sharp increase in damage to the available conifer cover (Fig. 4.9). The RTA of conifer damage produced 5 terminal nodes (Fig. 4.12); the first split partitioned the data based on the presence or absence of ultramafic soils. The lowest levels of conifer damage occurred on non-ultramafic soils during periods A and C. The highest levels of conifer damage occurred on ultramafic soils when the average temperature was >25°C.

Hardwood damage models

RFA explained 37% of variation in relative hardwood damage. Shrub-stratum cover was, by far, the most important predictor, followed by temperature, elevation, and relative humidity (Fig. 4.8). Increasing elevation was associated with increasing damage to the available hardwood cover (Fig. 4.9). The RTA of hardwood damage produced 7 terminal nodes (Fig. 4.13); the first split partitioned the data based on whether shrub-stratum cover was > 7%. High hardwood damage was associated with low humidity and high shrub-stratum cover. The lowest hardwood damage occurred at elevations below 666 m when shrub-stratum was below 7%.

Burn variability models

RFA explained 22% of variation in the burn variability data and identified shrub cover as the most important predictor variable (Fig. 4.8). Temperature, burn period, and relative humidity were ranked second, third, and forth, respectively. The RTA of burn variability produced 7 terminal nodes (Fig. 4.14); the first split partitioned the data based on the presence or absence of ultramafic soils. Plots that burned during low relative humidity (<17%) on ultramafic soils had the least variability. Plots that burned during period C on non-ultramafic soils had high variability.

Plantations

Median crown damage within plantations was 89% (Table 4.5). RFA explained 32% of variability in crown damage and identified plantation age as, by far, the most important predictor variable (Fig. 4.15). Average annual precipitation, elevation and topographic position were ranked second, third, and forth respectively. Increasing average annual precipitation was associated with decreasing crown damage. The RTA of plantation damage produced 3 terminal nodes (Fig. 4.16) and both splits were based on plantation age. Median damage in plantations > 35 years was 17%. Plantations < 35 years and > 17 years had a median crown damage of 75%. Plantations < 17 years had a median crown damage of 98%. A polynomial GLS regression of age on crown damage was significant at $p < 0.0001$ (Fig. 4.17). The inclusion of the polynomial term significantly improved the fit compared to a simple linear model ($\Delta AIC = 14.1$).

DISCUSSION

Large fire mosaics typically contain a substantial amount of unburned area (Agee 1993, Turner and Romme 1994, Pyne et al. 1996, DeLong and Kessler 2000).

For example, unburned patches made up over 25% of the area within the 1988 Yellowstone Fires (Turner et al. 1994) and over 15% of the 2002 Hayman Fire (Graham 2003). In this respect, the Biscuit Fire was unique. There was evidence of burn damage on some portion of nearly every plot (97%) and the vast majority of cells (88%) I examined. To the extent I could measure surface fuels with aerial photos (i.e. the shrub stratum), 96% were damaged by the fire. Similarly, in an examination of 180 Forest Service inventory field plots within the Biscuit Fire perimeter, 178 had evidence of a surface fire (Campbell et al. 2007). Halofsky and Hibbs (In Press.) sampled riparian areas within the Biscuit Fire and 90% of their plots had evidence of surface fire; the remaining unburned riparian plots were all directly adjacent to large streams and rivers (order 3 and above). Based on these data, it seems reasonable to conclude that that virtually all of the area within the burn perimeter—even unproductive ultramafic sites and mesic riparian areas—had fuel conditions (in terms of quantity, connectivity, and moisture) that were sufficient to carry surface fire. The ubiquity of surface fire speaks to the extreme climatic and weather conditions at the time of burning. The extent to which it was also a product of elevated fuel hazard resulting from fire suppression is unknown, although the reburn area offers some insight into that issue (see below).

While the Biscuit Fire may have been omnipresent on the surface, it was far from uniform in the tree-stratum. Indeed, almost half of the conifer cover was undamaged by the fire and just 10% of plots experienced total crown damage.

Understanding the patterns of Biscuit Fire crown damage was the motivation for this study and is what guided the development of my five research questions:

1. What was the relative importance of weather, topography, and fuel for predicting patterns of crown damage?

RFA explained between 17 and 45% of variation in the various measures of crown damage. It is difficult to gauge this against other studies of wildfire effects because none have used RFA, which employs a unique method of cross validation to determine the proportion of variance explained. Given the highly stochastic nature of fire behavior and my coarse measurements of fire weather and suppression, I believe that RFA was largely successful in measuring and ranking variable importance.

Overall, topographic variables were less important predictors of crown damage than were weather or fuel. I had hypothesized greater levels of damage on southerly aspects, which has been reported elsewhere in the region, presumably owing to drier and more vulnerable (smaller) fuels (Weatherspoon and Skinner 1995, Alexander et al. 2006). However, aspect was not significantly related to crown damage in the Biscuit Fire. Interestingly, though, summary statistics show slightly higher rates of damage on east facing slopes. It is possible that countervailing effects of dry winds out of the northeast, greater solar radiation in the southwest, and a maritime climate influence on west facing slopes confounded any relationship between aspect and the pattern of crown damage. I found no significant relationship between elevation and total crown damage or conifer damage. However, higher elevations were associated with greater hardwood damage (Figs 4.8 and 4.13). On one hand, this is surprising, given that lower elevations are typically warmer and drier. On the other, hardwood trees tend to be larger at lower elevations within the region (Atzet et al. 1991) and may have been more resistant to fire. This hypothesis is supported by an examination of 175 forest inventory plots within the study area: average pre-fire hardwood DBH was 8cm on plots above 666 m and 16cm on lower plots. Steeper slopes and higher topographic positions have been associated with higher burn severity elsewhere (Albini 1976, Kushla and Ripple 1997, Taylor and Skinner 1998, Lentile et al. 2006b) and I expected the Biscuit Fire to show similar patters. However, I found no significant relationships. I considered the possibility that the size of my plots was too large to detect relationships between topographical variables and crown damage, but re-running the RFA using a single cell (0.25 ha) from each plot as the sample unit did not significantly change the relative importance of predictor variables.

Weather variables were among the most important predictor variables for all response variables considered. In fact, average daily temperature was ranked either first or second in each RFA. Higher temperatures correspond with lower fine fuel moisture, less energy required for pre-heating of fuels (Pyne et al. 1996). Therefore, it was not surprising that higher temperatures were associated with greater levels crown damage and lower levels of burn variability. Lower relative humidity, which largely coincides with higher temperatures, was also an important predictor of crown damage. There was a striking example of this within the RTA of burn variability (Fig. 4.14) where, of the seven terminal nodes, the most and least variable pattern of crown damage was based only on whether or not relative humidity was below 17.6%.

Burn period was also among the top predictors for all response variables, and, by all measures, crown damage was most severe during period B. Burn period is strongly related to the rate of spread of the Biscuit Fire. Half of the area burned occurred during the 9 days that constituted period B, when wind speeds were highest and predominantly out of the northeast (Table 4.2). This finding is consistent with accounts of widespread torching, crowning, running, and spotting during this period reported by Forest Service personnel (USDA Forest Service 2002). It is common during large fires for a disproportionate area to burn in a very short time (Bessie and Johnson 1995, Agee 1997). For example, half of the 55,000 ha Hayman Fire in Colorado, which also occurred in 2002, burned in a single day (Graham 2003). It was impossible to attribute differences in fire effects between burn periods to changing weather or to increasing fire suppression efforts. The 2002 fire season was the nation's second largest on record (NIFC 2008) and in mid July (period A), inter-agency fire fighting resources were dispersed throughout the country and unable to converge on the Biscuit Fire (GAO 2004). By August 5 (the start of period C), however, the Biscuit Fire had become the region's highest priority fire and extensive resources were dedicated to its suppression (in sum, $153 million was spent on Biscuit Fire suppression (GAO 2004)). Throughout the duration of the fire, but mostly during Period C, fire fighters set extensive controlled fires, called "burn-outs," to reduce available fuels and contain the fire (USDA Forest Service 2002). There are no official records describing their location or severity, but they are believed to be widespread. It is impossible to know how many of my plots were within burn-outs, though, undoubtedly, many were. I considered fire suppression to be yet another potential influence on the pattern of severity—increasing damage in some areas and decreasing it in others. And although burn period likely captured some of variation explained by suppression activities, most was impossible to reconstruct. Therefore, the influence of fire suppression, like local weather, is an unknown source of error that contributed to the unexplained variance within my models.

Despite the importance of weather for predicting patterns of Biscuit Fire damage, and the propensity for extreme weather to override other aspects of the fire environment (Bessie and Johnson 1995, Pyne et al. 1996), fuel conditions were also among the most important predictors of crown damage. My remaining research questions are essentially refinements on the fuels component of this first question; therefore, my discussion of the influence of fuels on crown damage is given below.

2. Did patterns of crown damage differ between cover types and did the presence of some cover types affect damage in other cover types in close proximity?

Not surprisingly, shrub-stratum cover experienced the highest levels of damage within the Biscuit Fire. Median damage to shrub-stratum cover was 96% and, accordingly, increasing pre-fire cover of shrub-stratum vegetation corresponded to increased total crown damage. Due largely to predictably high levels of damage within the shrub-stratum, total crown damage had the highest proportion of variance explained out of the four response variables modeled. Shrub-stratum fuels are available on the surface, have high surface-area-to-volume ratios, and are associated with flashy and sometimes intense fire (Albini 1976, Anderson 1982, Graham et al. 2004). Because I used aerial photography, my estimates of shrub-stratum cover and damage apply only to areas where shrub-stratum cover is the over story and a tree-strata is sparse or nonexistent, such as typically occurs on less productive sites or after disturbance.

Within the tree stratum, hardwood cover experienced greater damage than conifer cover. This, too, was expected given their respective anatomies and life history strategies. Though it is impossible to know from these data, it is likely that actual fire-related mortality was lower in hardwood trees compared to conifers. Hardwoods in this region (mostly tanoak and madrone) are aggressive basal sprouters that quickly reoccupy sites after disturbance (Atzet et al. 1991). Only very young hardwoods are typically killed by fire (Tappeiner et al. 1984). Lacking the ability to sprout, most conifers in the region have evolved a “resistor strategy” for dealing to fire (with the notable exception of knobecone pine, which is serotinous). Their ability to resist increases with size, which is consistent with the fact that median crown damage within large conifer cover was 32%, compared to 62% within small conifer cover.

Damage to both hardwood and conifer increased with the amount of shrub cover within the plot (Fig. 4.8). Shrub-stratum cover can increase vertical continuity, effectively carrying fire and heat into the tree-stratum (Sandberg et al. 2001). Even when present at low levels, shrub-stratum cover was related to significant increases in total crown damage, beyond the damage to the shrub-stratum itself. This is exemplified within the RTA of total damage (Fig. 4.11), where closed canopy conditions (<2% shrub-stratum cover) were associated with some of the lowest levels of total crown damage. Presumably, canopy shade in these areas resulted in less vertical fuel continuity, cooler temperatures, and higher fuel moisture, which, in turn, reduced overstory burn damage. It is important to note that my estimates of total crown damage in closed canopy forests was also influenced by my use of aerial photos; I was unable to measure surface fire effects that were obscured by the overstory.

Contrary to expectations, I found no evidence that the presence of hardwood cover or small conifer cover in the plot had an impact on damage to other cover types. The RFA of conifer damage ranked hardwood cover 9th and small conifer cover 14th for relative importance out of 20 predictor variables (Fig.4.8). In addition, mixed-sized conifer cover experienced levels of damage that were intermediate between small and large (median = 52%), which suggests that multi-storied conifer stands did not increase the level of damage by increasing vertical fuel continuity. Instead, it may be simply that the small tree component of the mixed-sized stands was heavily damaged while the large tree component was not.

3. Did the pattern of crown damage differ between areas with and without a recent history of fire?

I had approached this question with multiple working hypotheses because plausible arguments could be made in support of higher, lower, or no difference in crown damage. I found that Biscuit Fire crown damage within the twice-burned landscape was similar to edaphically comparable areas without a recent history of fire. In fact, RFA ranked the variable for “Inside Silver Fire” among lowest predictors for all response variables considered (Fig. 4.8). However, the similarity in total damage between the once and twice-burned landscapes was partially the result of different pre-fire cover types canceling each other out. The Silver Fire burned at lower severity than the Biscuit Fire and had much more variable patterns of damage at the plot-scale (Chapter 3). This produced a complex legacy of fuel. As expected, the level of pre-fire shrub-stratum cover was higher in the twice-burned landscape, but, because the Silver Fire was not particularly severe and tree crowns filled in some of the damaged overstory during the intervening years, the difference was not too great (median = 11% versus 3%). If this were the only difference, the twice-burned landscape may have had greater total crown damage. However, presumably as a result of damage during the Silver Fire, the twice burned landscape had lower pre-fire hardwood cover, which was more likely to burn during the Biscuit Fire. The effect of greater hardwood cover outside the Silver Fire effectively increased levels of total damage such that difference between median total crown damage was less than 5%. Interestingly, the once-burned landscape also had higher levels of pre-fire small and mixed-conifer cover, while the twice-burned landscape had considerably more pre-fire large conifer cover. This may also be due to the Silver Fire reducing the proportion of small conifers in 1987, but without pre-Silver Fire data it is impossible to know. The magnitude of difference in pre-fire vegetation conditions is too large to be a product of the Silver Fire alone, which suggest there were other differences between the two landscapes unrelated to fire history. Nonetheless, these findings suggest that using fire intervals to assess fire hazard in this region may be overly simplistic and that different aspects of the fuel environment (i.e. cover types) have different post-fire trajectories over time. It is unclear if the effect of the Silver Fire would have been different with shorter or longer intervals between fires.

4. Did the pattern of crown damage differ between ultramafic and non-ultramafic soils?

Ultramafic sites have low plant productivity and conventional wisdom suggests that, due to low fuel, loads they do not typically experience high severity fire (Whittaker 1960, Atzet and Wheeler 1982, Kruckeberg 1984, Taylor and Skinner 2003, Skinner et al. 2006). Therefore, it was surprising to find that the ultramafic plots experienced higher total crown damage and higher levels of damage to conifer and hardwood cover. Upon closer examination, however, there are several attributes of the fire environment that seem to explain why ultramafic sites experienced higher damage in this case. First, it is important to note that “ultramafic” as a plot descriptor ranked 10th out of 20 variables for predictive importance (Fig. 4.8). The most important predictor of total crown damage was shrub-stratum cover, which was more than eight times higher on ultramafic sites. It is possible that fire exclusion may have prevented some low severity burns in the past, which increased shrub biomass and that, in turn, increased horizontal fuel continuity facilitating fire spread. But, this theory seems tenuous without additional data documenting long term changes in shrub biomass on ultramafic soils. In any case, much higher shrub-stratum cover in ultramafic plots can account for the difference in total crown damage. Also, the fact that much of the total crown damage in ultramafic areas was due to shrub-stratum damage suggests that although severity (fire effect on vegetation) was higher in these regions burn intensity (energy released) was probably not.

With regard to higher damage in the tree-stratum, note that although the first split in the RTA of conifer damage was based on the presence of ultramafic soils, RFA identified average temperature as the most important predictor, and ranked ultramafic sixth. This contrast highlights the strength and weaknesses of the two techniques. RTA is a "greedy algorithm," meaning that it is very sensitive to splits made at the top of the tree (Venables and Ripley 1997). All subsequent splits are conditional on the splits above, which can lead to the exclusion of important variables from the regression tree when there is correlation among one or more predictor variables. By using bootstrap resampling and iteratively selecting subsets of predictors then averaging over many trees, RFA and makes a more robust, and less biased, judgment of predictor importance (Prasad et al. 2006). Daily average temperature, therefore, was a more important predictor of relative conifer damage. Nonetheless, the RTA showed that splitting the plots based on ultramafic soils created two groups that were most homogeneous with regard to the proportion of conifer cover damaged. This is because a higher proportion of ultramafic plots burned during period B (53% versus 40%) and ultramafic sites had much higher shrub-stratum cover—both of which were associated with higher conifer damage. In effect, the indicator for ultramafic soils combined two of the top three predictors of conifer damage as identified by RFA.

Furthermore, conifer trees were smaller on ultramafic sites (Table 4.5), which would have made them more susceptible damage when confronted with a surface fire. In addition, open tree canopies, as are typical on ultramafic sites, may have resulted in lower crown base heights, which offer greater potential for surface fire to transition to the crown (Van Wagner 1973, 1977). A examination of crown ratios on 175 forest inventory plots measured eight to ten-years before the fire supports this hypothesis: average live crown ratios on ultramafic sites was 58% (n=54) and 45% on non-ultramafic sites (n=119). Open conditions may have also resulted in hotter and drier microclimatic conditions on the surface, which can increase surface fire intensity, spread rates, and crown scorch (Pollet and Omi 2002). Finally, higher wind velocity at the surface, also a function of open tree canopies, may have lowered fuel moisture and increased fire intensity and spread rates.

5. Did the pattern of crown damage differ between plantations and unmanaged forests and how did plantation age affect the level of damage?

Given their ubiquity on western landscapes, there has been surprisingly little research regarding fire hazard in even aged silviculture systems. The empirical research that has been done suggests that plantations are associated with elevated fire hazard (Weatherspoon and Skinner 1995, Odion et al. 2004). This is consistent with my findings from the Biscuit Fire, where plantations had higher median levels of damage (89%) than did the unmanaged stands (74%). Not surprisingly, the level of damage in plantations was strongly related to its age. The RTA for crown damage in the plantations produced only two significant partitions, and both were based on age—one at 17 years and the other at 35. Plantations younger than 17 years tended to experience near total crown damage. High fire hazard in young plantations has also been found through fire behavior modeling studies, where higher rates of mortality were predicted for conifer plantations compared to old and young growth reserves across all diameter classes up to 50 cm DBH, regardless of weather conditions (Stephens and Moghaddas 2005). It's worth noting, however, that young stands, whether managed or not, tended to burn at high severity in the Biscuit Fire, as was show in areas that regenerated after the 1987 Silver Fire. (Thompson et al. 2007 and see chapter 3). Stephens and Moghaddas (2005) also reported that once trees were >50 cm DBH, predicted mortality was similar between plantations and unmanaged stands. This is also consistent with my finding that plantations older than 35 years experienced significantly lower mortality within the Biscuit Fire. By the time a Douglas-fir plantation in southwest Oregon is 35 years old, trees are typically between 12 and 20 meters tall and have crown base heights > 3.5 meters off the ground, depending on site class and stem density (Hann and Scrivani 1987, Hanus et al. 2000). The GLS regression suggested that plantation's ability to resist crown damage is non-linear and increases more sharply after approximately age 20.

Unfortunately, I had no reliable records documenting the management history within the plantations. In the only detailed examination of fire effects within plantations that I am aware of, the method of site preparation was the most important predictor of plantation crown damage: broadcast burned sites experienced significantly less damage than untreated or piled-and-burned sites (Weatherspoon and Skinner 1995). Similarly, in a fortuitous experiment regarding fire effects after thinning within the Biscuit Fire, tree mortality was lowest (5%) on sites that were thinned in 1996 then broadcast burned in 2001, just one year before the fire, intermediate in unmanaged sites (53-54%), and highest in sites that were thinned in 1996 but not broadcast burned (80-100%; Raymond and Peterson 2005). Studies like the two mentioned above, where good records exist describing historical management, are very rare. Given the difficulty of conducting experiments with large wildfires, it is important that good records are kept so that more can be learned from wildfires.

Ecological and management implications

Large heterogeneous burn mosaics, such as occurred on the Biscuit Fire, have lasting effects on ecosystem and successional dynamics (Turner and Romme 1994, DeLong and Kessler 2000, Zenner 2005). One concern following large fires has been that severely burned patches will lack a proximal seed source to regenerate the conifer overstory (Romme et al. 1998). This, combined with a history of regeneration failures after logging (Hobbs et al. 1992), initially raised concerns that the Biscuit burn area would not return to conifer dominance (Sessions et al. 2004). However, recent studies have found abundant natural conifer regeneration occupying sites 9–19 years after stand-replacing wildfires in the region (Shatford et al. 2007). My findings suggest that despite the size and severity of the Biscuit Fire, there remains considerable live conifer cover to assist regeneration throughout the burn area. This is supported by early regeneration surveys, four years after the fire, that found natural conifer regeneration in excess of 1000 stems ha-1 out to 400 m from a seed source (Donato 2008). Also, based on satellite-derived burn severity and vegetation maps, Donato (2008) estimated that that 58% of the non-ultramafic Biscuit Fire area was within 200 m of a live tree edge, and 81% was within 400 m. Donato (2008) did not consider regeneration on the severely burned ultramafic sites, where limited soil nutrients and water holding capacity will likely result in a substantially longer period of recovery. Moreover, these sites also have tremendous botanical diversity and high rates of endemism (Whittaker 1960). The extent to which the burn damage in these areas may have impacted regional biodiversity is unknown, but potentially significant.

The other major biodiversity concern within the Biscuit Fire relates to old growth forests. Federal land management on the RSNF region of the fire is largely focused on maintaining and promoting development of late-successional forest ecosystems (FEMAT 1993, USDA-USDI 1994). The Biscuit Fire burned through almost 70,000 ha of late successional reserves, which were established to expressly for this purpose. In the aftermath of the fire there was some speculation that the size and severity of the Biscuit Fire might be related to presence of older forests, which were thought to be uncharacteristically dense and hazardous due to fire suppression (Spies et al. 2006). My findings suggest, however, that the older forests were the most resistant to damage. Median crown damage to large conifer cover was just 32% and increasing proportions of large conifer cover and closed canopy forests were associated with lower levels of total crown damage. This suggests that treating old forests in this region may not be necessary to increase their resistance to fire, and may, in fact, be counterproductive.

Much of the non-ultramafic portion of the study area has high forest productivity compared to other fire-prone ecosystems in the western U.S. (Waring et al. 2006). As a result biomass accumulation, mainly in the form of sprouting hardwoods and shrubs, is rapid in the wake of fires (Shatford et al. 2006, Donato 2008). High levels of competition in this early seral environment can result in protracted rates of succession to a tree-stratum (Hobbs et al. 1992, Shatford and Hibbs 2006, Lopez-Ortiz 2007). In the interim, early seral, shrub-stratum vegetation is vulnerable to repeated high severity fire (Thompson et al. 2007, Donato 2008). Given the extent of damage caused by the Biscuit Fire, the hazard presented by regenerating vegetation will likely be a persistent management concern on the RSNF. Forest managers have few options for reducing fire risk in this environment. Evidence from within the Silver Fire suggests that traditional silviculture methods (i.e. salvage logging and planting) do not reduce the hazard presented by early seral vegetation (Thompson et al. 2007), at least in the short term. It is possible that tree planting may accelerate transition out of the shrub-stratum and toward fire resilience. In the years following the Biscuit Fire, most of the severely burned plantations were replanted by the RSNF. Protecting these new plantations from future fires long enough for them to become resistant to fire will be a significant challenge for managers. More research is clearly needed to help managers develop options for managing regenerating vegetation in the wake of high severity fire. Managers may consider strategically placing, and frequently maintaining, thinning and prescribed burning treatments in configurations that might slow the spread of future fires enabling protection of key structures and habitat conditions (e.g. spotted owl habitat areas) within the landscape (Ager et al. 2007).

Although is clear that the Biscuit Fire was uncharacteristic in terms of its size, it is difficult to compare the level of crown damage in the Biscuit Fire to past fires in the region. Regrettably, there is no standard method for quantifying burn severity (Jain 2004, Lentile et al. 2006a). A comparison of two studies that both used aerial photos to measure crown damage after the complex of 1987 fires that burned throughout the Klamath region in northern California highlights how different severity classification schemes can produce widely differing conclusions. Odion et al. (2004) used a three class system: “low” corresponded to <50% crown scorch, “moderate” corresponded to 50-100% crown scorch and “high” corresponded to 100% of crowns scorched or consumed. In contrast, Weatherspoon and Skinner (who looked at a sub-set of the same fires) used four severity classes: “low” corresponded to <10% crown scorch, “moderate” corresponded to 10 to 50% crown scorch, “high” corresponded to >50% of crowns scorch and “extreme” corresponded to > 50% crowns consumed. With their three class system, Odion et al. (2004) estimated that the proportion of crown damage was 59% low, 29% moderate, and 12% high severity. With their four class system, Weatherspoon and Skinner (1995) estimated that approximately 40% of the area burned with low severity, 30% with moderate, 25% with high, and 5% with extreme. Applying the cutoffs from Odion et al. (2004) to my photo-plots yields 32% low, 58% moderate, and 10% high. Applying cutoffs from Weatherspoon and Skinner (1995) (and collapsing their high and extreme classes) yields 9% low, 23% moderate, and 68% high. This is a striking difference; more than six times as much high severity in one classification scheme as compared to the other. Depending on which of these two approaches is used, the Biscuit Fire was either a characteristically low-to-moderate severity fire, or it contained more than twice as much high severity area as the 1987 fires.

Such ambiguities in burn severity classification schemes are what led me to report crown damage on a continuous scale. Even still, characterizing Biscuit Fire "severity" is problematic. Unquestionably, the Biscuit Fire resulted in extensive crown damage over a very large area. However, it is also true that almost half the conifer cover was not damaged. Therefore, the moniker "mixed-severity fire" seems apt, albeit not very informative. Indeed, the very idea of fire severity is troublesome and the informal use of the term often leads to confusion (Jain 2003). Even when it is defined as a fire's impact on vegetation (sensu Agee 1993)—which is a comparatively narrow use of the term—there is still a wide margin of just what is meant by burn severity. Is complete crown damage to shrub-stratum vegetation equally severe as complete crown damage to the tree-stratum? Is complete crown damage in a stand of mature conifers equally severe as complete crown damage in a stand of evergreen hardwoods, which will aggressively sprout back to reoccupy the site in just a few years? It is only through explicit descriptions of fire effects that managers can effectively respond to large landscape fires. Simplistic characterizations of fire effects will lead to unsuitable one-size-fits-all management protocols that do not adequately respond to any one condition.

This study, like all retrospective studies of wildfire effects, was opportunistic and observational. As such, the scope of inference is limited to the Biscuit Fire. Patterns of crown damage were unique and it is impossible to know the extent to which they are generalizable to other large mixed-severity wildfires, even within the bioregion. Experimental replication of large landscape fires is obviously impractical; moreover, large landscape fires occur during extreme weather and through a fuel environment that experimental fires could never reproduce (van Mantgem et al. 2001). Therefore, studying large infrequent wildfires, like the Biscuit Fire, yields knowledge that could not be gained any other way (Turner and Dale 1998). And, despite the fact that formal inference is limited, these findings are important, as they offer a formal accounting of the effects of one of the most ecologically significant disturbances to impact Pacific Northwest forests in the last 150 years.

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Table 4.1. Weather summary statistics from four Remote Automated Weather Stations (RAWS) distributed within and nearby the fire boundary. Weather variables are daily averages between 10:00 and 17:00. Dates for minimum and maximum values are given in parentheses.

RAWS Station	Latitude	Longitude	Elevation (m)	Min & Max Daily Temperature (C°)	Min & Max Daily Rel. Humidity (%)	Min & Max Daily Wind Speed (km/hr)
Agness	42:55:20	-124:05:80	46	21, 35 (8/24, 7/29)	20, 58 (8/15, 7/22)	6, 20 (8/13, 7/30)
Illionois Valley	42:06:14	-123:41:07	423	20, 39 (8/05, 8/14)	9, 39 (8/15, 7/22)	5, 14 (8/13,7/27)
Onion Mountain	42:45:40	-123:61:55	1353	12, 33 (8/05, 8/13)	7, 56 (8/15, 8/05)	6, 14 (8/04, 7/28)
Quail Prarie	42:14:31	-124:02:43	970	15, 34 (8/05, 8/13)	10, 65 (8/15, 7/22)	5, 19 (7/15, 8/22)

Table 4.2. Dates, area, and weather information for burn periods describing the spread of fire and the resources used for fire suppression. Weather variables are averages of the daily average between 10:00 and 17:00 within the Burn Period. Wind direction has been cosine transformed such that a value of -1 is associated with southwesterly winds and a value of +1 is associated with northeasterly winds.

Burn Period	Start-Stop	No. Days	Hectares (% total)	Temperature (C°)	Relative Humidity (%)	Wind Speed (km/hr)	Wind Direction	Suppression Effort
A	7/13-7/26	14	8995 (5)	26.2	36.1	11.8	0.03	Low
B	7/27-8/04	9	102723 (50)	26.6	32.4	12.5	0.42	Moderate
C	8/05-9/06	30	91480(45)	28.4	25.4	10.9	0.52	High

Table 4.3. Summary statistics for response and predictor variables used in the random forest and regression tree analysis of unmanaged stands within the Biscuit Fire. Plots with < 5% pre-fire cover of a response variable were excluded from that analysis (see text).

	All Plots (n=761)			
Response Variables	Median	Mean	Min	Max
All Crown Damage	73.9	64.8	0	100
Relative Conifer Damage	53.2	52.3	0	100
Relative Hardwood Damage	88.1	74.3	0	100
Burn Variability	16.8	16.3	0	43.7
Predictor Variables				
Large Conifer Cover	4.9	17.1	0	100
Small Conifer Cover	8.1	13.3	0	85.4
Mixed-size Conifer Cover	6.8	11.8	0	83.2
Hardwood Cover	22.7	28.1	0	92.8
Shrub Cover	14.6	23.4	0	92.8
Bare/Grass Cover	2.8	6.2	0	80.6
Shannon Diversity	0.64	0.61	0	0.95
Elevation (m)	781	775	92	1476
Topographic Position (Fine)	1.7	1.45	-58.4	55.6
Topographic Position (Coarse)	3.8	2.7	-271.8	275.6
Slope (%)	49.8	49.9	9.5	92.3
Beer's Aspect	0.02	0.02	-0.97	0.98
Average annual precipitation (cm)	279	284	160	424
Temperature (C)	28.7	27.4	15.2	34.9
Relative Humidity (%)	28.5	29.3	9.3	53.6
Wind Speed (km/hr)	11.6	11.7	5.1	19.1
Wind Direction *	0.54	0.43	-0.58	0.91
Burn Period	A=8%, B=44%, C=48%			
Ultramafic	Ultramafic=68%, Nonultramafic=32%			
Inside Silver Fire	Inside Silver=23%, Outside Silver=77%			

Table 4.4. Summary statistics for response and predictor variables used in the random forest and regression tree analysis of plantations within the Biscuit Fire .

	Median	Mean	Min	Max
Response Variable				
All Crown Damage	89.8	77	0	100
Predictor Variables				
Age (yrs)	21	22	5	42
Harvest Size (ha)	9.7	14.1	1.5	95
Veg Cover %	91.1	89.8	47	100
Bare/Grass Cover %	6.8	10.3	0	53
Elevation (m)	935	885	265	1346
Topographic position (Fine)	1.11	3.4	-25.7	49.4
Topographic position (Coarse)	20.3	26.8	-176.4	203.3
Slope (%)	39.9	40	12	79
Beer's Aspect	-0.25	-0.2	-0.97	0.99
Average annual precipitation (cm)	322	320	171	439
Temperature (C)	26.5	26.6	16.6	35.8
Relative Humidity (%)	26.1	31	10	65.5
Wind Speed (km/hr)	8.3	9	4.2	18.2
Wind Direction (cosine transformed)	0.33	0.24	-0.3	0.75

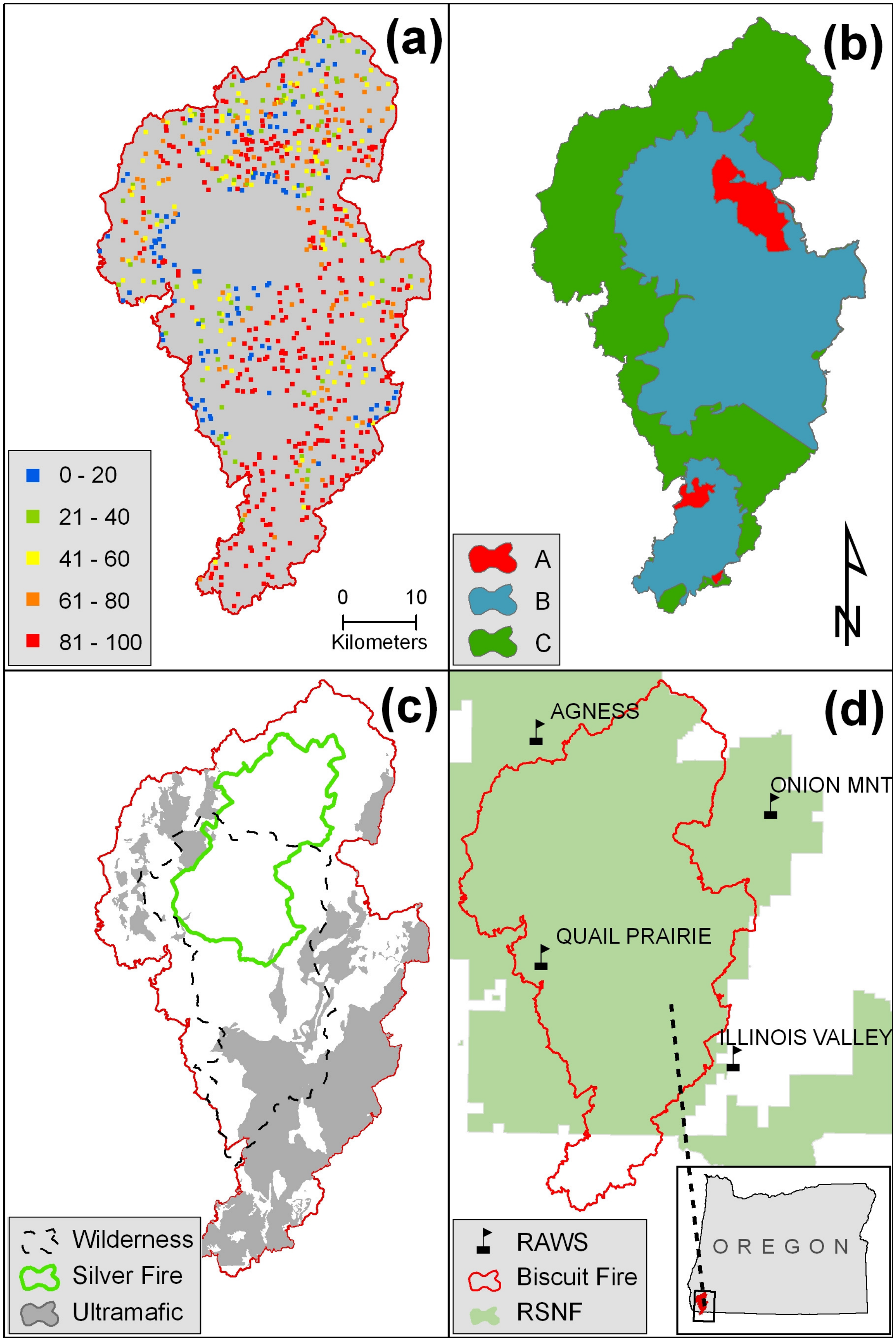
Table 4.5. Summary of vegetation and fire damage by landscape-type interpreted from aerial photos taken before (2000) and after (2002) the 2002 Biscuit Fire on 761 photo-plots (6.25ha). Crown damage is calculated for each plot as a percent of the pre-fire cover; plots without pre-fire cover in a cover class were excluded.

	All Plots (n=761)				Non-ultramafic (n=529)				Ultramafic (n = 232)			
	Median	Mean	Min	Max	Median	Mean	Min	Max	Median	Mean	Min	Max
Pre-fire Shrub	14.6	23.4	0.0	92.8	5.6	13.4	0.0	92.8	46.1	45.9	5.6	89.3
Post-fire Shrub	0.0	4.2	0.0	86.3	0.0	1.2	0.0	39.5	1.8	4.6	0.0	89.3
Pre-fire Hardwood	22.7	28.1	0.0	93.0	30.4	33.0	0.0	92.8	17.3	16.9	0.0	46.0
Post-fire Hardwood	1.4	8.7	0.0	81.9	3.2	11.8	0.0	81.9	0.3	1.8	0.0	29.0
Pre-fire Small Conifer	8.2	13.3	0.0	85.4	4.8	11.6	0.0	85.4	13.4	17.1	0.0	81.0
Post-fireSmall Conifer	1.7	5.8	0.0	81.8	0.0	1.4	0.0	81.8	2.4	4.9	0.0	36.3
Pre-fire Large Conifer	4.0	17.0	0.0	100.0	11.8	23.5	0.0	100.0	0.0	2.1	0.0	36.2
Post-fire Large Conifer	0.6	10.5	0.0	100.0	4.2	14.5	0.0	100.0	0.0	0.9	0.0	27.8
Pre-fire Mixed-size Conifer	6.8	11.8	0.0	83.2	8.0	13.5	0.0	83.2	3.8	7.8	0.0	73.2
Post-fire Mixed-size Conifer	1.2	6.1	0.0	79.5	2.0	7.8	0.0	79.4	0.2	2.3	0.0	37.0
Pre-fire Bare/Grass	2.8	6.3	0.0	80.7	1.2	4.7	0.0	70.0	5.5	9.6	0.0	80.6
Post-fire Bare/Grass	2.8	6.3	0.0	80.7	1.2	4.7	0.0	70.0	5.5	9.6	0.0	80.6
Total Damage	73.9	64.8	0.0	100.0	58.9	56.8	0.0	100.0	91.9	83.0	0.0	100.0
Shrub Damage	96.1	89.4	0.0	100.0	97.4	89.3	0.0	100.0	95.0	88.8	0.0	100.0
Hardwood Damage	88.3	74.2	0.0	100.0	76.3	68.1	0.0	100.0	95.0	88.0	0.0	100.0
All Conifer Damage	52.2	52.1	0.0	100.0	37.1	44.5	0.0	100.0	78.2	69.0	0.0	100.0
Small Conifer Damage	61.7	55.5	0.0	100.0	45.0	47.9	0.0	100.0	78.7	69.1	0.0	100.0
Large Conifer Damage	32.4	42.3	0.0	100.0	27.3	39.4	0.0	100.0	73.4	59.3	0.0	100.0
Mixed-size Conifer Damage	52.6	51.8	0.0	100.0	38.7	45.0	0.0	100.0	79.3	67.3	0.0	100.0

Table 4.5 Continued

	Outside Silver Non-ultramafic (n=348)				Inside Silver Non-ultramafic (n=181)			
	Median	Mean	Min	Max	Median	Mean	Min	Max
Pre-fire Shrub	3.7	9.4	0.0	65.0	11.4	21.3	0.0	92.8
Post-fire Shrub	0.5	3.5	0.0	39.0	0.8	1.4	0.0	24.2
Pre-fire Hardwood	39.6	41.6	0.0	92.8	8.1	16.2	0.0	74.6
Post-fire Hardwood	6.4	14.0	0.0	81.9	0.8	5.4	0.0	74.6
Pre-fire Small Conifer	6.9	14.2	0.0	85.0	0.4	6.5	0.0	83.4
Post-fireSmall Conifer	2.6	7.7	0.0	81.0	0.0	3.2	0.0	77.2
Pre-fire Large Conifer	9.6	15.9	0.0	100.0	32.3	38.0	0.0	100.0
Post-fire Large Conifer	2.9	9.6	0.0	100.0	8.4	23.9	0.0	100.0
Pre-fire Mixed-size Conifer	11.2	15.2	0.0	81.8	0.0	10.2	0.0	83.2
Post-fire Mixed-size Conifer	4.0	8.9	0.0	79.5	0.0	5.7	0.0	77.0
Pre-fire Bare/Grass	0.2	3.5	0.0	70.0	4.4	7.2	0.0	43.8
Post-fire Bare/Grass	0.2	3.5	0.0	70.0	4.4	7.2	0.0	43.8
Total Damage	56.8	56.0	0.0	100.0	61.8	58.4	0.0	100.0
Shrub Damage	97.8	87.0	0.0	100.0	95.9	92.0	0.0	100.0
Hardwood Damage	75.5	66.1	0.0	100.0	85.3	72.6	0.0	100.0
All Conifer Damage	36.1	44.1	0.0	100.0	38.2	45.2	0.0	100.0
Small Conifer Damage	41.7	46.2	0.0	100.0	56.8	53.5	0.0	100.0
Large Conifer Damage	25.0	38.9	0.0	100.0	33.3	40.2	0.0	100.0
Mixed-size Conifer Damage	35.7	43.2	0.0	100.0	45.6	50.1	0.0	100.0

Figure 4.1. Maps of the Biscuit Fire in southwestern Oregon and northwestern California, USA. (a) Distribution of 761 randomly placed aerial photo plots color coded to show average crown damage within each plot; (b) Location of the three burn periods (c) Location of ultramafic soils, the 1987 Silver Fire and the Kalmiopsis wilderness; (d) Locations of four remote automated weather stations (RAWS) and the Rouge Siskiyou National Forest (RSNF).


(a)
0 - 20
21 - 40
41 - 60
61 - 80
81 - 100
0 10
Kilometers
(b)
A
B
C
N
(c)
Wilderness
Silver Fire
Ultramafic
(d)
AGNESS
ONION MNT
QUAIL PRAIRIE
ILLINOIS VALLEY
OREGON
RAWS
Biscuit Fire
RSNF

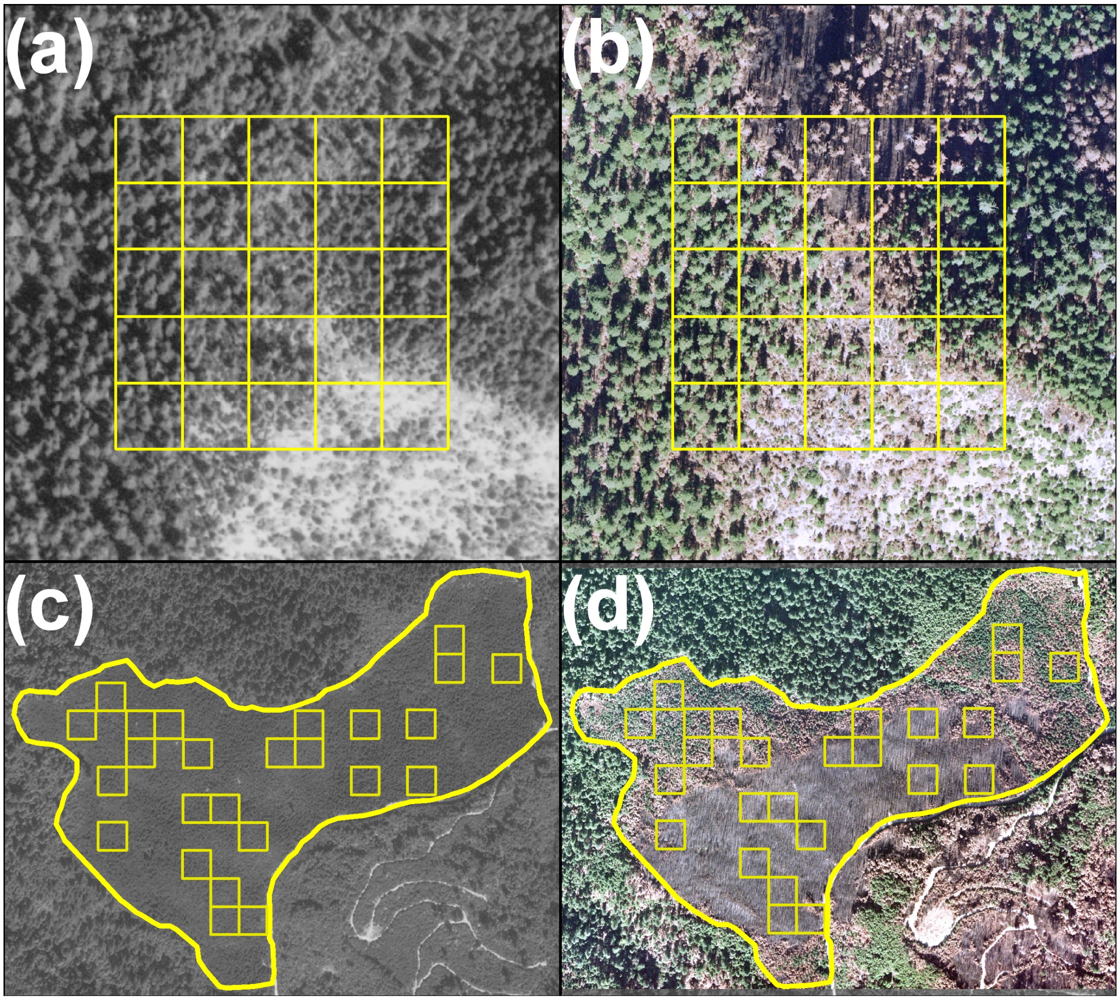


Figure 4.2. Examples of pre- and post-fire aerial photo-plots in unmanaged (a and b) and managed (c and d) stands ; Unmanaged plots are 5-by-5 aggregations of cells (6..25ha). If managed plots were > 6.25ha, 25 cells were randomly placed throughout the unit. Management units smaller than 6.25ha included as many cells as would fit in side the perimeter; no unit smaller than 1.25 ha was used.

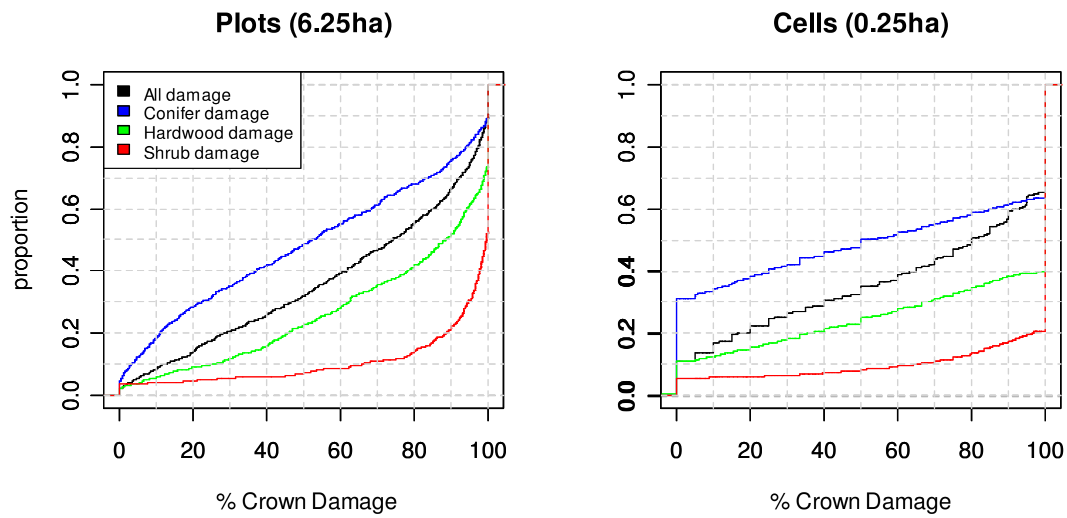


Figure 4.3. Empirical cumulative distributions for crown damage in the Biscuit Fire at the plot (6.25ha) and sub-plot (0.25ha) scale.

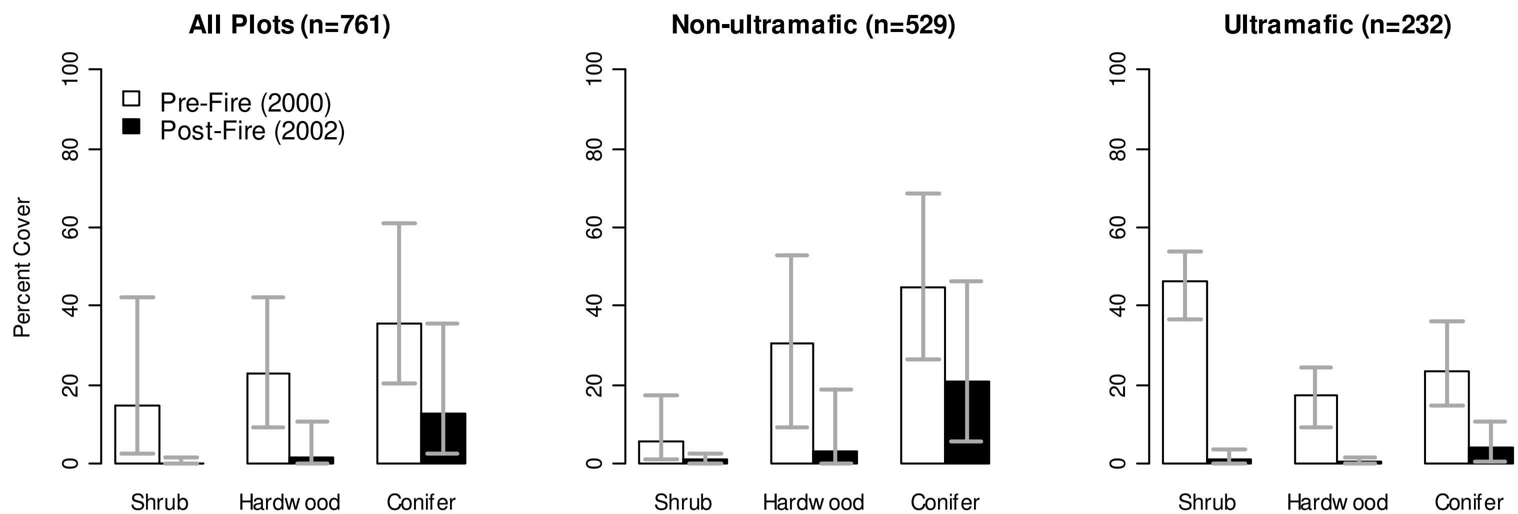


Figure 4.4. Median percent cover of shrub-stratum, hardwood, and conifer cover on all plots, non-ultramafic plots, and ultramafic plots in 2000 and in 2002. Gray bars are inner quartile range (25^{th} & 75^{th} percentiles).

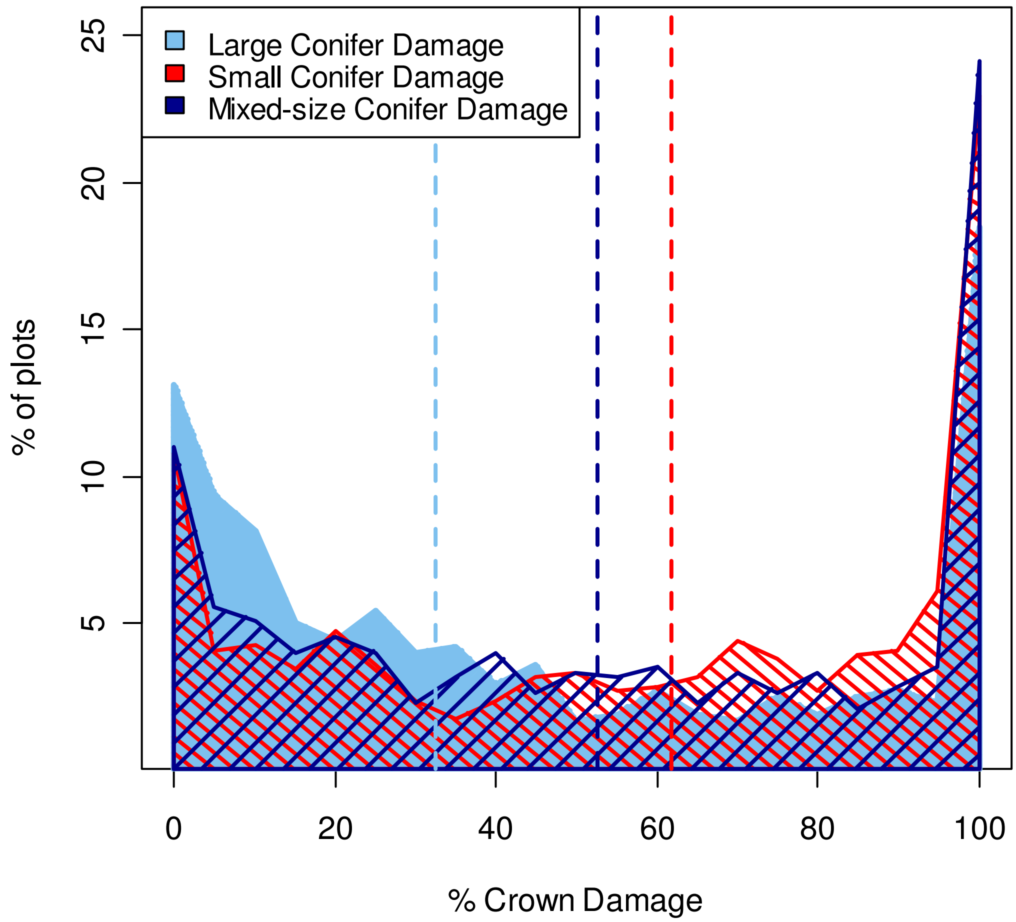


Figure 4.5. Distributions of conifer damage by size class. Vertical lines indicate median values for crown damage in each size class. Crown damage is calculated for each plot as a percent of the pre-fire cover; plots without pre-fire cover in a cover class were excluded.

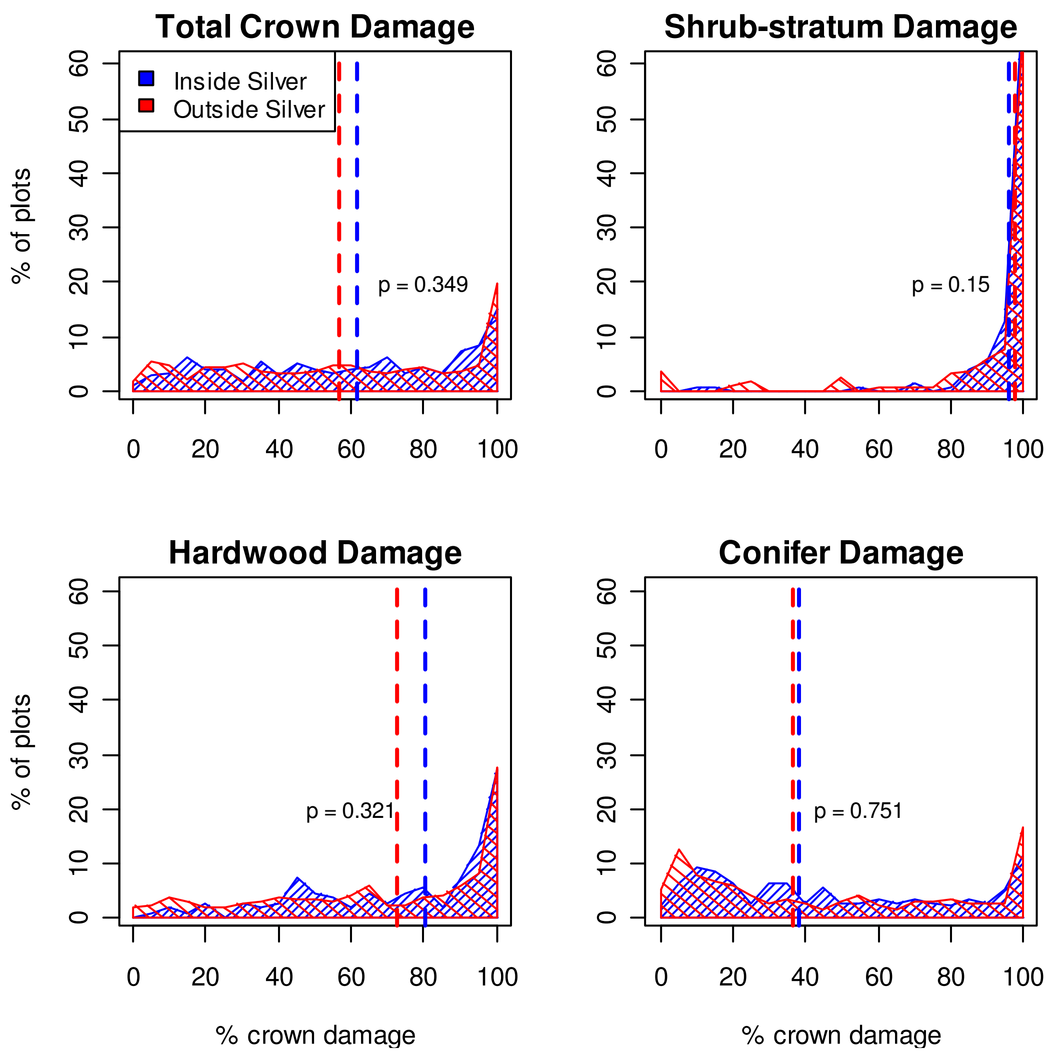


Figure 4.6. Distributions of crown damage on plots inside and edaphically similar areas outside the region of the 1987 Silver Fire. Crown damage is calculated for each plot as a percent of the pre-fire cover; plots without pre-fire cover in a cover class were excluded. Vertical lines indicate median values for crown damage. P-values are from Monte Carlo randomization tests for differences in the median damage.

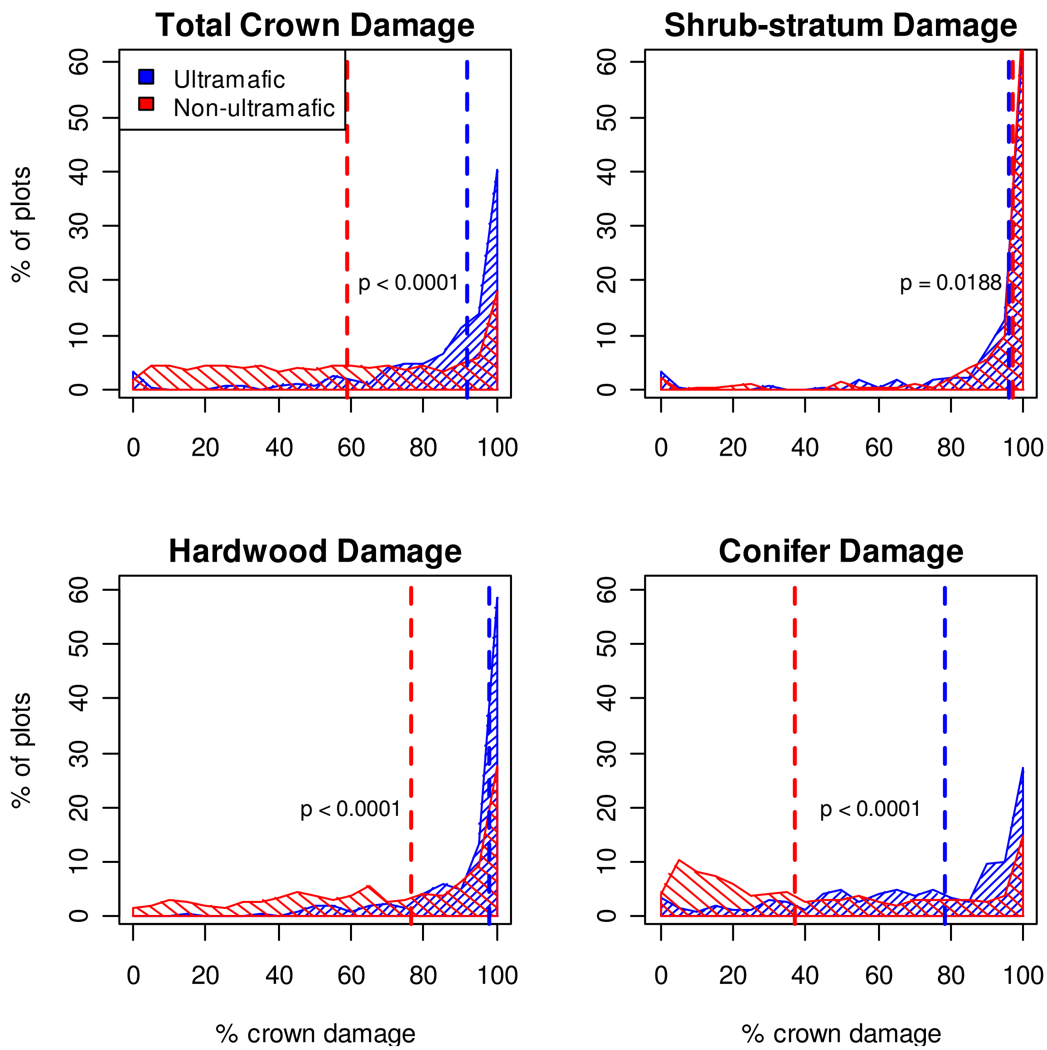


Figure 4.7. Distributions of crown damage on ultramafic and non-ultramafic plots. Vertical lines indicate median values for crown damage. Crown damage is calculated for each plot as a percent of the pre-fire cover; plots without pre-fire cover in a cover class were excluded. P-values are from Monte Carlo randomization tests for differences in the median damage.

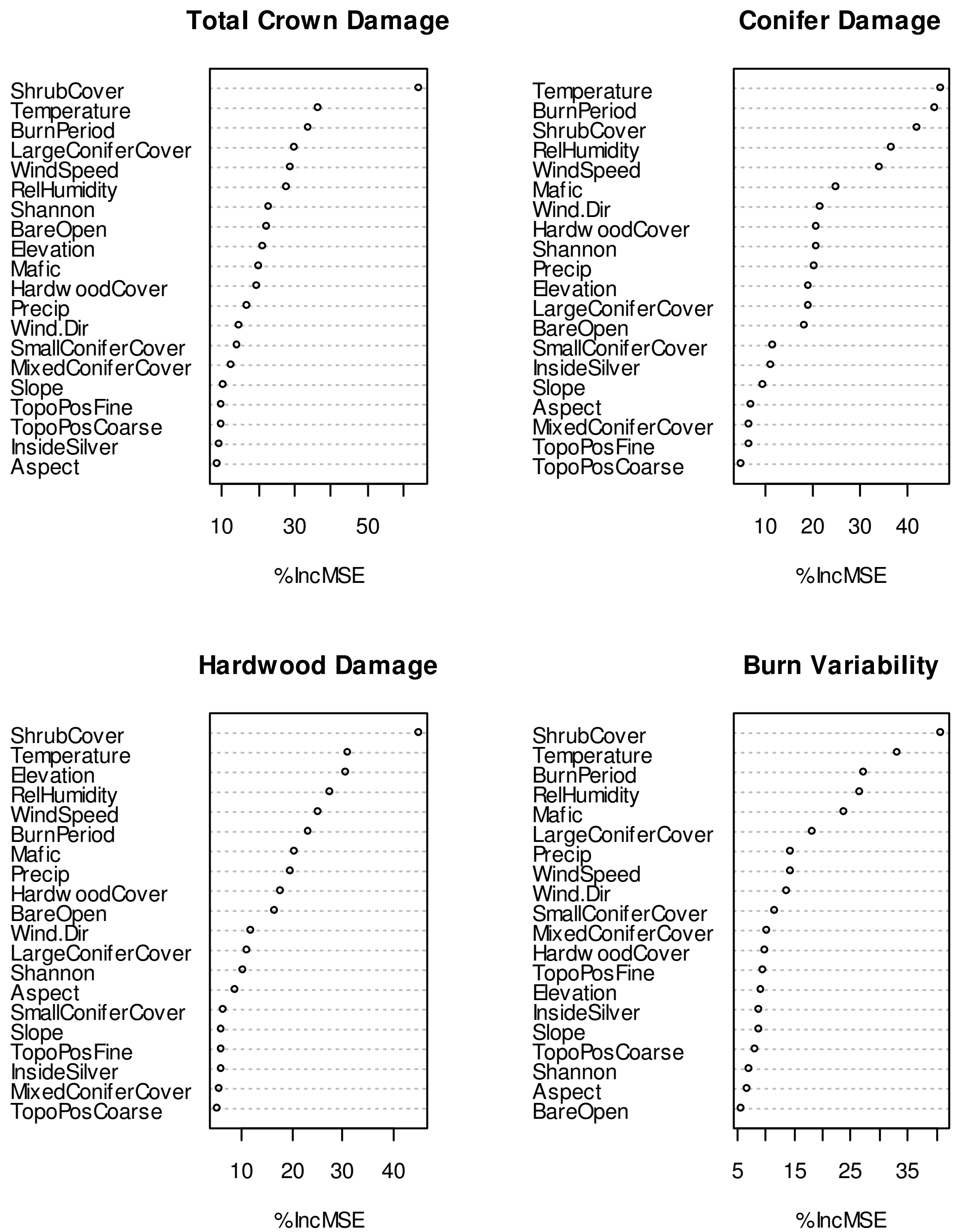


Figure 4.8. Variable importance plots for predictor variables from random forests models for: total crown damage, conifer damage, hardwood damage, and burn variability. Predictor variables are along the y-axis and the average increase in the mean square error when data for that variable are permuted and all other are left unchanged is on the x-axis.

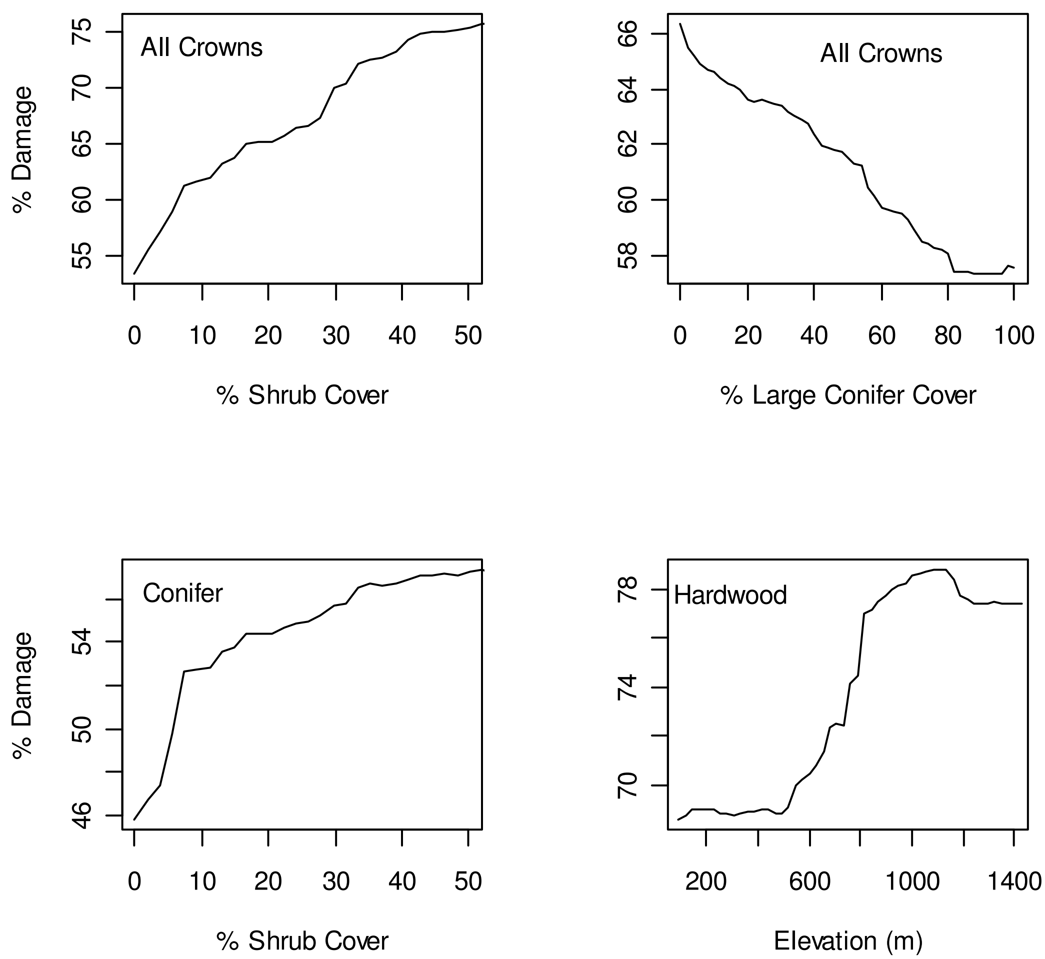
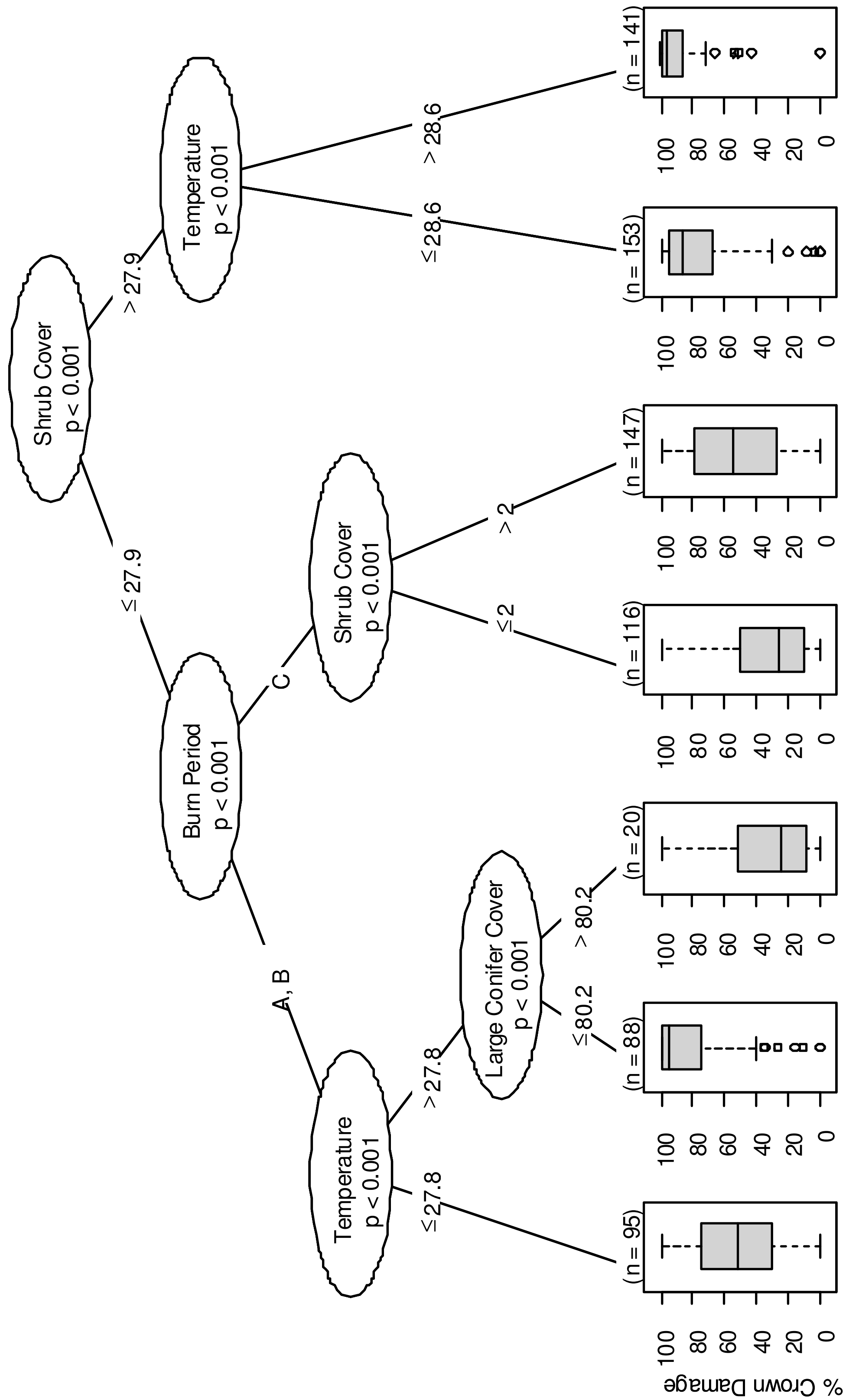


Figure 4.9. Partial dependence plots for random forests predictions of all crown damage on percent shrub cover; total damage on large conifer cover; conifer damage on percent shrub-stratum cover, and hardwood damage on elevation. Partial dependence is the predicted value of the response based on the value of one predictor variable after averaging out the effects of the other predictor variables in the model.

Figure 4.10. Regression tree for total crown damage based on the top six predictor variables from the random forest analysis (see figure 4.8). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range. Dots represent more extreme data points



Shrub Cover
p < 0.001
≤ 27.9
> 27.9
Burn Period
p < 0.001
A, B
C
Temperature
p < 0.001
≤ 28.6
> 28.6
Temperature
p < 0.001
≤ 27.8
> 27.8
Large Conifer Cover
p < 0.001
≤ 80.2
> 80.2
Shrub Cover
p < 0.001
≤ 2
> 2
(n = 95)
(n = 88)
(n = 20)
(n = 116)
(n = 147)
(n = 153)
(n = 141)
100
80
60
40
20
0
% Crown Damage

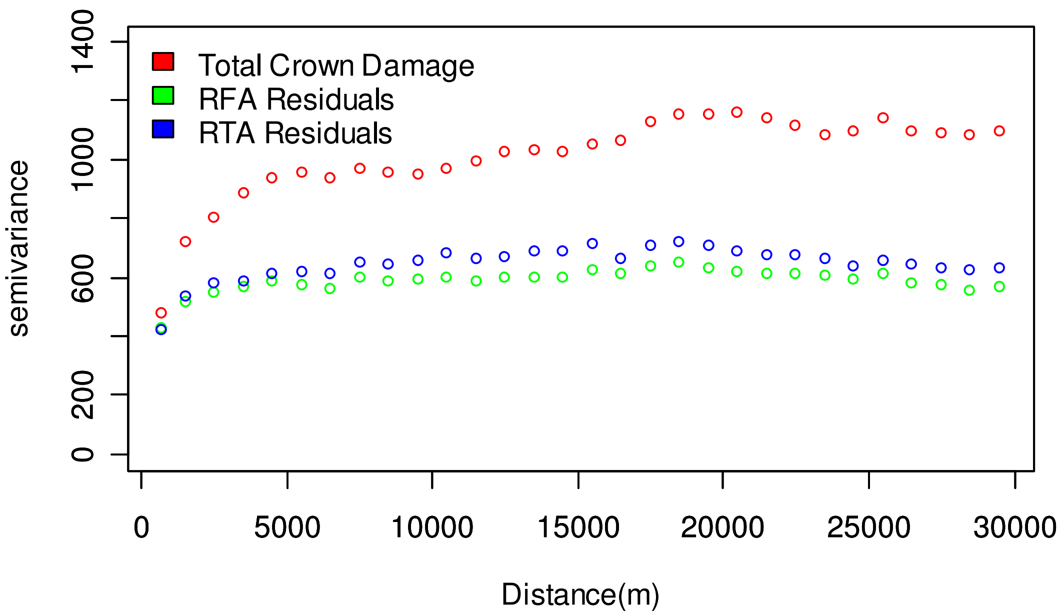


Figure 4.11. Semivariogram depicting spatial autocorrelation in total crown damage plots, and the residuals from the regression tree analysis and random forest analysis for predicting all canopy damage.

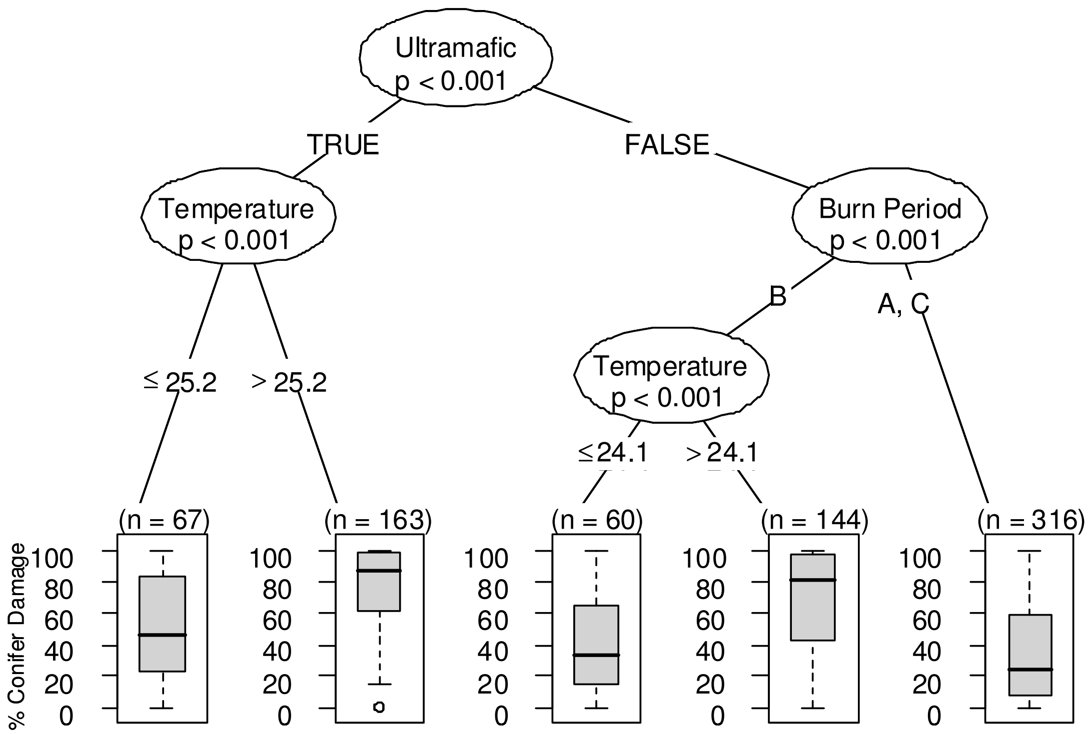
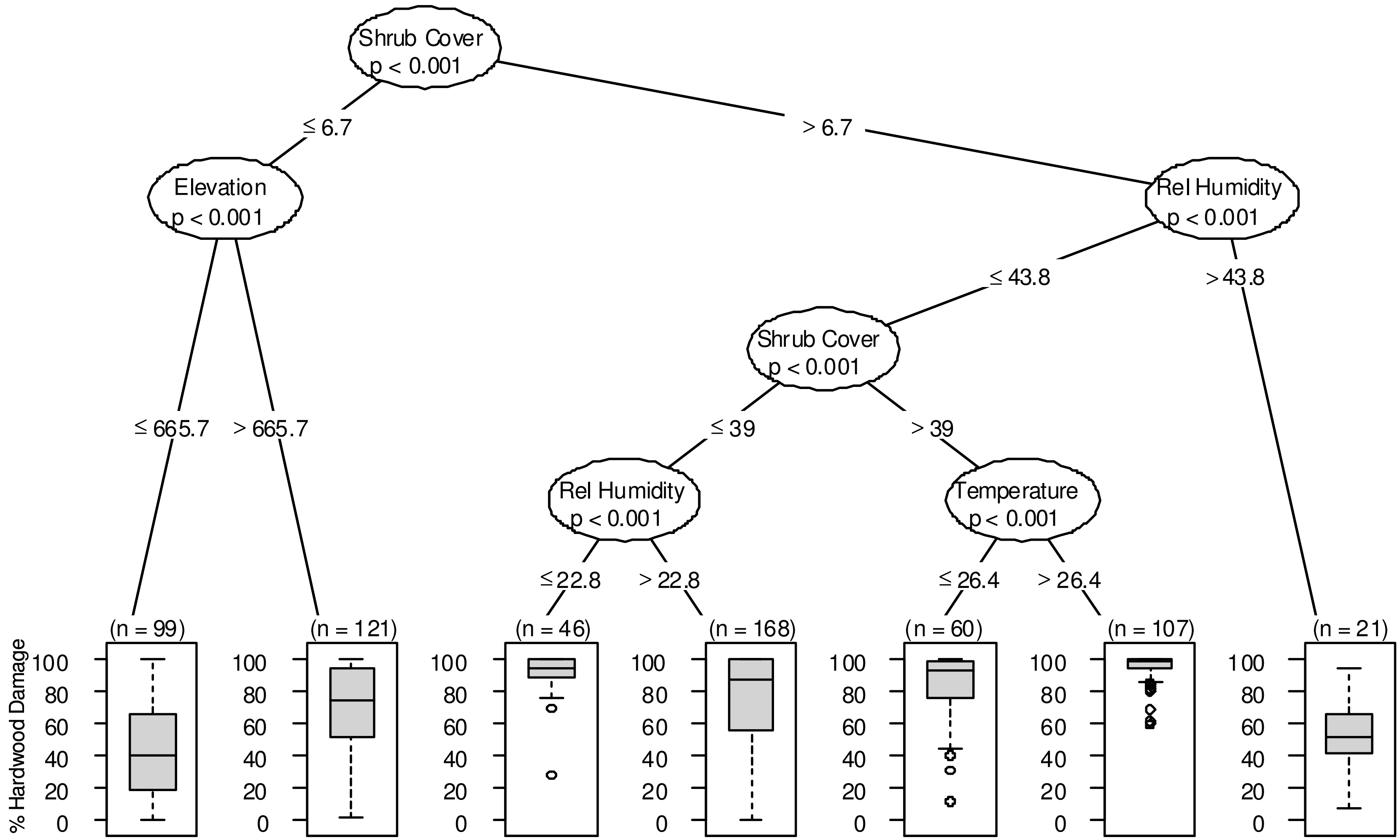


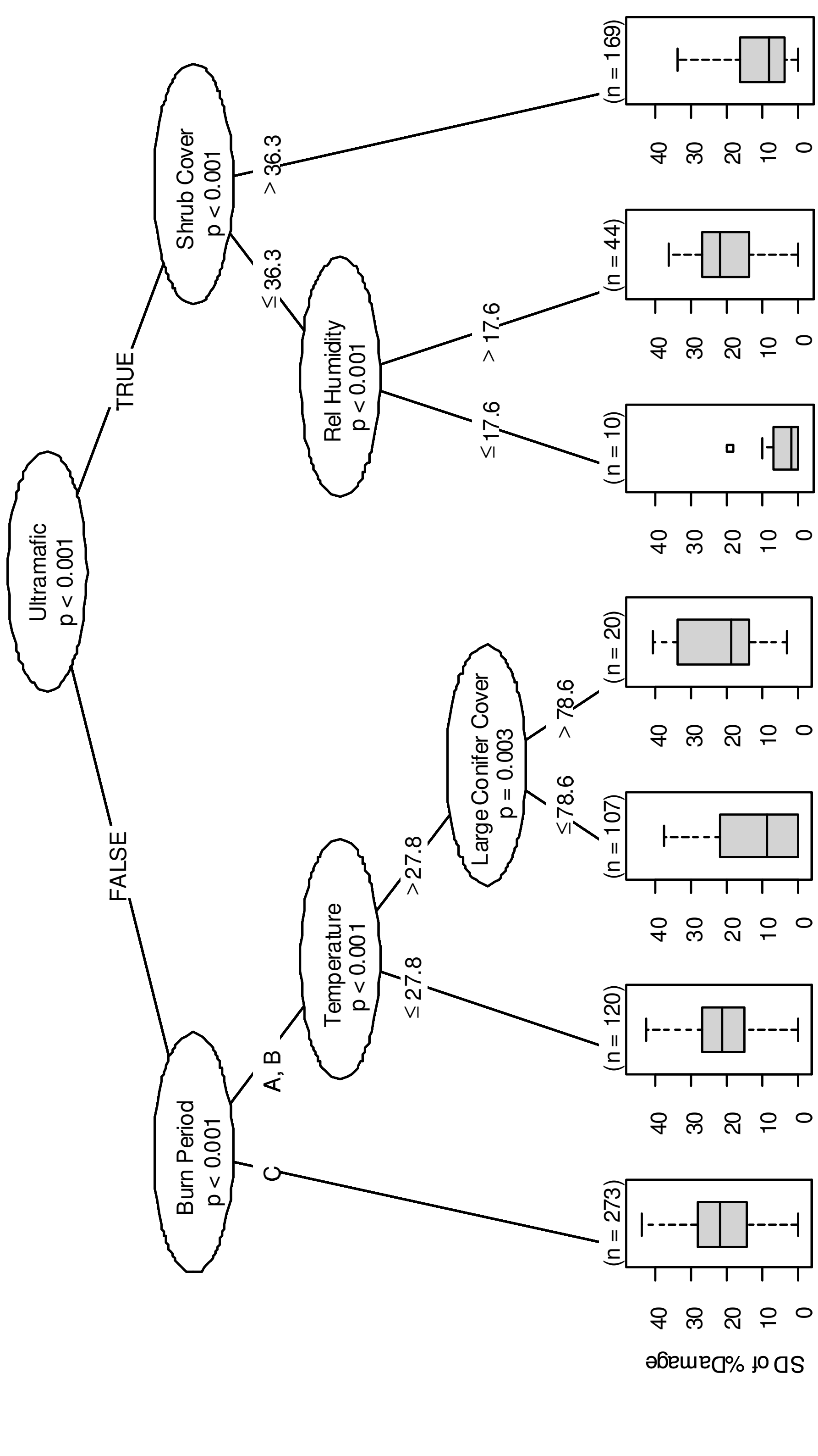
Figure 4.12. Regression tree for conifer damage based on the top six predictor variables from the random forest analysis (see figure 4.8). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range. Dots represent more extreme data points

Figure 4.13. Regression tree for hardwood damage based on the top six predictor variables from the random forest analysis (see figure 4.8). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range. Dots represent more extreme data points



Shrub Cover
p < 0.001
≤ 6.7
> 6.7
Elevation
p < 0.001
Rel Humidity
p < 0.001
≤ 43.8
> 43.8
Shrub Cover
p < 0.001
≤ 665.7
> 665.7
≤ 39
> 39
Rel Humidity
p < 0.001
Temperature
p < 0.001
≤ 22.8
> 22.8
≤ 26.4
> 26.4
(n = 99)
(n = 121)
(n = 46)
(n = 168)
(n = 60)
(n = 107)
(n = 21)
% Hardwood Damage
100
80
60
40
20
0

Figure 4.14. Regression tree for burn variability (i.e. standard deviation of crown damage) based on the top six predictor variables from the random forest analysis (see figure 4.8). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range. Dots represent more extreme data points


Ultramafic
p < 0.001
TRUE
FALSE
Shrub Cover
p < 0.001
≤ 36.3
> 36.3
Rel Humidity
p < 0.001
≤ 17.6
> 17.6
Burn Period
p < 0.001
A, B
C
Temperature
p < 0.001
≤ 27.8
> 27.8
Large Conifer Cover
p = 0.003
≤ 78.6
> 78.6
(n = 273)
(n = 120)
(n = 107)
(n = 20)
(n = 10)
(n = 44)
(n = 169)
SD of %Damage
0
10
20
30
40

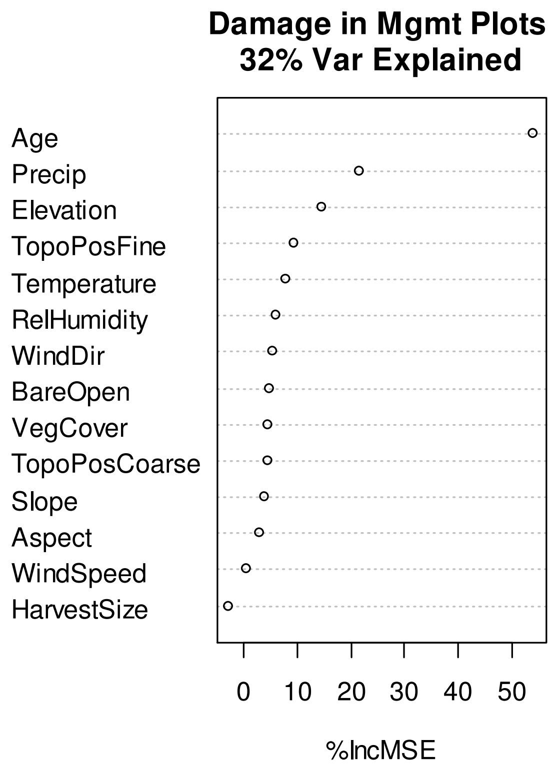


Figure 4.15. Variable importance plots for predictor variables from random forests models for of crown damage within conifer plantations. Predictor variables are along the y-axis and the average increase in the mean square error when data for that variable are permuted and all other are left unchanged is on the x-axis.

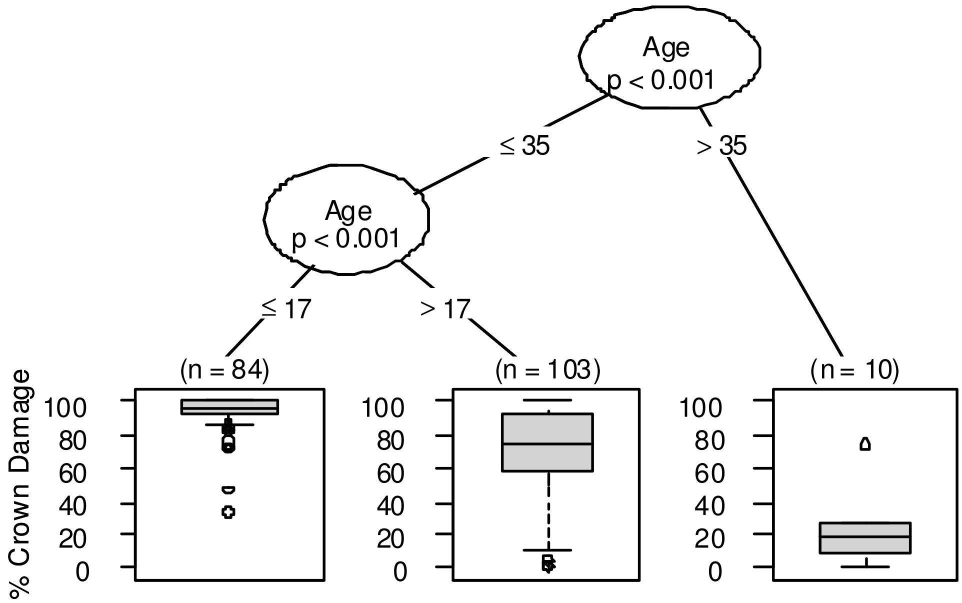


Figure 4.16. Regression tree for crown damage in plantations based on the top six predictor variables (though only age was significantly related to damage) from the random forest analysis (see figure 4.15). Each split in the regression tree is conditional on the splits above. P-values at each node are from a Monte Carlo randomization test. In order for a split to occur the p-value must be < 0.005. Box plots at terminal nodes show the distribution of the data within that branch of the tree. Boxes represent inner-quartile range; horizontal lines within the box represent median values; whiskers extend to the most extreme data point that is no more than 1.5-times the inner-quartile range. Dots represent more extreme data points

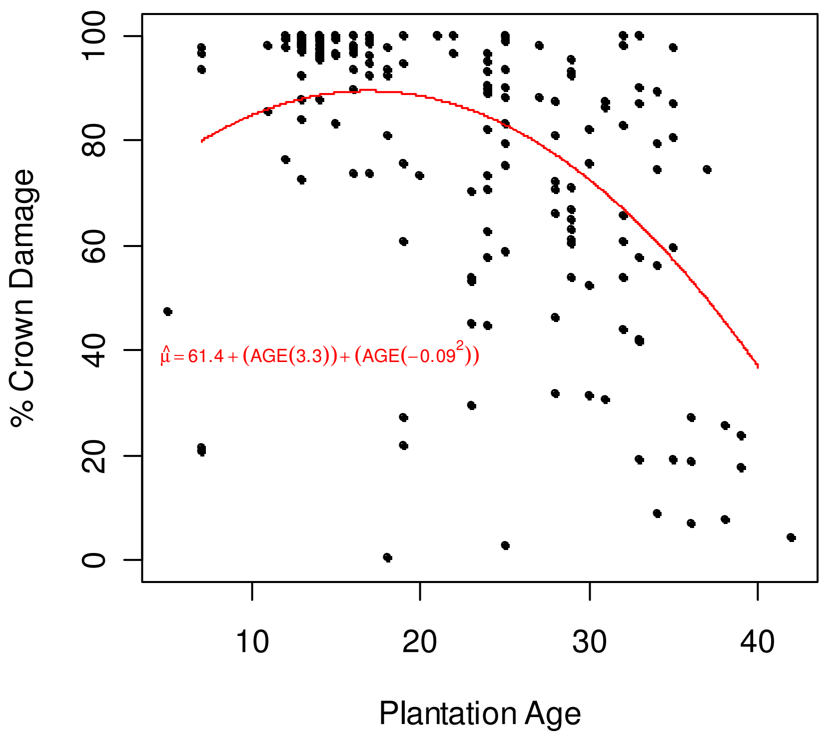


Figure 4.17. Relationship between plantation age and percent crown damage using generalized least squares regression with a spatial spherical correlation structure to accommodate positive spatial autocorrelation.

CHAPTER 5: CONCLUSION

In this dissertation, I investigated factors associated with crown damage within the 200,000 ha Biscuit Fire, the largest fire ever recorded in Oregon. My overarching goal was to describe the relative importance of weather, topography and fuel on patterns of crown damage. The heterogeneity of these factors, in space and in time, is a fundamental determinant of fire severity mosaics (Pyne et al. 1996), and over large spatial scales, greater heterogeneity of these factors is thought to be associated with lower overall severity (Minnich 1983).

In chapter two, I capitalized on a unique configuration of recurring wildfires and post-fire logging to assess the effects of compounding disturbance on burn severity. In 1987, the Silver Fire burned heterogeneously through >38,000 ha of mixed-conifer/evergreen hardwood forests. From 1988 to 1990, approximately 850 ha of the severely burned patches were logged and planted with conifers. In 2002, the Biscuit Fire reburned the entire region. This arrangement of disturbances afforded me the opportunity to address two specific questions in a hypotheses-testing framework. First, was severity in the Biscuit Fire associated with severity in the Silver Fire in unmanaged areas? And second, did areas that were salvaged-logged and planted with conifers after the Silver Fire burn more or less severely in the Biscuit Fire than comparable unmanaged areas? I used the Landsat-based differenced normalized burn ratio (dNBR; Key and Benson 2004) to estimate fire severity and found that areas that burned severely in 1987 tended to re-burn at high severity in 2002, after controlling for the influence of several topographical and biophysical covariates. Areas unaffected by the initial fire tended to burn at the lowest severities in 2002. In addition, areas that were salvage-logged and planted after the initial fire burned more severely than comparable unmanaged areas, suggesting that post-fire logging and planting did not reduce future fire severity as had been suggested by some. The degree to which the areas that experienced stand-replacement fires in 1987 reburned severely in 2002 raises serious concerns regarding the fire hazard on nearly 100,000 ha of the Biscuit Fire that experienced high rates of crown damage and will be dominated by shrub-stratum vegetation for many years.

In chapter three, I continued to focus on the twice-burned landscape, but increased my ecological resolution by interpreting changes in shrub-stratum, hardwood, and conifer cover on digital aerial photo-plots taken at three points in time (1987, 2000, and 2002). Although my general conclusions were congruent to the earlier analysis, I added considerable ecological information. For example, I again found that areas that burned severely in the 1987 Silver Fire reburned severely in the 2002 Biscuit Fire. But, based on the photo analysis, I showed that these areas contained primarily shrub-stratum vegetation and that damage within the tree-stratum was largely independent of the legacy of the Silver Fire. Instead, tree-stratum damage was most strongly related to weather conditions at the time of burning. I also found that hardwood cover experienced greater damage than conifer cover and that large conifer cover experienced the lowest levels of damage. Areas that were salvage logged and planted after the Silver Fire experienced high rates of crown damage during the

Biscuit Fire and had slightly higher rates of damage than areas that burned severely during the Silver Fire but were unmanaged. Higher topographic positions and greater pre-fire vegetation cover appear to explain the small difference. These findings support those in chapter two, which indicated that salvage-logging followed by conifer planting did not reduce future fire severity, as had been hypothesized by some.

In chapter four, I expanded the geographical scope of my analysis to include the entire Biscuit Fire region. I found that, overall, weather and fuels were much more important predictors of crown damage than topography. This was surprising because topography is thought to be a major determinant of fire severity and is often put forth as a primary criterion for allocating fuel treatments and biological reserves (e.g. Taylor and Skinner 2003, Rouge Siskiyou National Forest 2004). Also, I found that, unlike many other large fires, the Biscuit Fire contained almost no unburned patches—97% of plots had some evidence of fire. This speaks to the extreme weather conditions that spread surface fire through even the most fuel limited sites. However, while fire was ubiquitous on the surface, there was substantial burn heterogeneity within the tree-stratum. Indeed, only 10% of photo-plots experienced complete crown damage and almost 50% of the pre-fire conifer cover was undamaged. Throughout the burn, the areas that experienced the least crown damage were those that burned on days during relatively mild weather conditions and within closed canopy forest, particularly where there was a significant component of large conifer cover. These areas will likely be most resistant to future fires as well, which suggests that fuel treatments to counteract the perceived increase in fire hazard (e.g. Spies et al 2006) may be unnecessary, ineffective, or counter productive in these forest types. One third of the fire burned over ultramafic soils, which have low productivity and were expected to burn at lower severities. Surprisingly, therefore, these areas experienced the highest levels of total crown damage and tree-stratum damage within the burn area. This was likely due to much higher levels of shrub-stratum vegetation, as opposed to closed canopy forest. In addition, the ultramafic areas had smaller trees and happened to burn during more extreme weather conditions. Forest recovery will likely be longer on the ultramafic sites due to the unique chemical and physical properties of the soil. Also surprising were the patterns of crown damage inside the Silver Fire perimeter compared to similar areas without a recent history of fire. Given that fire suppression is thought to have increased fuel loads and fire hazard, I expected that areas that were under-burned during the Silver Fire would have had lower levels of damage during the Biscuit Fire. But, in fact, there was no difference between the once and twice-burned regions, either in terms of overall damage or in damage to conifer or hardwood cover. This finding underscores the inadequacy of using fire return intervals and condition class (i.e. the number of interval missed) as a proxy for fire hazard, particularly in mixed severity regimes. As expected, young plantations experienced high levels of crown damage, but by age 35 they were comparatively resistant. Most of the plantations that burned severely have since been replanted (Rogue Siskiyou National Forest 2004) and protecting these young plantations from fire long enough for them to become resistant to fire will be a significant challenge for managers.

Large wildfires, like the Biscuit Fire, have a widely disproportionate impact on forest ecosystems relative to their frequency. In fact, just one-percent of fires are responsible for as much as 96% of the area burned (Strauss et al. 1989). Moreover, a trend toward increasingly larger fires is expected to continue coincident with climate warming (Westerling et al. 2006). Large fires are not necessarily more severe and severe fires are not necessarily large (Agee 1997). However, in recent years, several large, severe fires have raised concerns that land-use change, fire suppression, and climate warming, will conspire to make future fires both large and severe (Covington 2000, Fried et al. 2004, Jain and Graham 2007). Understanding large wildfire dynamics is, therefore, increasingly important for policy-makers and forest managers (Turner et al. 2003, Jain and Graham 2007). Currently, however, the vast majority of scientific information regarding wildfire dynamics is derived from small plots within small wildfires (Turner and Dale 1998, Schmoldt et al. 1999). Consequently, existing management protocols for anticipating fire effects and the parameters used as inputs into fire simulation models are not derived from the type of wildfires that are most pervasive on the landscape. Not surprisingly, therefore, improving scientific understanding of large wildfire behavior has been identified as a significant research priority (Schmoldt et al. 1999).

In this sense, the Biscuit Fire provided a tremendous boon to research. And hopefully, this dissertation, along with all the other investigations into the causes and consequences of the Biscuit Fire (e.g. Azuma et al. 2004, Raymond 2004, Sessions et al. 2004, Ratchford et al. 2005, Donato et al. 2006, Campbell et al. 2007, Fontaine

2007, Thompson et al. 2007, Donato 2008, Halofsky and Hibbs In Press.) will help advance our understanding of large wildfires.

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APPENDIX A: CORRELATIONS BETWEEN LANDSAT DERIVED dNBR AND RdNBR AND AERIAL PHOTO INTERPRETATION OF BISCUIT FIRE SEVERITY

In Chapter 2, I quantified burn severity using the Landsat-derived differenced normalized burn ratio (dNBR), which is a measure of pre- to post-fire change in the ratio of near-infrared (Band 4, 0.76 - 0.90µm) to shortwave infrared (Band 7, 2.08-2.35µm) spectral reflectance. Band 4 is associated with foliage on green trees and understory, while Band 7 is associated with dry and blackened soil (Key and Benson 2004). dNBR is calculated as:

$$dNBR = \left(\frac{prefire(Band4) - prefire(Band7)}{prefire(Band4) + prefire(Band7)} - \frac{postfire(Band4) - postfire(Band7)}{postfire(Band4) + postfire(Band7)} \right)$$

dNBR has been widely used to quantify burn damage, particularly in forested ecosystems (e.g. Odion et al. 2004, Bigler et al. 2005, Finney et al. 2005, Collins et al. 2006, Kulakowski and Veblen 2007, Safford et al. 2007, Thompson et al. 2007, Wimberly and Reilly 2007) .

Recently, Miller and Thode (2007) proposed a relativized version of dNBR—called RdNBR—which they designed to avoid potential problems of using an absolute measure of change on heterogeneous landscapes. Miller and Thode (2007) argue that while absolute change is an appropriate measure of total carbon release or biomass damage, relative change—which adjusts for differences the pre-fire condition—is more appropriate for measuring “ecological burn severity.” RdNBR is calculated as:

$$\text{RdNBR}=\left(\frac{prefire(NBR)-postfire(NBR)}{\sqrt{\left|prefire(NBR)/1000\right|}}\right)$$

For the analyses in chapter 4, I interpreted fire-related crown damage within 761 aerial photo plots (6.25 ha). This presented an opportunity to compare the correspondence between dNBR and photo interpreted crown damage, and to judge whether RdNBR would have improved our characterization of burn damage in chapter 2. I compared the average values of dNBR and RdNBR with the photo-plot areas to the total crown damage interpreted from the photos using polynomial regressions, which described the relationships well (Fig. A.1). R-squared values were 0.93 and 0.92 for the dNBR and RdNBR, respectively. There was no indication that using RdNBR would have altered our analysis or conclusions in Chapter 2.

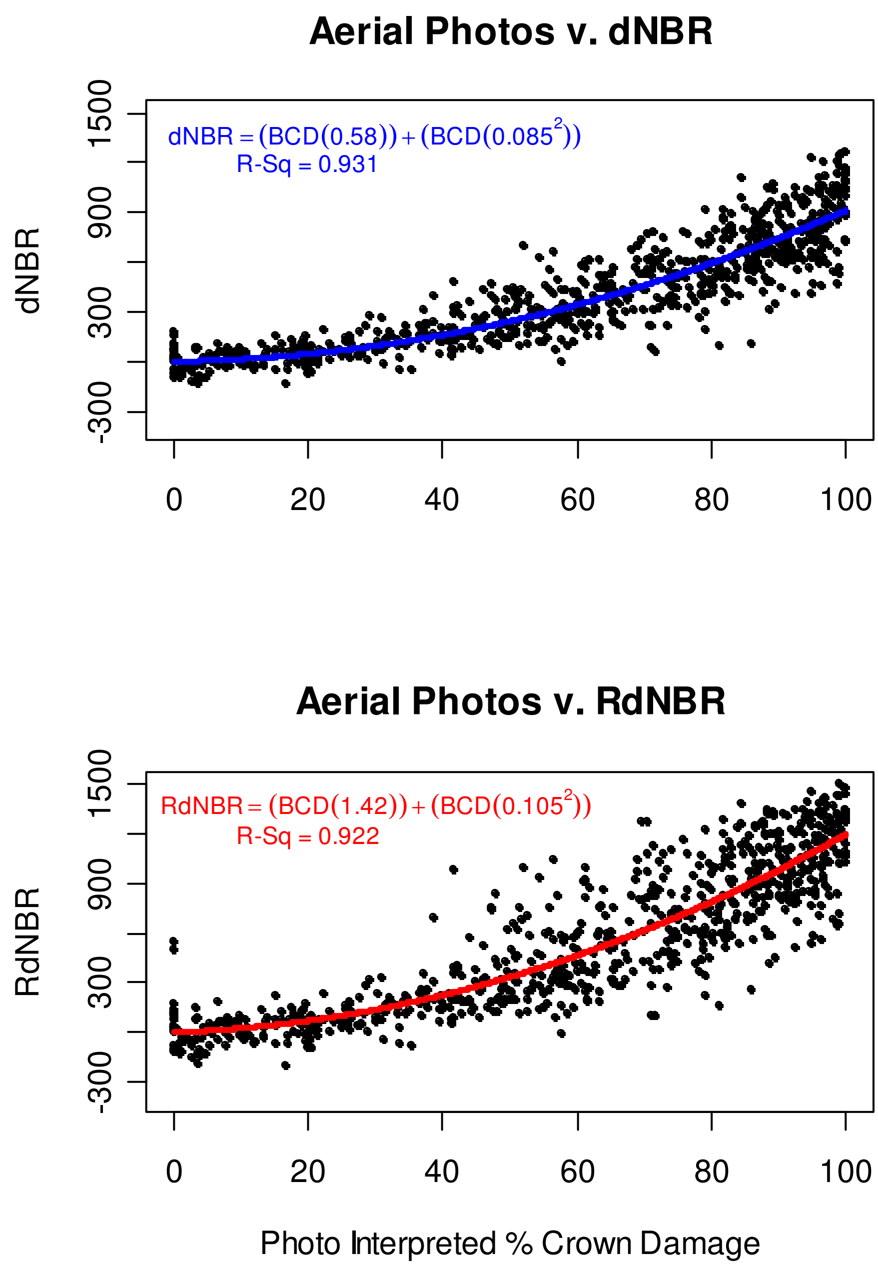

Figure A.1. Correspondence between aerial-photo interpreted Biscuit Fire crown damage and two Landsat-derived metrics of burn severity: the difference normalized burn ratio (dNBR) and the relativized differenced normalized burn ratio (RdNBR). BCD = Biscuit Crown Damage from photos