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The effect of prescribed fire on sugar pine mortality in Sequoia and Kings Canyon National Parks

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The effect of prescribed fire on sugar pine mortality in Sequoia and Kings Canyon National Parks

By

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A dissertation submitted in partial satisfaction of the requirements for the degree of Doctor of Philosophy in Environmental Science, Policy, and Management in the Graduate Division of the University of California, Berkeley

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Abstract

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Fire is one of the main agents controlling composition, structure, and function of mixed conifer forests in the Sierra Nevada. Over the past century, there has been a dramatic change in the role of fire in these forests as fire regimes shifted from low intensity, frequent fires prior to European settlement in the region, to extended periods of fire exclusion due to a policy of active fire suppression. This has led to many unintended consequences such as increased stand density, shifts in species composition towards more shade tolerant species, and an increase in fire severity and extent due to increased fuels and changing climate. In the past several decades, prescribed fire has become one of the most important tools for forest restoration and management, yet its effects are still not fully understood. This uncertainty is especially true for sugar pine, a species that is being affected not only by changing forest conditions, but also by the introduced pathogen white pine blister rust. These multiple factors, and potential interactions among them, have led to population declines of sugar pine in some areas and have raised concerns about potentially elevated rates of mortality following prescribed fire. The goal of this research was to better understand what processes control post-fire mortality of sugar pine, use this information to produce more accurate predictive models of post-fire mortality, and evaluate the effectiveness of simple management actions that could be used to ameliorate the risk of mortality following fire.

The first chapter serves as an introduction to the mixed conifer ecosystem and sugar pine in particular. It discusses the main factors that control mortality following fire and highlights some of the main findings of this research. In the second chapter, structural equation modeling (SEM) was used to assess the direct and indirect effects of multiple variables related to tree health, beetle activity, blister rust, and fire effects and their relative importance in controlling sugar pine mortality following prescribed fire in Sequoia and Kings Canyon National Parks. Multiple factors are known to influence mortality following fire including fuel loads, fire intensity, beetle activity, and tree size, yet little is known about how these factors interact to control post-fire mortality. A total of 436 sugar pine were measured within three separate prescribed fires. SEM was used to evaluate a network of causal relationships between factors that affect post-fire mortality of sugar pine and to assess both direct and indirect effects. Several factors were found to significantly influence post-fire mortality, with crown volume scorch, diameter at breast height (dbh), and post-fire beetle activity showing the strongest effects; though the magnitude of these effects differed among sites. Other factors such as blister rust infection and pre-fire beetle activity had little impact on post-fire mortality. A causal model was developed that considered both indirect and direct effects of multiple factors associated with post-fire mortality while
demonstrating the variability in the relative strength of these causal relationships based on specific site attributes. This model can be used in forest management to provide a clear understanding of how fire effects interrelate with multiple processes to control post-fire sugar pine mortality.

The third chapter examined whether the inclusion of pre-fire tree health (based on tree ring records) in models looking at post-fire sugar pine mortality improved model fit over models based on measures of fire effects alone. This study was conducted within an old-growth mixed conifer forest in Sequoia National Park that had been prescribed burned during 2001 or 2002. Fire effects measured by percent crown volume scorched and stem char height, and pre-fire tree health measured by multiple indices of growth calculated from tree cores and measures of crown health were assessed for 105 sugar pine. Health status (live or dead) was observed prior to the fire, immediately post-fire and five years post-fire. Logistic regression models were used to evaluate the effects of fire and pre-fire tree health on post-fire mortality. Models based only on tree size and fire effects were compared to models that included fire effects and measures of pre-fire tree health using corrected Akaike Information Criterion (AICc). Five years following fire, the model that best predicted mortality included dbh, crown volume scorch, 30 year growth trend, and count of sharp declines over a 30 year period. The inclusion of long-term measures of growth markedly improved model fit compared to models based only on fire effects (ΔAICc = 26.4). However, immediately after fire, models that included measures of pre-fire tree health resulted in only marginal improvements over models based only on measures of fire effects (ΔAICc = 2.1). These results imply that multiple processes, in addition to fire, are functioning to influence delayed mortality and that the inclusion of measures of tree health can provide more accurate predictions of post-fire mortality.

Finally, chapter four examined whether raking away duff and litter from the base of the stem can be used as an effective means of reducing sugar pine mortality following prescribed fire. This study was conducted in three prescribed fires in Sequoia and Kings Canyon National Parks and included 457 trees, half of which were raked to mineral soil to 0.5 m away from the stem. Fire effects were assessed and tree mortality was followed for three years after prescribed fires. Overall, raking had no detectable effect on mortality as raked trees averaged 30 % mortality compared to 36 % for unraked trees. There was a significant interaction, however, between raking and average pre-treatment forest floor fuel depth: The predicted probability of survival of a 50 cm dbh tree was 0.94 vs. 0.96 when average pre-treatment fuel depth was 0 cm for a raked and unraked tree, respectively. When average pre-treatment forest floor fuel depth was 30 cm, the predicted probability of survival for a raked 50 cm tree was 0.60 compared to only 0.07 for an unraked tree. Raking did not affect mortality when fire intensity, measured as percent crown volume scorch, was very low (0 % scorch) or very high (>80 % scorch), but the raking treatment significantly increased the proportion of trees that survived by 9.6 % for trees that burned under moderate fire intensity (1 % to 80 % scorch). Raking significantly reduced the likelihood of bole charring and bark beetle activity three years post-fire. This implies that raking can be an effective management tool to reduce tree mortality following prescribed fire under specific fuel and burning conditions.
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Chapter 1

Introduction

Jonathan C. B. Nesmith

Sugar pine (Pinus lambertiana Douglas) is regarded by many as one of the most beautiful of all pines and is renowned for its large cones, tremendous height, and majestic stature. It was first recognized as a new species in 1826 by David Douglas who went on to describe it as “the most princely of the genus and probably the finest specimen of the American vegetation”. John Muir said it was “the noblest of the pines yet discovered, surpassing all others not merely in size but also in kingly beauty and majesty”. Needless to say, there has been a revenant attitude towards sugar pine from its earliest discovery that has continued to today. Efforts to ensure its protection and conservation began as early as the 1930’s (Freemen 1931) due to threats from logging, grazing, wildfire, and the newly introduced pathogen white pine blister rust (Cronartium ribicola J.C. Fisch. ex Raben) (hereafter referred to as blister rust). Today, sugar pine continues to be threatened by a host of interacting stressors including blister rust, logging, climate change, and altered fire regimes, and there is growing concern that these compounding stressors may lead to significant population declines (van Mantgem et al. 2004, Tomback and Achuff 2010). In this Chapter, I will discuss the current state of sugar pine within the mixed conifer forest ecosystem in the Sierra Nevada. I will focus on the role of disturbance and how past and present management choices have influenced current forest conditions. I will then outline current management options for sugar pine protection and restoration, focusing primarily on prescribed fire and the benefits and potential concerns of its use. Finally, I will summarize the main results of my research and discuss their implications for management of sugar pine in the mixed conifer ecosystem.

Sugar pine can be found as far north as the Cascade range in Oregon to as far south as Baja California, Mexico, with the majority of its range occurring within the mixed conifer forest of the Sierra Nevada (Kinloch and Achuff 2010). Its elevation range spans from of roughly 335 to 3,000 m and it is most often found on mesic sites with deep soils (Kinloch and Scheuner 1990). Sugar pine is often found in association with several other tree species including white fir (Abies concolor [Gord. & Glend.] Lindl. Ex Hildebr.), ponderosa pine (Pinus ponderosa C. Lawson), incense cedar (Calocedrus decurrens [Torr.] Florin), and Douglas-fir (Pseudotsuga menziesii [Mirb.] Franco) (Evans et al. 2011). Other common species include Jeffrey pine (Pinus jeffreyi Balf.), giant sequoia (Sequoiadendron gigantium [Lindl.] J. Buchholtz), canyon live oak (Quercus chrysolepis Liebm.), and black oak (Quercus kelloggii Newberry). While sugar pine is a common species within the mixed conifer forest, it is never found in pure stands (Rundel et al. 1977). Sugar pine is the tallest of all pines, reaching nearly 60 m in height. It is also well known for its large cones and characteristic branching structure.

The main driver of forest composition and structure in Sierran mixed conifer forests has been fire (McKelvey et al. 1996, Skinner and Chang 1996). Historic fire return intervals ranged between 5 and 17 years in the northern Sierra Nevada (Beaty and Taylor 2008). This was due in part to an extensive history of Native American burning dating back thousands of years in this region (Anderson 2005). In addition, Sierran mixed conifer forests are found in an area with a Mediterranean climate of wet winters and dry summers that has strongly influenced fire
frequency and severity in this forest type (Gill and Taylor 2009). The main effect of fire in this region has been the patchy reduction of fuels that has created the multi-aged, heterogeneous stand structure exemplified by this forest type (Skinner and Chang 1996). Mixed conifer forests often display a clustered spatial pattern with a mixture of canopy gaps and dense foliage (North et al. 2007, Stephens et al. 2007). This clumped spatial pattern also can serve as a key factor in forest resiliency following fire (Stephens et al. 2008) as canopy gaps reduce the chance of sustained crown fires and have also been shown to be important for regeneration of pine species, including sugar pine (Gray et al. 2005). Mixed conifer forests, however, have become more homogeneous in stand structure since Euro-American settlement at a landscape scale largely due to fire suppression (Beaty and Taylor 2001, Evans et al. 2011). Fire also serves to stimulate new growth and creates conditions highly favorable for germination of new seedlings by removing duff and litter layers, removing competition from shrubs, and freeing up nutrients. This is especially important for sugar pine due to its relative shade intolerance, as well as other species common to mixed conifer forests such as ponderosa pine and giant sequoia. The management policy of fire suppression over the past century has created forest conditions that are quite different from what the first Euro-American settlers would have encountered (van Wagtendonk 2007). Fire suppression has led to increased fuel loads (Weatherspoon and Skinner 1995, Agee and Skinner 2005), which in turn has led to more severe, larger fires (Miller et al. 2009). Sugar pine may be particularly sensitive to an increase in fuel loads and fire severity as large trees tend to accumulate large duff and litter layers at their base, which can lead to high mortality rates even after relatively low intensity fire (Maloney et al. 2008).

An additional effect of fire is it often increases beetle activity, leading to additional post-fire mortality. In some cases, the proximate cause of much of the mortality that is observed following fire is from beetle attacks, not from the fire itself (Fettig et al. 2010). The main species of beetles that attack sugar pine are mountain pine beetle (Dendroctonus ponderosae), red turpentine beetle (D. valens), and various species of engraver beetles (Ipps ssp.). In addition to heightened beetle activity following fire (Breece et al. 2008, Maloney et al. 2008), beetles have become an increasingly acute concern because population sizes and dynamics have been changing due to climate change leading to epidemic outbreaks in some areas (Breshears et al. 2005). For sugar pine, beetles can pose a major threat to trees that have already been weakened by either fire damage, blister rust, or drought and understanding how beetle activity interacts with these other factors is an important objective to better understand post-fire mortality.

Another important disturbance in the mixed conifer ecosystem, particularly for sugar pine, has been logging. The California gold rush of 1849 brought a huge influx of people into the Sierra Nevada and sugar pine was highly valued for its strait trunks and high quality wood. From 1848 to 1989 almost 24 billion board feet of sugar pine lumber was produced from this region (Cermak 1996). Many of the oldest and largest trees were cut during this time period. In addition, logging changed forest structure at a landscape level due to clear-cutting of large areas leading to relatively homogeneous stands of same even-aged trees (McKelvey and Johnston 1992). Logging also changed species composition by removing only high value species such as sugar pine from some areas. Today, logging practices do not pose a major threat to sugar pine in a majority of its range. The main impact of active logging on sugar pine populations is the planting of seedlings that have been bred with some genetic resistance to blister rust following harvesting (Dulitz et al. 1996).

A relatively new stressor of sugar pine is blister rust. Blister rust has become established throughout much of the range of sugar pine, despite major efforts of eradication and control.
during the early and mid 1900’s (Tomback and Achuff 2010). Rusts are one of the most ecologically and economically important types of fungal diseases (Smith 1970) and blister rust is one of the most devastating of these rusts. This fungus creates large cankers on species of white pines that girdle the tree and can kill it within a number of years. Blister rust was first introduced into the United States in the early 1900’s. Since then, it has spread and reached epidemic levels in some parts of the Rocky Mountains (Kendall and Keane 2001, Smith et al. 2008). From its initial introduction in British Columbia, it quickly spread down the coast of the Pacific Northwest due to the wet, cool climate favorable for blister rust infection (Tomback and Achuff 2010). It continued to spread south through Oregon and California, but its rate of spread slowed due to the hotter, drier summers. While the conditions needed for blister rust spread are less frequent farther south, periods of favorable conditions do occur and blister rust has become established throughout most of the range of sugar pine. Years when conditions are particularly favorable can result in wave years where the spread of blister rust explodes and many new areas are infected. This occurred in both 1976 and 1983 (Kinloch and Dulitz 1990). These wave years have allowed blister rust to continue to spread to as far south as the Mexican border (Schwandt et al. 2010), and it has been detected in areas that were previously thought to be too dry to support the disease. The spread of blister rust has continued despite major eradication and control efforts in the 1950’s and 1960’s through Ribes eradication and direct control programs (Tomback and Achuff 2010). Today, breeding programs focused on genetic resistance are an integral part of many restoration efforts (Schwandt et al. 2010).

Due to rising costs of fire suppression and concerns for human safety and wildlife, current management of mixed conifer forest is often focused almost exclusively on fuel reduction to reduce the risk of high severity wildfire (North et al. 2009). This is particularly true in the wildland urban interface where the risk to human safety and property is highest. Two of the main tools used for forest restoration are thinning and prescribed fire. A reduction in ladder fuels is often accomplished via thinning while a reduction in surface fuels can be best achieved through prescribed fire (North et al. 2009). There has been extensive research done on the ecological effects of these two treatment options and whether our current management practices are effectively mimicking the historic role of fire within the mixed conifer ecosystem (Bartuszevige and Kennedy 2009, Schwilk et al. 2009). While management decisions are often constrained by fuel reduction objectives and fire safety restrictions, ecological restoration is often another primary goal to improve forest health and promote resilience in the face of changing climate and disturbance regimes (Keeley and Stephenson 2000, Stephens et al. 2010). Prescribed fire can promote restoration by reducing surface fuel loads and creating canopy gaps to stimulate regeneration (North et al. 2009). Especially if areas that are burned repeatedly, the reduction in fuel loads can also reduce the chance of high severity fire (Stephens and Moghaddas 2005). Other benefits of prescribed fire include increased nutrient cycling and growth of residual trees (Hurteau and North 2009) as well as altering stand density (van Mantgem et al. 2011) and species composition. There are several factors that can limit the use of prescribed fire including available resources such as fire personnel and funding, as well as air quality considerations and lack of public support for the use of prescribed fire. Other limiting factors include timing of when burns can be implemented and the limited size of area that can be treated. Despite these limitations, prescribed fire remains one of the most important management options for restoration within the mixed conifer ecosystem.

The goal of this research was to identify the factors associated with sugar pine mortality following prescribed fire and evaluate the effectiveness of local fuels removal from the base of
target trees to reduce probability of mortality. The impact of multiple factors on post-fire mortality including tree size, blister rust occurrence, pre-fire tree health, and pre- and post-fire beetle activity were considered in conjunction with fire effects using several different statistical methods and datasets. The main findings were that post-fire mortality was controlled by tree size, crown scorch, and post-fire beetle activity. However, several other factors including fuel loads and pre-fire tree health also significantly affected post-fire probability of mortality. This is the first study to show that tree health, measured either by crown vigor or growth rate, significantly impacted probability of survival following fire for sugar pine. Interestingly, blister rust was not found to be a significant factor affecting short-term (three years) mortality following fire. Removal of surface fuels around the base of the stem was also found to be an effective means of reducing probability of mortality following fire under certain conditions. These included areas with high fuel loads and in areas that would be expected to burn under moderate fire intensity.

This research has provided managers with important new information regarding the effects of prescribed fire on sugar pine mortality and potential management options for ameliorating some of the negative effects of fire. This research substantiated the concern some managers have expressed about using prescribed fire under conditions of high surface fuel loads, but it suggested that by removing surface fuels around the base of these trees, chances of mortality could be greatly reduced. In addition, managers need to be aware of stand health as healthy trees were less likely to die following fire. Blister rust occurrence, however, was not a major contributing factor to increased mortality following fire. The use of prescribed fire for the restoration of sugar pine should continue to be one of the main tools to ensure the continued success of this species, especially in the face of changing forest conditions due to climate change and human activity.

References


Chapter 2

Direct and indirect determinants of sugar pine mortality following prescribed fire in Sequoia and Kings Canyon National Parks

Jonathan C. B. Nesmith

Multiple factors are known to influence mortality following fire including fuel loads, fire intensity, beetle activity, and tree size, yet little is known about how these factors interact to control post-fire mortality. The objective of this study was to examine the direct and indirect effects of multiple variables related to tree health, beetle activity, blister rust, and fire effects to better assess their relative importance in controlling sugar pine mortality following prescribed fire. This study was conducted on 436 sugar pine, measured within three separate prescribed fires in Sequoia and Kings Canyon National Parks. Structural equation modeling was used to evaluate a network of causal relationships between factors that affect post-fire mortality of sugar pine to assess both direct and indirect effects. Several factors including tree diameter at breast height (dbh), crown vigor, duff and surface fuel depth, crown volume scorch, and post-fire beetle activity were found to significantly influence post-fire mortality. Of these variables, the strongest effects on mortality were due to crown volume scorch, dbh and post-fire beetle activity, though the magnitude of these effects differed among sites. Other factors including blister rust infection and pre-fire beetle activity had little impact on post-fire mortality. This study developed a causal model that considered both indirect and direct effects of multiple factors associated with post-fire mortality. It also demonstrated the variability in the relative strength of these causal relationships based on specific attributes. The use of exploratory structural equation modeling led to the discovery of several unanticipated causal relationships between variables which would not have been found using a more traditional analytical technique such as logistic regression. A clear understanding of how fire effects interact with multiple processes such as beetle activity and pathogen dynamics to control mortality following fire is essential to achieve management goals.

Introduction

Disturbance is one of the main drivers of forest composition, structure, and function (Dale et al. 2001, Bond et al. 2005). Some of the main agents of disturbance in the western U.S. are fire, beetles, and pathogens (Parker et al. 2006, Tomback and Achuff 2010). These factors interact with climate, site characteristics, and species composition to determine forest conditions and function (Dale et al. 2001). The influence of these different disturbances has changed over the last century due to human activities such as fire suppression, exotic species introductions, livestock grazing, timber harvest, and climate change (Dale et al. 2000, Syphard et al. 2007). Changes in forest composition, structure, and function have important implications for nutrient cycling, carbon sequestration, and spatial patterns of mortality.

The mixed conifer ecosystem of the western USA has historically experienced a mixed severity fire regime with varying degrees of understory fire and stand-replacing fire (Telfer 2000). Extended periods of fire exclusion over the past century have led to increased fuel loads
(Agee and Skinner 2005, Keifer et al. 2006), shifts in species composition towards more shade tolerant species, and shifts in size distribution towards smaller trees (Beaty and Taylor 2008). These changes have resulted in forests experiencing more frequent, higher severity fires (Westerling et al. 2006, Miller et al. 2009). Fire suppression has also affected forest health by altering competition for resources and increasing forest susceptibility to beetle and pathogen outbreaks (Harvey 1994, Parker et al. 2006). These changes in forest health, along with warming temperatures and other factors have increased beetle activity in many areas leading to extreme outbreaks and landscape scale mortality events (Raffa et al. 2008).

Several introduced pathogens, including the exotic pathogen white pine blister rust (Cronartium ribicola J.C. Fisch. ex Rabenh) (hereafter blister rust) have altered forest composition and function throughout the west (Tomback and Achuff 2010). Blister rust was first introduced to North America in the early 1900’s via nursery stock and has spread throughout the western USA. The factors that control the composition, structure, and function of Sierran mixed conifer forests, including blister rust, fire, and beetles are all influenced by climate, which is expected to continue to change rapidly over the next century (IPCC 2007, Ogden and Innes 2007). Therefore, understanding of how these processes interact is critical for projecting future forest conditions.

One species in particular that has been affected by changing forest conditions, climate, and disturbance regimes is sugar pine (Pinus lambertiana Douglas), which is experiencing elevated mortality and population declines in some areas (van Mantgem et al. 2004). Sugar pine is an important species in Sierran mixed conifer forests and provides both ecological and societal value (Kinloch and Scheuner 1990, Tomback and Achuff 2010). Sugar pine mortality has been accelerated by several factors including bark beetle activity and blister rust (Tomback and Achuff 2010). Blister rust and bark beetles in particular have been recognized as important agents that affect the health of sugar pine, yet little work has been done to investigate how these factors interact with prescribed fire to control post-fire mortality. Bark beetles are often the proximate cause of mortality in trees infected by blister rust (Nathan Stephenson, unpublished data) as well as in trees that have been damaged by fire (Fettig et al. 2010). The possibility exists that infected trees may be more susceptible to fire-caused damage via increased bark beetle activity following prescribed fire. The combination of these factors could contribute to local extinctions, especially in the face of changing fire regimes (Ferrell 1996, van Mantgem et al. 2004). The effect of prescribed fire on sugar pine in the Sierra Nevada has therefore become an important issue among fire management professionals and the general public (Kinloch et al. 1996, van Mantgem et al. 2004). This issue is particularly important in the Sierra Nevada where prescribed fire is considered one of the primary tools for forest management and restoration (Keeley and Stephenson 2000, Schmidt et al. 2006, North et al. 2009). A better understanding of the cumulative influence (both direct and indirect effects) of these factors on sugar pine health, in combination with estimates of fire damage can increase our ability to accurately predict mortality following prescribed fire (Parker et al. 2006). The objective of this study was to examine the direct and indirect effects of multiple variables related to tree health, beetle activity, blister rust, and fire effects to better assess their relative importance in controlling sugar pine mortality following prescribed fire.

One tool for evaluating a network of multiple variables to assess both direct and indirect causal relationships is structural equation modeling (SEM). SEM uses a multiequational framework to simultaneously test causal pathways among multiple interacting factors (Bollen 1989, Grace and Keeley 2006). This approach has many advantages over other statistical
approaches such as logistic regression in that it allows for the partitioning of direct and indirect effects of multiple variables and better assess the relative importance of each variable to determine which causal pathways are most influential (Grace and Keeley 2006). This approach has been gaining in popularity in the ecological literature and has proven to be an effective means of evaluating complex relationships among interacting factors (Arhonditsis et al. 2006, Grace and Keeley 2006, Youngblood et al. 2009). However, SEM has not been used to understand patterns of tree mortality, which seem ideally suited for this approach due to the multiple interacting factors that are often involved (Franklin et al. 1987). Therefore, in this study SEM was used to examine the causal mechanisms that control the important ecological process of tree mortality.

The first step in any SEM analysis is to develop a conceptual model based on pre-existing knowledge of the system being studied (Figure 2.1). This model is based on expected causal relationships among factors affecting post-fire mortality of sugar pine including fire effects, environmental variables, and tree health. Factors that have been shown to affect post-fire mortality include tree size (Regelbrugge and Conrad 1993), fire effects such as crown volume scorch and fuel consumption (Stephens and Finney 2002), post-fire beetle activity (Breece et al. 2008), and factors related to pre-fire tree health such as radial growth rate (van Mantgem et al. 2003). Fire effects are also expected to be influenced by fuel loads, tree size, pre-fire tree health, and localized environmental conditions such as slope (Weise and Biging 1997). Post-fire beetle activity is expected to be affected by tree size, pre-fire tree health, and fire effects (Bradley and Tueller 2001).

Figure 2.1. Conceptual model describing causal relationships between tree size, tree health, site conditions, fire effects, and beetle activity to post-fire mortality of sugar pine.
SEM analysis can be confirmatory or exploratory in nature depending on the goal of the researcher. In a confirmatory approach, a structural model relating the observed or latent variables to each other through causal pathways is proposed. A latent variable is an unmeasured variable that represents the underlying cause, whose value is estimated from other measured variables (e.g. intelligence measured by IQ) (Grace 2006). The fit of this model is then evaluated based on how well the implied covariance structure of the model fits the observed covariance matrix of the data (Grace 2006). The proposed model is then either accepted or rejected based on a formal hypothesis test of model fit such as a chi-square test (Grace 2006). Unlike in most standard statistical frameworks, the goal of SEM is the acceptance of the null hypothesis. This indicates that there is no significant difference between the proposed model’s covariance structure and that implied by the data (Arhonditsis et al. 2006).

The alternative exploratory approach is to modify the initial proposed tentative model based on differences between the implied and observed covariance matrices when the causal relations among variables are less certain. In this approach, differences in the implied covariance structure and the observed covariance matrix can be used to lead to the discovery of overlooked causal relationships or an alteration of one’s existing theory of how the variables interact (Grace 2006). If the researcher chooses to use an exploratory approach for model evaluation, however, then the results must be viewed as tentative and not as a rigorous hypothesis test as with the confirmatory approach and the new model should be evaluated in subsequent studies (McCune and Grace 2002). One advantage of an exploratory approach is that it can often lead to unexpected findings and offers the potential to rapidly grow current ecological theory. In this study, an exploratory approach was used because while many of the causal relationships among mortality and pre-fire tree health, beetle activity, blister rust occurrence, and fire effects have been well documented, they have rarely been investigated simultaneously to account for interactions and indirect affects between them (but see Youngblood et al. 2009). Therefore, modifications to the initial proposed model were planned based on the results from the analysis of the initial proposed model.

**Methods**

**Study Sites**

This study was conducted in an old-growth mixed conifer forest within Sequoia and Kings Canyon National Parks, California, USA. The area experiences a Mediterranean climate of wet winters and dry summers, averaging 1400 mm of annual precipitation (Stephenson 1988, van Mantgem et al. 2006). Soils are derived from decomposed granite and are predominantly coarse loams (Huntington and Akeson 1987). Mean daily temperatures range between 0 ºC during the winter to 18 ºC during the summer. Elevation of the sites ranged between 1800 m to 2300 m. Prior to European settlement, the area experienced frequent low severity fires, with fire return intervals of nearby areas estimated at seven years (Caprio and Swetnam 1995).

Three sites (Cabin Creek, Redwood Canyon, and Wall Spring) were prescribed burned during the summer or fall of 2006 or 2007. Cabin Creek (178 ha) is located in Sequoia National Park (36°37' N, 118°50' W) and was burned 8-10 November 2006 using aerial and ground ignitions, resulting in a combination of heading and backing fires. The area is primarily southwest facing with a mixture of white fir (*Abies concolor* [Gord. & Glend.] Lindl. Ex Hildebr.), red fir (*Abies magnifica* A. Murray), Jeffrey pine (*Pinus jeffreyi* Balf.), sugar pine, and California black oak (*Quercus kelloggii* Newberry). The site had not been burned in over 60
years. At the start of the burn, fuel moisture was high and relative humidity was 33 % with a maximum daily temperature of 18 °C. Snow was present on the ground in small amounts in some of the more shaded areas of the site from a storm that occurred a few days prior to the burn, leading to a relatively low intensity burn.

Redwood Canyon (250 ha) is located in Kings Canyon National Park (36°42’ N, 118°55’ W) and was prescribed burned 5-9 July 2006 using aerial and ground ignitions, resulting in a combination of heading and backing fires. The area is primarily southwest facing with a mix of giant sequoia (Sequoiadendron gigantium [Lindl.] J. Buchholz), white fir, sugar pine, incense cedar (Calocedrus decurrens [Torr.] Florin), ponderosa pine (Pinus ponderosa C. Lawson), and California black oak. The site had last burned in 1970 in a prescribed fire (Kilgore 1973). At the start of the burn, fuel moisture of 1000 h fuels ranged from 16 % to 31 % and relative humidity was between 26 % and 75 % with a maximum daily temperature of 27°C.

Wall Spring (71 ha) is located in Sequoia National Park (36°33’ N, 118°46’ W) and was prescribed burned 30 September-2 October 2007 using ground ignitions, resulting in a combination of heading and backing fires. The area is primarily west facing with a mix of white fir, giant sequoia, sugar pine, incense cedar, red fir, ponderosa pine, and California black oak. The site had not burned in over 100 years (Caprio and Swetman 1995). At the start of the burn, fuel moisture of 1000 h fuels ranged between 10 % and 40 % with relative humidity between 22 % and 44 % and a maximum daily temperature of 20°C.

Fire severity differed substantially among sites due in part to varied weather patterns during the burns, varied fuel loads, and stand structure (Nesmith et al. 2010). On average, Cabin Creek experienced the lowest fire intensities as crown torching was isolated to small pockets of trees, usually in areas dominated by white fir with low sugar pine density. Wall Spring displayed a more patchy distribution of surface burning and crown torching with larger and more frequent patches of high intensity fire. Redwood Canyon displayed the highest average fire intensities as large patches of trees with completely scorched crowns occurred within the site. As is often the case with fire in the mixed conifer forest, however, fire severity varied more within sites than between them (van Mantgem and Schwilk 2009, Nesmith et al. 2011) as multiple areas that were unburned as well as areas that experienced complete overstory mortality could be found within each site (Nesmith et al. 2010).

Experimental Design

Twelve, ten, and seven one-hectare (100 m × 100 m) plots were randomly located within Cabin Creek, Redwood Canyon, and Wall Spring, respectively. The plots within Redwood Canyon were selected from an 84 ha area within the larger burn to exclude areas with low sugar pine density. Within each plot, all sugar pine ≥10 cm at breast height (1.37 m) were tagged and mapped prior to the burn and multiple measures of tree health, size, and fuels were recorded. The number of sugar pine per plot ranged between 1 and 14 at Cabin Creek, 3 and 29 at Redwood Canyon, and 15 and 55 at Wall Spring, with a total of 54, 147, and 235 sugar pine being measured at the three sites, respectively, for a total sample size of 436 trees. Measures of tree size included stem diameter at breast height (dbh), tree height, height to live crown, and canopy class. Tree height and height to live crown were measured using an Impulse® handheld laser rangefinder (Laser Technologies, St. Paul, Minnesota, USA). Several factors associated with pre-fire tree health were recorded including crown vigor, beetle activity, and blister rust infection status. Crown vigor was estimated by visual inspection and assigned a rating of 1 (healthy), 2 (minor signs of stress such as branch flagging, little recent growth, or sparse foliage),
3 (substantial signs of stress such as top dieback, branch flagging, or yellowing foliage), or 4 (major signs of stress where imminent mortality is likely such as a broken top, severe foliage loss, or dieback) based on a rating system developed by Salman and Bongberg (1942). Similar rating systems are commonly employed to assess tree health in other forest types (Mangold 1998, Innes 1993). A tree was considered alive if it still had any green foliage above 1.37 m. Beetle activity was recorded based on visual inspection of the stem. Beetle activity was assigned a rating from 0 (no activity) to 3 (extensive signs of beetle activity such as pitch tubes, frass, or visible galleries), though for analysis it was converted to a binary variable of present or absent. Blister rust infection status was evaluated by visual inspection of the main stem and branches and was based on the number of cankers and other symptoms of blister rust that were present. Trees were assigned a rating of 0 (no cankers present), 1 (1-3 branch cankers present), 2 (4-9 branch cankers present), 3 (10-25 branch cankers present), 4 (26+ branch cankers present), or 5 (canker present on main stem) based on a rating system used by Duriscoe and Duriscoe (2002) in Sequoia and Kings Canyon National Parks. For analysis, blister rust infection status was also converted to a binary variable of present or absent as the occurrence of trees with a rating of 2, 3 or 4 was very low.

Local pre-fire fuel depth was recorded for each tree by measuring the height of surface fuels (litter + downed wood) and duff depth at the base of the tree in the four cardinal directions approximately 30 cm from the stem. These measures were then averaged to produce average pre-fire surface fuel depth and pre-fire duff depth values for each tree. To reduce soil disturbance and the time required to collect the fuel data, pre-fire duff depth was censored at 30 cm. This resulted in average pre-fire duff depth being underestimated in 6% of all trees. In addition to fuel depth, slope, and aspect were also recorded for each tree using a clinometer and compass.

Post-fire measures of fire effects were recorded within six weeks of each burn and included measures of crown scorch, duff and litter consumption (hereafter referred to as fuel consumption), and stem char. Crown scorch was measured by maximum height of scorching using a laser rangefinder as well as by crown volume scorched based on a visual estimate. Stem char was measured by maximum stem char height and by the percentage of the circumference of the base of the stem that was charred. Fuel consumption was measured by placing a large steel railroad spike on the uphill side of each tree so that it was level with the top of the litter layer. Following fire, the length of the nail that was exposed was recorded. Lastly, health status (live or dead) and beetle activity of each tree was recorded immediately following fire, as well as one, two, and three years after fire.

Data Analysis

Many of the fire effects variables were highly correlated (e.g. $R^2 = 0.75$ for crown volume scorch and max crown scorch ht), therefore, for simplicity, only crown volume scorch and fuel consumption were used in the initial SEM model. These two variables were also selected because they are commonly used measures of crown scorch and stem damage, respectively (Stephens and Finney 2002). An alternative approach to the analysis would have been to use latent variables of crown or stem damage estimated by the observed measures of crown scorch or stem char. This approach was not used because the additional model complexity that is introduced by the use of latent variables was not necessary to address the main research questions of this study.
Crown volume scorched displayed a highly non-normal, bimodal distribution with a high percentage of trees having either 0% or 100% crown scorch. It was therefore converted to a categorical variable of no scorch (0% crown volume scorch), moderate scorch (1% - 80%), or high scorch (>80%). 80% scorch was chosen as the cut-off for the high scorch category as this is a value often cited as leading to mortality in conifers (McHugh and Kolb 2003, Fowler et al. 2010). Fuel consumption was also converted to a binary categorical variable of yes (fuel consumption > 0) or no (fuel consumption = 0) because of the high proportion of zero values in the data. These data transformations were necessary to meet assumptions about the shape of the distribution of residuals in SEM because both crown scorch and fuel consumption were included as endogenous variables in the initial proposed model.

Statistical Models

The initial structural equation model was guided by the conceptual model (Figure 2.1) and related three year post-fire mortality of sugar pine to several factors including dbh, slope, pre-fire surface fuel depth, pre-fire duff depth, blister rust occurrence, pre-fire beetle activity, crown vigor, crown volume scorch, fuel consumption, and one-year post-fire beetle activity (Figure 2.2). One-year post-fire beetle activity was used instead of immediate or two- or three-year post-fire beetle activity because this time was soon enough after fire that pitch tubes and other signs of beetle activity were most visible and long enough after fire that most fire-related beetle attacks had already occurred. Due to expected variation among the three sites, multi-group analysis was used (using site as the grouping variable) to account for interactions between individual sites and the model parameters, as well as to determine whether individual parameters in the model differed significantly among sites (Pugesek and Tomer 1996, Grace 2006). In multi-group analysis, a single model is fit to all groups simultaneously and individual parameters can be constrained to be equal across groups to test for significant differences among groups for that parameter. In addition, plot was used as a cluster variable to account for possible non-independence among trees due to spatial correlations. The cluster variable is treated as a random effect in SEM, just as is done in generalized linear mixed models (Graham 2008). Because of the nested structure of the data within the multi-group framework using categorical endogenous variables, parameter estimates were estimated by weighted least square parameter estimates (WLSMV) using a diagonal weight matrix and theta parameterization (Muthén and Muthén 2007). Using a WLSMV estimator allows for a combination of categorical and continuous response variables and allows for multi-group analysis (Muthén et al. 1997). Theta parameterization allows for the residual variances of observed categorical response variables to be parameters in the model. All SEM analyses were conducted using Mplus, version 5 (Muthén and Muthén 2007).

The fit of the initial proposed model (Figure 2.2) was evaluated based on several measures of model fit including chi-square goodness of fit test, root mean square error of approximation (RMSEA), and comparative fit index (CFI). A chi-square value >0.05 indicates adequate model fit (Barret 2007). Browne and Cudeck (1993) suggest RMSEA values <0.08 indicate satisfactory agreement between the data and proposed model. For CFI, a value close to one indicates good model fit and 0.95 is often cited as a cut-off for adequate model fit (Hu and Bentler 1999). Tests of model fit are based on the unstandardized path coefficients. This is because when the covariance matrices are analyzed, significance testing of parameters is based on the standard errors of the unstandardized coefficients (Grace 2006). Standardized path coefficients, however, are used to make direct comparisons between path coefficients that are measured on different
scales. Both unstandardized and standardized coefficients were calculated to assess the statistical significance of the parameters within the model as well as the relative importance of each variable in controlling post-fire mortality. Cumulative effects of each variable on post-fire mortality were also calculated to assess its overall importance in controlling post-fire mortality. This was calculated by summing all indirect and direct effects linking the variable of interest to post-fire health status. The partitioning of the overall effect into direct and indirect effects is one of the main advantages of using SEM compared to traditional analytical techniques and provides a more informative description of the effect of each variable on post-fire mortality.

Figure 2.2. Initial proposed structural equation model of the causal pathways linking multiple factors that affect post-fire mortality of sugar pine at Sequoia and Kings Canyon National Parks. Slope is the average slope at each tree, DBH is diameter at breast height, wpbr is a binary (yes or no) variable of blister rust infection status, Vigor is a categorical rating of crown vigor (1 = healthy, 2 = good, 3 = moderate, 4 = poor), BeetAct is a binary variable of pre-fire beetle activity, duff is average duff depth at the base of a tree, SurFuels is average height of litter and downed fuels at the base of a tree, DPinCat is a binary variable of litter and duff consumption at the base of each tree, VolScCat is a categorical variable of crown volume scorch (low = 0 %, moderate = 1-80 %, high = 81-100 %), Beet2 is a binary variable of one year post-fire beetle activity, and Status in post-fire health status (live or dead) three years following fire. Arrows represent causal pathways between parameters.
When using an exploratory approach in SEM, if model fit is found to be inadequate, the results from the initial model assessment can be used to develop a revised model based on modification indices and standard errors of the path coefficients (Grace 2006). However, any changes to the initial model should always be done in the context of relevant biological theory and not based solely on modification indices or residuals. Following the assessment of the initial model, a revised model was proposed that removed some of the initial parameters included in the model and added additional pathways and an analysis of the modified model was performed. Specific changes to the initial model structure were evaluated by assessing the corresponding change to overall model fit (Kline 2010) using a modified chi-squared difference test to account for the fact that the difference in chi-square values did not follow a chi-square distribution due to the use of the WLSMV estimator (Muthén and Muthén 2007).

Results

Pre-treatment conditions varied among sites as was expected with the largest differences being in tree size, beetle activity, and occurrence of blister rust (Table 2.1). Sugar pine at Wall Spring were smaller than at the other two sites with an average dbh of 34 cm compared to 59 cm at both Cabin Creek and Redwood Canyon. Trees at Cabin Creek grew under more open canopies compared to the denser forests of Redwood Canyon and Wall Spring, averaging a live crown ratio of 76 % compared to 63 and 62 %, respectively (Table 2.1). Average crown vigor was similar among sites, while pre-fire beetle activity was much more prevalent at Redwood Canyon (11 %) and Wall Spring (7 %) compared to Cabin Creek (2 %). Redwood Canyon displayed the highest occurrence of blister rust infection (28 %), while relatively little blister rust was found at Cabin Creek (17 %). Despite the widely different burn histories among sites, average pre-fire duff depth and pre-fire surface fuel depth were relatively similar, ranging between a total fuel depth (surface fuels + duff) of 9.4 cm at Redwood Canyon to 11.1 cm at Wall Spring (Table 2.1). Slope was variable within sites, but average slope was relatively similar among sites, with Wall Spring being the steepest followed by Cabin Creek and Redwood Canyon (Table 2.1).

The mortality rate of sugar pine three years following prescribed fire was 11 %, 45 %, and 36 % at Cabin Creek, Redwood Canyon, and Wall Spring, respectively, for an overall mortality rate of 36 % (Table 2.2). Fire effects differed substantially among sites as average crown volume scorched was 16 %, 50 %, and 26 % at Cabin Creek, Redwood Canyon, and Wall Spring, respectively. Similar differences were recorded for relative stem char height and basal charring with the largest average charring occurring at Redwood Canyon and the smallest at Cabin Creek (Table 2.2). Fuel consumption was highest at Wall Spring, averaging 12.0 cm compared to 7.1 cm at Cabin Creek. Beetle activity one year post-fire increased dramatically compared to pre-fire beetle activity at all three sites and was highest at Redwood Canyon (52 %), followed by Cabin Creek (28 %) and Wall Spring (24 %).
Table 2.1. Average pre-fire conditions of sugar pine at three prescribed fires in Sequoia and Kings Canyon National Parks. Trees is the number of trees sampled at each site, LCR is live crown ratio, Crown vigor is the average rating within a site (values range from 1 (healthy) to 4 (near death)), Pre-fire beetles is the number of trees that displayed signs of beetle activity, surface fuels is the average depth of litter and downed fuels at the base of a tree. Values in parentheses are standard deviations.

<table>
<thead>
<tr>
<th>Site</th>
<th>Cabin Creek</th>
<th>Redwood Canyon</th>
<th>Wall Spring</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trees</td>
<td>54</td>
<td>147</td>
<td>235</td>
</tr>
<tr>
<td>Slope (%)</td>
<td>20 (10)</td>
<td>15 (6)</td>
<td>22 (10)</td>
</tr>
<tr>
<td>Dbh (cm)</td>
<td>58 (46)</td>
<td>59 (40)</td>
<td>33 (27)</td>
</tr>
<tr>
<td>Height (m)</td>
<td>19.9 (14.1)</td>
<td>26.3 (15.6)</td>
<td>17.7 (13.4)</td>
</tr>
<tr>
<td>LCR (%)</td>
<td>78 (12)</td>
<td>64 (14)</td>
<td>62 (16)</td>
</tr>
<tr>
<td>Crown vigor</td>
<td>1.6 (0.6)</td>
<td>1.6 (0.9)</td>
<td>1.6 (0.9)</td>
</tr>
<tr>
<td>Pre-fire beetles</td>
<td>2</td>
<td>13</td>
<td>15</td>
</tr>
<tr>
<td>Blister rust</td>
<td>4</td>
<td>40</td>
<td>58</td>
</tr>
<tr>
<td>Surface Fuel Depth (cm)</td>
<td>5.1 (3.9)</td>
<td>5.7 (4.9)</td>
<td>6.5 (5.3)</td>
</tr>
<tr>
<td>Duff Depth (cm)</td>
<td>5.1 (5.0)</td>
<td>3.7 (2.5)</td>
<td>4.6 (3.4)</td>
</tr>
</tbody>
</table>

Table 2.2. Average Post-fire conditions of sugar pine at three prescribed fires in Sequoia and Kings Canyon National Parks. Dead trees is the number of trees that were dead three years following fire, Crown scorch is the percent crown volume that was scorched, Rel. max scorch ht. is the ratio of the maximum scorch height to tree height, Rel. max char ht. is the ratio of the maximum stem char height to tree height, Circ. Stem char is the percentage of the circumference of the base of the stem that was charred, fuel consumption is the height of litter and duff at the base of the tree consumed by the fire, and Post-fire beetles is the number of trees that displayed signs of beetle activity one year following fire. Values in parentheses are standard deviations.

<table>
<thead>
<tr>
<th>Site</th>
<th>Cabin Creek</th>
<th>Redwood Canyon</th>
<th>Wall Spring</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dead Trees</td>
<td>6</td>
<td>66</td>
<td>99</td>
</tr>
<tr>
<td>Crown scorch (%)</td>
<td>15 (28)</td>
<td>50 (41)</td>
<td>28 (36)</td>
</tr>
<tr>
<td>Rel. max scorch ht.</td>
<td>24.8 (35.0)</td>
<td>66.1 (38.2)</td>
<td>42.4 (40.7)</td>
</tr>
<tr>
<td>Rel. max char ht.</td>
<td>9.9 (17.0)</td>
<td>18.6 (18.8)</td>
<td>14.9 (19.4)</td>
</tr>
<tr>
<td>Circ. stem char (%)</td>
<td>46 (45)</td>
<td>64 (35)</td>
<td>69 (34)</td>
</tr>
<tr>
<td>Fuel consumption (cm)</td>
<td>7.1 (8.4)</td>
<td>9.0 (6.5)</td>
<td>12.0 (8.5)</td>
</tr>
<tr>
<td>Post-fire beetles</td>
<td>16</td>
<td>86</td>
<td>65</td>
</tr>
</tbody>
</table>

The fit of the initial proposed model (Figure 2.2) using multi-group SEM was poor (Chi-square $p = 0.025$, RMSEA = 0.098, CFI = 0.942) indicating significant differences in the covariance matrices of the proposed model and the data. There were several variables that did not show strong associations with post-fire mortality, either directly or mediated through other variables. These included slope, pre-fire beetle activity, and fuel consumption. Blister rust occurrence was also found to not be a significant variable relating to post-fire mortality and was not included in the final model. The effects of duff and surface fuels on post-fire mortality were better explained through indirect causal pathways with crown volume scorch and post-fire beetle activity rather than mediated through fuel consumption. Therefore, fuel consumption was also
removed from the model. The exclusion of these variables improved model fit, implying that these parameters were not necessary to explain post-fire mortality. Once these changes to the model had been made, model fit improved dramatically (Chi-square $p = 0.1424$, RMSEA = 0.058, CFI = 0.978).

The revised model is shown for each site in Figures 2.3, 2.4 and 2.5. As is often the case in ecological data, substantial differences among sites were apparent, leading to significant differences (chi square difference test $p < 0.05$) among multiple parameters as indicated by the multi-group analysis (Table 2.3). There were significant differences among sites in the magnitude of the effect of pre-fire surface fuel depth and crown volume scorch on post-fire beetle activity and the effect of pre-fire surface fuel depth on crown volume scorch. The effect of pre-fire surface fuel depth on crown volume scorch was not significant at Cabin Creek or Redwood Canyon, but was highly significant at Wall Spring, with increasing pre-fire surface fuel depth leading to increased crown scorch (Table 2.3, Figure 2.5). The effect of pre-fire surface fuel depth on post-fire beetle activity was not significant at Cabin Creek, but was significant at Redwood Canyon and Wall Spring as trees with higher pre-fire surface fuel depth were more likely to have increased beetle activity one year after fire (Figures 2.4 and 2.5). In addition, crown volume scorch significantly increased post-fire beetle activity at Cabin Creek and Redwood Canyon, but not at Wall Spring (Figures 2.3 and 2.4). There was a negative relationship between dbh and crown volume scorch at all three sites as a larger dbh resulted in decreased crown volume scorch with the largest effect occurring at Wall Spring, but the difference in effect size among sites was not significant at the 95 % confidence level (chi-square test $p = 0.0717$).
Figure 2.3. Structural equation model for post-fire sugar pine survivorship at Cabin Creek. The strength of each pathway is illustrated by the width of the line and significance of the pathway is shown by the style of the line. The numbers above each pathway are the standardized coefficients.
Figure 2.4. SEM model for post-fire sugar pine survivorship at Redwood Canyon. The strength of each pathway is illustrated by the width of the line and significance of the pathway is shown by the style of the line. The numbers above each pathway are the standardized coefficients.

The width of the line represents the relative importance of the pathway in the model based on the standard coefficients.

- Solid line: $p < 0.05$
- Dashed line: $p < 0.10$
- Dot-dash line: $p > 0.10$
In addition to differences in the magnitude of causal pathways among sites, the relative importance of the different parameters on post-fire survivorship differed as well. Post-fire survivorship at Cabin creek was dependent on crown volume scorch but was not directly influenced by any of the other parameters (Figure 2.3). The main effect of dbh on post-fire survivorship was an indirect effect mediated through crown volume scorch where larger trees received less scorch, which increased their probability of survival, though this effect was not significant at the 95% confidence level ($p = 0.053$). Post-fire beetle activity at Cabin Creek was affected by crown volume scorch and pre-fire duff depth, with higher scorch and a deeper duff layer leading to a higher probability of beetle activity one year after fire. However, post-fire beetle activity was not a significant factor in controlling post-fire mortality at Cabin Creek. Neither pre-fire surface fuel depth nor crown vigor significantly influenced crown volume scorch, post-fire beetle activity, or post-fire mortality at Cabin Creek (Figure 2.3). Overall, crown volume scorch had the largest cumulative (direct + indirect effects) effect on post-fire survivorship at Cabin Creek with an estimated cumulative effect of -0.372 (standardized coefficient) and was the only variable that significantly influenced post-fire survivorship at this site.

**Figure 2.5.** SEM model for post-fire sugar pine survivorship at Wall Spring. The strength of each pathway is illustrated by the width of the line and significance of the pathway is shown by the style of the line. The numbers above each pathway are the standardized coefficients. The width of the line represents the relative importance of the pathway in the model based on the standard coefficients.
Table 2.3. Unstandardized coefficient estimates for the final revised multigroup structural equation model of the causal pathways between factors that control post-fire mortality of sugar pine in Sequoia and Kings Canyon National Parks. Status is post-fire health status (live or dead) three years following fire, VolScCat is a categorical variable of crown volume scorch (low = 0 %, moderate = 1-80 %, high = 81-100 %), Vigor is a categorical rating of crown vigor (1 = healthy, 2 = good, 3 = moderate, 4 = poor), Beetles is a binary variable of one year post-fire beetle activity, SurFuels is average height of litter and downed fuels at the base of a tree, and Duff is average duff depth at the base of a tree. * indicates significant difference (chi-square significance test $p<0.05$) in coefficient estimates among sites based on multi-group SEM. The variables in bold (Status, Beetles, and VolScCat) indicate the dependent variables and the variables listed underneath them are the variables they are being regressed on in the SEM.

<table>
<thead>
<tr>
<th>Status</th>
<th>Cabin Creek</th>
<th>Redwood Canyon</th>
<th>Wall Spring</th>
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<tr>
<td></td>
<td>Estimate</td>
<td>S.E.</td>
<td>$p$-value</td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dbh</td>
<td>0.047</td>
<td>0.058</td>
<td>0.414</td>
</tr>
<tr>
<td>VolScCat</td>
<td>-1.198</td>
<td>0.59</td>
<td>0.042</td>
</tr>
<tr>
<td>Vigor</td>
<td>-0.061</td>
<td>1.403</td>
<td>0.965</td>
</tr>
<tr>
<td>Beetles</td>
<td>0.112</td>
<td>0.338</td>
<td>0.741</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Beetles</td>
<td></td>
<td></td>
<td></td>
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<td></td>
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Direct effects that were associated with post-fire survivorship at Redwood Canyon included crown volume scorch and dbh. Increasing dbh increased the probability of post-fire survivorship while increasing scorch decreased probability of survivorship (Figure 2.4). Dbh also indirectly affected post-fire survivorship as increasing dbh reduced crown volume scorch. No other variables besides dbh exerted any substantial influence on crown volume scorch at Redwood Canyon. Post-fire beetle activity was affected by crown volume scorch and pre-fire surface fuel depth, as increasing scorch and surface fuel depth resulted in increased probability of post-fire beetle activity. There was some evidence that post-fire beetle activity also decreased probability of post-fire survivorship, but this effect was only suggestive (p = 0.07). Neither crown vigor nor pre-fire duff depth significantly influenced crown scorch, post-fire beetle activity, or post-fire survivorship at Redwood Canyon (Figure 2.4). Overall, crown volume scorch had the largest cumulative effect (-0.790 standardized coefficient) on post-fire survivorship at Redwood Canyon, followed by dbh (0.399 standardized coefficient) and post-fire beetle activity (-0.324 standardized coefficient). The cumulative effects of the other variables on post-fire survivorship were not significant at this site.

Post-fire survivorship at Wall Spring was significantly influenced by all parameters in the model with the strongest effects attributed to crown volume scorch, post-fire beetle activity, and dbh. Increasing crown volume scorch and post-fire beetle activity both significantly decreased post-fire survivorship at Wall Spring (Figure 2.5). The main effect of dbh on post-fire survivorship was indirect, mediated through crown scorch, though the direct effect of dbh on post-fire survivorship was also significant as increasing dbh increased the probability of survival following fire (Figure 2.5). In addition to the direct effects of crown scorch, dbh, and post-fire beetle activity on post-fire survivorship, post-fire survivorship was also directly affected by pre-fire crown vigor as trees with a higher crown vigor rating (poorer health) were more likely to die following fire. Crown volume scorch was directly controlled by pre-fire surface fuel depth, dbh, and pre-fire crown vigor (Figure 2.5). Crown scorch was lower for trees with larger dbh and higher for trees with higher pre-fire surface fuel depth and higher crown vigor rating, with dbh having the largest relative effect. Post-fire beetle activity was affected by both pre-fire surface fuel depth and pre-fire duff depth with increasing fuel depths leading to increased probability of post-fire beetle activity. Post-fire beetle activity was not significantly affected by crown volume scorch at Wall Spring (p = 0.648) (Figure 2.5). The largest cumulative effect on post-fire survivorship was from crown volume scorch (-0.632 standardized coefficient), but dbh was also quite influential (0.617 standardized coefficient). Other variables that had significant cumulative effects on post-fire survivorship included post-fire beetle activity (-0.406 standardized coefficient) and pre-fire duff depth (-0.100 standardized coefficient). The cumulative effect of crown vigor and pre-fire surface fuel depth on post-fire survivorship was not significant at this site.

Discussion

Ecological systems are often controlled by multiple interacting processes which can be difficult to understand using traditional statistical approaches (Grace 2006). Through the use of SEM, I was able to identify how multiple processes influenced post-fire mortality of sugar pine and identify the main causal pathways driving post-fire mortality. The result was a model that reflected the main biological processes and causal mechanisms, some of which had not been anticipated at the beginning of the study. The analysis also indicated that some factors that had
initially been assumed to be important in driving post-fire mortality in this system, such as white pine blister rust, were not as influential as had been expected.

The main causes of post-fire mortality for sugar pine were post-fire beetle activity and crown scorch. These are the two most common agents of mortality following fire in many forest types across the western USA (Ryan and Reinhardt 1988, Schwilk et al. 2006, Breece et al. 2008). Tree size was also an important factor in controlling mortality, mainly due to its effect on crown scorch, as larger trees were less likely to be severely scorched during fire and therefore less likely to die (Stephens and Finney 2002, Kobziar et al. 2006). Tree dbh also had a less important, yet still significant direct effect on probability of mortality that was not mediated through crown volume scorch. This was likely due to the increased defense to fire provided by thicker bark afforded by larger size that directly reduced the damage to tree cambium from stem char, leading to a higher survival rate of larger trees (van Mantgem and Schwartz 2004, Kobziar et al. 2006). Crown scorch is regarded as a better predictor of mortality than stem char in most conifers (van Wagner 1973, Fowler and Seig 2004) and this was supported by this study as well. Other studies, however, have found that incorporating measures of stem char improve predictions of post-fire mortality (van Mantgem and Schwartz 2004, Keyser et al. 2006).

Fuel loads were also significant factors in controlling post-fire mortality, though their effects on mortality were indirect, mediated through either crown volume scorch or post-fire beetle activity. Increased pre-fire surface fuel depth led to higher crown scorch and increased post-fire beetle activity, which in turn, increased post-fire mortality. Increased pre-fire duff depth led to higher rates of beetle activity, which resulted in a higher probability of mortality. The link between pre-fire duff depth and beetle activity is likely a reflection on the fact that beetles tend to attack trees whose stems have been damaged by fire, weakening their defenses and providing easier access to the cambium (Hood and Bentz 2007, Nesmith et al. 2010). Others have shown that deep duff layers can lead to severe stem injury due to prolonged heating at the base of the tree (Varner et al. 2009, Hood 2010), though no significant direct link between pre-fire duff depth and probability of mortality was found in this study. While fuel consumption was not included in the final model, it is possible that stem injury was better correlated with pre-fire duff depth than fuel consumption and the direct causal pathway between duff depth and post-fire beetle activity is a reflection of that.

Pre-fire crown vigor affected mortality both directly, as trees with poorer crown health were significantly more likely to die following fire than trees with healthy crowns, as well as indirectly, as trees with low crown vigor were more likely to display higher levels of crown scorch following fire. The fact that tree health can influence probability of mortality has been documented by others (van Mantgem et al. 2003, Perrakis et al. 2011, Kolb et al. 2007), but this is the first study to explore the causal mechanisms of this effect, showing that pre-fire tree health not only has a direct effect on probability of survival, but also influences the effect of fire damage, leading to higher levels of crown scorch from the fire. One possible explanation of the link between pre-fire crown vigor and crown scorch is that trees already showing signs of stress are more sensitive to heating and scorch more easily than trees with healthy foliage, though this is a novel hypothesis that has yet to be tested. However, drought stressed trees have been shown to be more sensitive to the effects of fire (Bigler et al. 2005, Bond et al. 2009). An alternative hypothesis is that the trees that displayed signs of stress were more often located in denser areas of the forest and were therefore more likely to experience higher amounts of crown scorch due to the higher local stand density. Because spatial data was only collected for sugar pine and not the surrounding trees, this hypothesis was not able to be tested.
Several of the causal relationships proposed in the initial model were not supported by the data and the revised model was much simpler than the initial proposed model (Figures 2.3-2.5). The revised model was dominated by a few very strong causal relationships between dbh, crown scorch, and post-fire beetle activity. Other research involving mechanisms that control ecological systems have found that a small number of processes often drive community composition and function in terrestrial ecosystems, though many other processes are often present (Holling 1992, Ellison et al. 2005). The results from this study support this theory. Some factors that were expected to play important roles in controlling post-fire mortality that were not included in the final model were pre-fire beetle activity, blister rust occurrence, fuel consumption, and slope. While pre-fire beetle activity was positively correlated to post-fire beetle activity, it did not have a significant direct effect on post-fire mortality and post-fire beetle activity was adequately explained by crown scorch and pre-fire fuel depth. Therefore, the inclusion of pre-fire beetle activity did not significantly improve overall model fit. Other studies, however, have found that areas that experienced high levels of beetle activity prior to fire were more likely to burn at higher severity (Lynch et al. 2006, Jenkins et al. 2008, but see Kulakowsky et al. 2003). In this study, pre-fire beetle activity was relatively minor, which may explain why it did not significantly impact post-fire mortality.

Blister rust was found at low to moderate abundance at the three sites (7 %, 27 %, and 25 %, at Cabin Creek, Redwood Canyon, and Wall Spring, respectively), with most detections being limited to one or two small cankers located on the lower branches. Blister rust occurrence did not have a significant effect on post-fire mortality, either directly, or indirectly mediated through other variables. There has been little work done on the effect of blister rust infection on fire severity or the effect of fire on blister rust occurrence and spread. However, Nesmith et al. (2010) found that the presence of blister rust increased the probability of stem charring, but did not significantly affect post-fire mortality. Blister rust infection is often associated with increased beetle activity (Tomback and Achuff 2010), but neither blister rust nor pre-fire beetle activity were abundant at the three study sites and there was not a significant association between blister rust infection and post-fire beetle activity. While blister rust has been linked to increased mortality rates in Sequoia National Park (van Mangem et al. 2004), there was no evidence that it affects short-term probability of fire-induced mortality.

Fuel consumption was highly variable, with many trees (15 %) remaining unburned around the base following fire, while fuel consumption depths up to 47 cm were recorded at other trees. The amount of fuel consumption was not an important predictor of mortality as its inclusion in the model did not significantly increase model fit. This is contrary to other studies which have found duff consumption at the base of a tree to be a significant factor in post-fire mortality (Stephens and Finney 2002, Varner et al. 2007, Hood 2010). In this study, mortality was better predicted by a direct pathway between pre-fire duff depth and mortality, indicating that our measure of fuel consumption was not as strongly correlated with tree mortality as pre-fire duff depth. This may be due to the fact that fuel consumption was transformed into a categorical variable due to the high variability of this measure during the burns. The lack of significance of this variable may also be due to including litter consumption as part of the total fuel consumption depth, which may have reduced the correlation with stem injury as prolonged heat is caused by duff consumption, while there is little contribution from the flash heating produced by the consumption of fine litter (Hood 2010).

Slope was not a significant predictor or crown scorch or post-fire mortality and its inclusion did not improve model fit. While slope has been shown to affect crown scorch via affects on
flame lengths (Weise and Biging 1996), this effect appeared to have been minor compared to fuel loads and other properties of the site such as fuel moisture and fire weather. While there was a positive correlation between slope and crown scorch, the inclusion of this causal pathway in the model did not significantly improve model fit and there was no evidence of any direct relationship between slope and post-fire mortality.

In this study the relative strength of several factors on post-fire mortality among sites was quite variable. Post-fire mortality was significantly influenced by post-fire beetle activity at Wall Spring, but it was not an important factor controlling post-fire mortality at Cabin Creek. This highlights the fact that many processes that influence post-fire mortality are controlled by specific attributes of a site such as stand structure, species composition, and the timing of the fire. In particular, post-fire beetle activity was highly variable by site, a common finding in areas that have been burned (Santoro et al. 2001, McHugh et al. 2003, Simard et al. 2008). The patchy nature of post-fire beetle activity led to significant differences in the magnitude of the effects of crown scorch and pre-fire surface fuel depth on post-fire beetle activity among sites. Volume scorch significantly increased beetle activity at Cabin Creek and Redwood Canyon, but was not a significant driver of post-fire beetle activity at Wall Spring. In addition, pre-fire surface fuel depth significantly increased beetle activity at Redwood Canyon and Wall Spring, but not at Cabin Creek.

The effect of pre-fire surface fuel depth on crown scorch also varied significantly among sites. Pre-fire surface fuel depth was not a significant factor controlling crown scorch at Cabin Creek and Redwood Canyon, but significantly increased crown scorch at Wall Spring. This may be due to the length of time between burns, as Wall Spring had not burned in >100 years and had the highest average pre-fire surface fuel depth of the three sites. While pre-fire surface fuel depth did not vary dramatically among sites, the composition and structure of these fuels were likely different among sites due to differences in disturbance history (Jenkins et al. 2008), and this was reflected in the significant difference in the effect of surface fuels on crown scorch among sites. For example, there was much less large woody debris at Cabin Creek compared to the other two sites, though this was not quantified during the study.

Despite large differences in fire history and the timing of burns among sites, the relationships among factors controlling post-fire mortality following fire were accurately explained by the same model. This helps substantiate the validity of the causal pathways within the model as it matched well with the data from a diverse set of site conditions. The model also captured strong differences among sites, primarily in the magnitude of the effect of post-fire beetle activity on post-fire mortality. By using SEM, additional causal pathways that had not been considered in the initial proposed model were also discovered. Specifically, the relationship between pre-fire tree crown vigor and crown volume scorch was found to be an important causal pathway. By using an exploratory approach, several new relationships between variables were discovered, suggesting several new, untested hypotheses. These causal relationships would not have been identified using a traditional logistic regression approach, and highlights one of the main advantages of an exploratory SEM approach. It is important to remember, however, that this model must be viewed as a tentative model that requires further validation as an exploratory approach was used to arrive at the final model structure.

**Conclusions**

Fire is one of the main tools used by managers for forest restoration. A clear understanding of how fire interacts with multiple processes such as beetle activity and pathogen dynamics to
control post-fire mortality is essential to achieve management goals (Parker et al. 2006). This is especially true for sugar pine, which is experiencing population declines in parts of its range due to multiple factors including fire suppression, climate change, timber harvest, and the introduced pathogen white pine blister rust (van Mantgem et al. 2004, Zeglen et al. 2010). In this study, several processes were found to influence post-fire mortality including pre-fire tree health and fuel loads, but the main drivers of post-fire mortality in this system were direct effects of the fire itself, especially crown scorch, and post-fire beetle activity. These factors and have been found to be important drivers of post-fire mortality in other forest types across the western U.S (Parker et al. 2006). Both crown scorch and post-fire beetle activity were strongly influenced by tree size, illustrating the importance of both direct and indirect causal mechanisms in controlling post-fire mortality.

Other factors including blister rust infection and pre-fire beetle activity were found to have little impact on post-fire mortality. One implication for managers is that the occurrence of blister rust within potential burn sites should not discourage the use of prescribed fire as blister rust infection did not significantly affect post-fire mortality of sugar pine. However, as blister rust occurrence was relatively low in the three burns used in this study this result should not be generalized to areas experiencing more severe outbreaks of the pathogen. In addition, in areas where post-fire beetle activity is a concern, managers should consider adjusting burn prescriptions to promote lower intensity fire as crown scorch significantly increased post-fire beetle activity in two of the three sites.

This study illustrated the variability in the importance of several factors across sites as site specific attributes such as disturbance history and beetle dynamics can lead to different factors driving post-fire mortality. However, the same model was able to adequately explain post-fire mortality across all three sites and demonstrated the universal importance of tree size and fire intensity in controlling post fire mortality. Structural equation modeling proved to be an effective tool for examining the direct and indirect effects of multiple interacting factors associated with post-fire mortality of sugar pine across multiple sites.

Acknowledgements

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References


Chapter 3

Using growth rate and fire damage to predict mortality of sugar pine following prescribed fire in Sequoia National Park

Jonathan C.B. Nesmith

Few studies have looked at how pre-fire tree health interacts with fire effects to control probability of mortality following fire. This is especially relevant for sugar pine due the introduction of the exotic pathogen white pine blister rust and altered forest conditions due to fire suppression that are contributing to population declines. By incorporating measures of both pre-fire tree health and fire severity, existing models of post-fire mortality that often rely on only measures of fire effects can be substantially improved. This study was conducted within an old-growth mixed conifer forest in Sequoia National Park, California, USA that had been prescribed burned during 2001 or 2002. Fire effects measured by percent crown volume scorched and stem char height, and pre-fire tree health measured by multiple indices of growth calculated from tree cores and measures of crown health were assessed for 105 sugar pine. Health status (live or dead) was observed prior to the fire and then immediately post-fire and five years post-fire and logistic regression models were used to evaluate the effects of fire and pre-fire tree health on post-fire mortality. Models based only on tree size and fire effects were compared to models that included fire effects and measures of pre-fire tree health using AICc. Five years following fire, the model that best predicted mortality includeddbh, crown volume scorched, 30 year growth trend, and count of sharp declines over a 30 year period. The inclusion of long-term measures of growth markedly improved model fit compared to models based only on fire effects (ΔAICc = 26.4). However, immediately after fire, models that included measures of pre-fire tree health resulted in only marginal improvements over models based only on measures of fire effects (ΔAICc = 2.1). These results imply that multiple processes, in addition to fire, are functioning to influence delayed mortality and that the inclusion of measures of tree health can provide more accurate predictions of post-fire mortality.

Introduction

Tree death is one of the most fundamental and yet complex processes controlling forest structure and dynamics (Franklin et al. 1987, Pedersen 1998, Das et al. 2007). To understand and predict how forests of today will look in the future, we must understand why trees die and what environmental and physical attributes or events predispose a tree to death (e.g., Waring 1987). This information is critical for improving predictions about an individual tree’s future probability of mortality based on the present and past conditions (Bigler and Bugmann 2004, Wunder et al. 2008). A basic understanding of tree mortality is central to many management goals focused on forest health, including post-fire salvage logging (McIver and Star 2000), forest pest management (Ferrell 1996), and forest restoration (Fulé et al. 2007).

Tree death can be a complex process dependent on multiple causes acting over different spatial and temporal scales. It can be sudden, caused by an event such as a flood or wildfire, or it
can be a slow accumulation of factors that eventually lead to death such as competition, herbivory, or disease (Franklin et al. 1987, Anderson 2000). Often, however, it is an interaction between short and long term stressors with both factors playing a role in controlling probability of mortality (Manion 1981, Pedersen 1998, Linares et al. 2010).

Sugar pine (Pinus lambertiana Douglas) is a gap-adapted species that is resistant to low severity fires due to its thick bark and open canopy (Kinloch et al. 1990, Habeck 1992). However, there has been growing concern that this species is being adversely affected by fire due to interacting factors, including the introduced pathogen white pine blister rust (Cronartium ribicola J.C. Fisch. ex Raben) (blister rust) and a history of fire exclusion, which has led to historically high fuel loads in many parts of its range (Skinner and Chang 1996, Stephens et al. 2009). Managers have become concerned that these stressors are leading to population declines (Smith 1992, van Mantgem et al. 2004) which may be exacerbated by the use of prescribed fire.

There has been extensive work done to develop models to predict mortality following fire (Ryan and Reinhart 1988, Regelbrugge and Conard 1993, Stephens and Finney 2002, van Mantgem and Schwartz 2004). These models have focused on correlating different measures of fire effects, such as stem charring or crown scorch, with probability of mortality. Few studies, however, have looked at how pre-disturbance tree health interacts with different forms of disturbance to control probability of mortality (but see van Mantgem et al. 2003).

Numerous empirical models have been developed to predict survival probability of trees in the absence of fire (‘background’ mortality) (e.g., Pedersen 1998, Bigler and Bugmann 2004, Das et al. 2007). The most common approach has been to use tree growth as a proxy for tree health, since growth rate has frequently been shown to be correlated with tree mortality risk (Wyckoff and Clark 2000, Bigler and Bugmann 2004). Most commonly, average recent growth has been used to predict mortality risk (Keane et al. 2001). van Mantgem et al. (2003) used a short term growth measure (average five year pre-fire growth rate) to demonstrate that the inclusion of tree health variables could improve mortality prediction post-fire for white fir (Abies concolor Gordon and Glend.). Das et al. (2007), however, found that several measures of growth, including average growth rate, growth trend measured as the slope of growth over time, and the number of sharp declines in growth rate could substantially improve predicted mortality. They also demonstrated that longer term measures of growth often provide more accurate predictions of mortality than short term ones. They hypothesized that long term measures provide a better measure of the cumulative stress that a given tree has experienced than short term measures. In sugar pine, for instance, they found that the 40 year trend in growth, combined with average growth and counts of recent abrupt declines in growth, substantially improved estimates of mortality risk compared to models based only on measures of recent growth.

The measures used by Das et al. (2007), however, pose a practical obstacle in that they require tree ring samples, which can be time consuming to collect, process, and analyze. Two alternative estimates of tree health are visual estimates of crown health (Salman and Bongberg 1942, Zarnoch et al. 2004) and live crown ratio (Dyer and Burkhart 1987, Dolph 1988). These measures do not provide a multi-year assessment of tree health like tree ring records do, but instead provide a single snapshot in time. They are much less time consuming to collect, however, and may provide an adequate estimate of tree health that can be used as an alternative to tree ring data.

This study focused on predicting mortality of sugar pine following prescribed fire. The main objective was to evaluate whether including measures of pre-fire tree health such as live crown
ratio and estimates of growth based on tree ring records, in addition to measures of fire effects, significantly improved prediction of post-fire mortality of sugar pine, both immediately post-fire and five years after fire. A better understanding of how tree health and fire effects control probability of mortality is essential for better understanding the effects of prescribed fire as a tool for forest management and restoration. Tree health can be very hard to measure, as it is a complex variable made up of many interacting factors. However, excluding pre-fire condition may well lead to poorer prediction of post-fire mortality, particularly for those trees that are not killed outright by the fire. This study will present a predictive model to calculate individual tree mortality probability following fire based on tree diameter, presence/absence of blister rust infections, crown condition, growth rate, and measures of fire effects. These models will be compared to more commonly used models based only on fire effects to test whether models that include both measures of tree health and fire effects more accurately predict post-fire mortality. This will allow managers to better plan for which trees may be at a higher risk to die following a fire. The results from this study will also allow managers to better assess how accurate more simply measured visual measures of tree health are relative to more detailed tree ring records in relation to post-fire risk of mortality.

**Methods**

This study was conducted within an old-growth mixed conifer forest in the Marble Fork drainage of Sequoia National Park, California, USA. Elevation ranges from 1900 to 2150 m within the study site. Soils consist primarily of coarse loams derived from decomposed granite. Average precipitation for this area is 1200 mm yr⁻¹ with most of this falling as snow. The most abundant overstory tree species are white fir, sugar pine, and incense cedar (*Calocedrus decurrens* Torrey). Red fir (*Abies magnifica* Murray), Jeffrey pine (*Pinus jeffreyi* Balf.), and ponderosa pine (*Pinus ponderosa* Lawson) also occur, but at lower abundance. The site has never been logged and had not experienced a stand-replacing fire in >100 years (Knapp et al. 2005).

Sugar pine were sampled from within five adjacent 15 ha to 20 ha prescribed burn units that were originally established as part of the national fire fire-surrogate (FFS) study. The purpose of the FFS study was to examine ecosystem response to silvicultural treatments designed to reduce fire hazard (McIver et al. 2001, Knapp et al. 2005, Waldrop and McIver 2006, Schwilk et al. 2006, Schwilk et al. 2009). The types of data that were collected included information on understory and overstory vegetation (Schwil et al. 2009, Stephens et al. 2009), fuels (Knapp et al. 2005, Stephens and Moghaddas 2005), soils (Boerner et al. 2009), insects (Apigian et al. 2006, Schwilk et al. 2009), and wildlife (Converse 2005). Within the burn units, 50 20 m × 50 m modified Whitaker plots (ten per burn unit) were established at permanent points along a 50 m grid system (for detailed methods of plot establishment see Schwilk et al. 2006). Prior to the prescribed fires, trees >1.37 m tall within these plots were tagged and mapped and diameter at breast height (dbh), tree height, height to live crown, blister rust infection status, and crown condition were recorded. Live crown ratio was calculated by dividing crown height (tree height - height to live crown) by total tree height. Blister rust status was recorded as a binary variable based on the presence of branch or stem cankers attributed to the pathogen. Crown condition was measured as a categorical variable based on a visual rating system developed by Salman and Bongberg (1942). Each tree was assigned a rating of one if it appeared healthy and vigorous, a rating of two if there were small defects, such as some short needles or minor patches of dead or needles (flagging), three if there were substantial signs of stress, such as broken tops or
significant shallow crowns, or a rating of four if there were major health issues such as top
dieback from blister rust or if the tree generally appeared near death. Visual health rating
protocols are commonly used to assess tree health in studies focused on measuring forest health

Prescribed burns were conducted in two of the burn units during September or October of
2001 and in the other three during June of 2002. Weather conditions were recorded hourly
during the burn and were similar within burns units that occurred during the same season. Pre-
fuel loads averaged 191.6 Mg ha\(^{-1}\) and pre-fuel moisture was similar between burns
within the same season (Knapp \textit{et al.} 2005). Fires were ignited using drip torches and were
primarily strip head fires that burned as surface fires at low to moderate intensity. Fuel loads
were reduced by 67\% in the early season burns and by 88 \% in the late season burns (Knapp \textit{et
al.} 2005).

Following the burns, fire effects were assessed by measuring percent crown volume scorched
(crown scorch), maximum stem char height (char height), and percent circumference of the base
of the stem that was charred (basal char) for each tagged tree. Trees were then recorded as live
or dead immediately (< 1 year) following fire during the summer of 2002, and then two, three,
and five years following fire. Health status (live or dead) was also recorded for sugar pine (n =
109) in one of the FFS control plots during these same remeasures to assess how mortality rates
in burned plots compared to background mortality rates in unburned plots. During the summer
of 2007, tree cores from 165 sugar pine \(\geq 10\) cm dbh were collected (96 dead and 69 live) within
the burned plots. Only trees \(\geq 10\) cm dbh at the time of the burn were used in this study to
ensure that long term growth records (at least 30 years) were available and because most trees
smaller than 10 cm were consumed by the fire. One or two cores were collected per tree at
breast height. Cores were then mounted and sanded to allow for an accurate measure of ring
width. Rings were measured using a dissecting microscope and sliding-stage micrometer to 0.01
mm accuracy. Many of the dead trees had significant rot, as they had been dead for several
years, resulting in only 105 trees (55 dead and 50 live) producing readable cores of at least 30
years in length. Nineteen of the live tree cores were excluded because of brakes in the cores or
insufficient number of rings. A master chronology was developed from the 21 oldest trees and
was used to check the cores for errors including missing or false rings using COFECHA1
(Grissino-Mayer 2001). Any errors that were identified were then verified by visual inspection
of the core. Only a small portion of the cores did not cross-date well to the master chronology
(nine cores has a correlation <0.1 with the master chronology) and all cores were retained in the
analysis.

\textbf{Data analysis}

The overall goal of the analysis was to compare how well different models predicted
immediate and delayed (five year) post-fire mortality based on measures of fire effects and tree
health. Analysis was conducted using the \texttt{lme4} package (Bates and Maechler 2010), Design
package (Harrell 2009), and \texttt{AICmodavg} (Mazerolle 2011) package in the R statistical program
(R Development Core Team 2010), and Splus version 8.1 (©MS Miami Corp). Logistic
regression models were used to model post-fire tree health status (live or dead). Given the
nested structure of the data, with trees nested within plots, within burn units, the analysis began
by testing whether a generalized mixed effects model approach (GLMM), which accounts for the
potential spatial correlation among trees substantially improved model fit over a logistic model
(GLM) that treated each tree as independent (Zuur 2009). All fire effects variables (crown
scorch, maximum stem char height, basal char, and burn season), tree health variables (crown health rating, live crown ratio, and blister rust status), and dbh were included as fixed effects. Burn unit and plot were included as nested random effects in the mixed effects model and model estimates were calculated using maximum likelihood methods. Model fit was assessed using corrected Akaike Information Criterion (AICc), with a meaningful change in model fit being indicated by a difference in AICc (ΔAICc) between models > two (Burnham and Anderson 2002). The logistic regression model without mixed effects had the lower AICc score (ΔAICc = 7), indicating that accounting for spatial correlations among trees at the plot and site scale using a mixed effects model did not substantially improve model fit compared to a logistic model that treated them as spatially independent. This is not uncommon in fire data as fire often produces heterogeneous effects on a small scale so that variance within a burn can be greater than between burns (van Mantgem and Schwilk 2009, Nesmith et al. 2011). Therefore, site effects were ignored and each tree was modeled as spatially independent.

The fit of models based on tree size, fire effects variables, tree health variables, and all variables combined were compared using AICc to assess whether the inclusion of measures of pre-fire tree health would substantially improve the predictive power of sugar pine mortality immediately following fire and five years post-fire. Because tree size is a significant factor related to post-fire mortality probability (see results), dbh was included in all models to account for differences in size between the live and dead trees and allow the other tree health and fire effects variables to be assessed after accounting for tree size.

Model selection was carried out over several steps. First, to evaluate the effect of fire effects on mortality, models based on all combinations of percent crown volume scorch, maximum stem char height, percent circumference char, and burn season were evaluated and the best model was selected based on AICc for both immediate and five year post-fire mortality. Next, models based on measures of pre-fire tree health, in addition to fire effects data and dbh were evaluated. Because there were many similar measures of pre-fire tree health (n = 33), and not all measures were expected to produce models with similar fit, the second step of the model selection process was to evaluate which individual measures of tree health performed the best based on AICc. Once the best fitting measures of tree health were determined, these were then tested in combination with the best fitting fire effects variables to determine whether the inclusion of pre-fire tree health variables improved model fit over models based only on fire effects.

The measures of tree health that were tested included live crown ratio, crown health rating, blister rust status, and multiple indices of growth measured from tree ring records. There were 30 different measures of growth in all, including annual growth immediately preceding fire (n=3), average growth over 5, 10, and 30 years (n=9), growth trend, defined as the linear rate of increase or decrease in growth over 5, 10, and 30 years (n=9), and count of sharp declines in growth over 5, 10, and 30 years (n=9). Sharp declines were defined as any annual decline in growth ≥ 50 % relative to the previous year. Time periods of 5, 10, and 30 years were selected because past research has found significant effects of growth over a five year period (van Mantgem et al. 2003) as well as ten year period (Das et al. 2007), and 30 years was the longest time period measured from the tree cores that could be assessed without having to further reduce the sample size. Each index of growth was calculated using radial increment, basal area increment, and relative basal area increment. However, dbh and average basal area increment were strongly correlated, causing the dbh coefficient to be insignificant and negative in many cases when both parameters were included in the same model. Therefore, average basal area was not used to calculate average growth, but was still included as a measure of growth trend and for
counts of rapid growth declines. Das et al. 2007 also found evidence of a positive correlation between basal area increment and dbh for live trees, whereas radial increment did not display a significant trend with tree dbh. This reduced the number of measures of growth that were tested from 30 to 26 (Table 3.1). Growth indices were calculated using different measures (radial increment, basal area, and relative basal area) because there is not a consensus in the literature as to which measure is best. Some have preferred absolute increment (Das et al. 2007), others basal area increment (Pedersen 1998, Bigler and Bugmann 2003), and others relative basal area increment (Disalvo and Hart 2002, Karlsson et al. 2006). Each has a different relationship with tree size and therefore produces slightly different results. No one measure preformed consistently better across all growth indices.

Each specific growth measure was paired with the variables from the best fire effects model for predicting immediate and five year post-fire mortality following the methods of Das et al. (2007) and model fit was evaluated using AICc. The best measures of each growth index (average growth, growth trend, sharp declines) were then selected to test in combination with the fire effects variables. Specific parameters were excluded from the model selection process if the model had a ΔAICc > 2 compared to the best growth parameter. This was done separately for the immediate and five year post-fire data (Table 3.1). For models based on immediate post-fire mortality data, all eight measures of average growth produced models within two AICc of the best model and were included in the model selection process. All measures of slope except 30 year relative basal area, 10 year average radial increment, and 10 year basal area were within two AICc and were included in the model selection process. Three measures of sharp declines produced models with substantially lower AICc values and included sharp decline measures calculated using 30 year average radial increment, 30 year basal area, and 30 year relative basal area. This filtering process reduced the number of growth parameters that were evaluated from 26 to 17 for the immediate post-fire mortality data (Table 3.1).

There was much greater differentiation between the different measures of growth to predict five year post-fire mortality compared to the immediate post-fire mortality data. The best predictor of mortality among the eight measures of average growth was 10-year average radial increment (Table 3.1). The best measure of slope was 30-year growth trend measured as basal area increment. The measures of sharp declines that best predicted five-year post-fire mortality were the count of declines within 30 years measured as average radial increment and basal area. The number of growth variables that were tested was therefore reduced to 4 from 26 for the five year post-fire data (Table 3.1).

The best measures from each growth category were then combined to create models that contained all possible combinations of parameters, with the restriction that only one measure of each type of growth (average, growth trend, or sharp decline) could be in the same model. For each set of models, the best model or models were selected based on AICc scores and the number of parameters in the model. First, all models were ranked based on AICc. Equivalent models (AICc within two units of the best ranked model) were then evaluated based on the number of parameters within the model and the model(s) with the lowest number of parameters was then chosen as the final model(s).
Table 3.1. Change in corrected Akaike information criterion (ΔAICc) of models to predict immediate and five year post-fire mortality of sugar pine in Sequoia National Park based on individual measures of pre-fire growth. All growth variables within each different measure of growth (Average growth, Growth trend, and Sharp declines) within two AICc were then used in the final model selection process to produce models based on both pre-fire tree health and fire effects. These variables are shown in bold. Dashes indicate variables that were not tested and NA indicates no difference between live and dead trees for that measure of growth.

<table>
<thead>
<tr>
<th>Immediate post-fire mortality</th>
<th>ΔAICc Average Growth</th>
<th>ΔAICc Growth Trend</th>
<th>ΔAICc Sharp Declines</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lastyrdiam</td>
<td>0</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>lastyearRba</td>
<td>0.75</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>5 yr increment</td>
<td>1.52</td>
<td>0.95</td>
<td>2.67</td>
</tr>
<tr>
<td>10 yr increment</td>
<td>0.82</td>
<td>2.23</td>
<td>2.68</td>
</tr>
<tr>
<td>30 yr increment</td>
<td>1.04</td>
<td>0.79</td>
<td>0</td>
</tr>
<tr>
<td>5 yr ba</td>
<td>-</td>
<td>0</td>
<td>2.67</td>
</tr>
<tr>
<td>10 yr ba</td>
<td>-</td>
<td>2.00</td>
<td>2.68</td>
</tr>
<tr>
<td>30 yr ba</td>
<td>-</td>
<td>0.47</td>
<td>0.14</td>
</tr>
<tr>
<td>5 yr relative ba</td>
<td>1.31</td>
<td>1.92</td>
<td>NA</td>
</tr>
<tr>
<td>10 yr relative ba</td>
<td>0.39</td>
<td>1.54</td>
<td>NA</td>
</tr>
<tr>
<td>30 yr relative ba</td>
<td>1.88</td>
<td>2.19</td>
<td>1.60</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Five-year post-fire mortality</th>
<th>ΔAICc Average Growth</th>
<th>ΔAICc Growth Trend</th>
<th>ΔAICc Sharp Declines</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lastyrdiam</td>
<td>6.92</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>lastyearRba</td>
<td>20.79</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>5 yr increment</td>
<td>4.02</td>
<td>15.477</td>
<td>15.73</td>
</tr>
<tr>
<td>10 yr increment</td>
<td>0</td>
<td>18.031</td>
<td>14.29</td>
</tr>
<tr>
<td>30 yr increment</td>
<td>2.14</td>
<td>15.851</td>
<td>1.23</td>
</tr>
<tr>
<td>5 yr ba</td>
<td>-</td>
<td>16.147</td>
<td>15.09</td>
</tr>
<tr>
<td>10 yr ba</td>
<td>-</td>
<td>18.50</td>
<td>13.40</td>
</tr>
<tr>
<td>30 yr ba</td>
<td>-</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>5 yr relative ba</td>
<td>20.74</td>
<td>18.745</td>
<td>NA</td>
</tr>
<tr>
<td>10 yr relative ba</td>
<td>20.216</td>
<td>17.721</td>
<td>NA</td>
</tr>
<tr>
<td>30 yr relative ba</td>
<td>20.964</td>
<td>17.831</td>
<td>13.51</td>
</tr>
</tbody>
</table>
Model goodness of fit for the best models was assessed using a sum of squares test for logistic models. A $p$-value $> 0.05$ indicates adequate model fit (Hosmer et al. 1997). Evidence ratios were also calculated for each model. The evidence ratio was calculated as the ratio of the corrected Akaike score weights of a given model against the corrected Akaike score of the best model. It provides a rough idea of how much better one model is relative to another with higher values indicating less evidence for a model given the data (Burnham and Anderson 2002). Model discrimination between dead and live trees was assessed using receiver operating characteristic (ROC). A higher ROC value indicates that a higher proportion of trees were correctly classified as live or dead by the model. The percentage of trees correctly classified by each model was also determined based on an optimal cut point determined by the data. The optimal cut point is the predicted survival probability that maximizes model sensitivity and specificity (Hosmer and Lemeshow, 2000).

Results

Of the 105 sugar pine, 55 (52.4 %) were dead five years following fire, with 40 dying during the first year (Figure 3.1). Annual mortality rate peaked at 23.8 % immediately post-fire and declined steadily over the course of the study to the point where no new mortality occurred between 2006 and 2007. The annual mortality rate four and five years post-fire was similar to background mortality rates of sugar pine within a nearby unburned control FFS plot (Figure 3.1).

![Figure 3.1](image.png)

**Figure 3.1.** Annual post-fire mortality rate of sugar pine in Sequoia National Park. By 2007, 55 of the 105 trees in burned plots had died (average annual mortality = 9 %). 34 of the 109 trees in the control plot had died (average annual mortality = 5 %).
The average pre-fire dbh of trees that were alive five years post-fire was significantly larger (Welch two-sample t-test, \( p < 0.001 \)) than that of dead trees averaging 50.7 cm compared to 28.1 cm, respectively. There were also significant differences in the amount of fire damage observed between live and dead trees. Crown volume scorched displayed a bimodal distribution, with the crown of most trees either being scorched completely or not at all. Median crown volume scorched was 5 \% in trees that were alive five years post-fire compared to 90 \% for dead trees (Wilcoxon rank sum test, \( p < 0.001 \)). Maximum stem char height was strongly skewed to the right with most trees receiving either no or very little (0.1 cm) charring, while a few trees were charred > 15 m up the stem (range = 0 to 19 m). While maximum stem char height differed between live (median = 95 cm) and dead (median = 150 cm) trees five years post-fire, this difference was not significant (Wilcoxon rank sum test, \( p = 0.533 \)). Basal char also displayed a bimodal distribution, and was significantly lower in live trees than dead trees, with a median value of 80 \% and 100 \%, respectively (Wilcoxon rank sum test, \( p = 0.002 \)). Trees were split evenly between burn units that were burned during the spring and those that burned during the fall with 53 trees burning during spring burns and 52 trees during fall burns and the mortality rate five years post-fire was similar between burn seasons (Fisher exact test, \( p = 0.12 \)).

Percent live crown ratio was similar between live and dead trees five years post-fire with a median value of 0.78 and 0.80, respectively (Wilcoxon rank sum test, \( p = 0.741 \)). Blister rust status was also similar between live and dead trees five years post-fire with five live trees (10 \%) showing signs of infection compared to six dead trees (11 \%). Pre-fire crown health rating, however, differed significantly (Fisher exact test, \( p < 0.001 \)) between live and dead trees five years post-fire (Figure 3.2). This difference was due primarily to the number of dead trees assigned a poor health rating prior to the burn, as only two live trees were assigned a crown health rating of four compared to 21 dead trees five years post-fire.
Models to predict immediate post-fire mortality based on measures of fire effects alone were extremely accurate (Table 3.2). The best predictors of mortality immediately post-fire when only fire effects variables were considered were dbh, crown scorch, and basal char (Table 3.3). The logistic regression model using these explanatory variables had a ROC of 0.98 indicating extremely high discrimination in the classification of live and dead trees. This model classified 92% of all trees correctly (92% of live trees classified correctly, 93% of dead trees classified correctly), using an optimal cut point of 0.81 survival probability (Figure 3.3).
Table 3.2. Corrected Akaike information criterion (AICc), receiver operating characteristic (ROC), sum of squares p-value (SSp), and Evidence Ratio (Evidence) of models for predicting sugar pine mortality in Sequoia National Park one year post-fire. Model Type refers to models that used only fire variables (Fire) or that included measures of pre-fire tree health in addition to fire effects variables (Combination). Status02 is a binary variable indicating whether a tree was alive or dead one-year post-fire. Explanatory variables included diameter at breast height (DBH), percent crown volume scorched (PerCrwnVolSc), Percent circumference of the bole that was charred (PercBoleCircChar), average 10 year growth increment measured as radial increment (avg10) or relative basal area increment (avg10rba), and growth one year preceding fire measured in radial increment (lastyeardiam), basal area (lastyearba) or relative basal area (lastyearrba).

<table>
<thead>
<tr>
<th>Model Type</th>
<th>Model</th>
<th>AICc</th>
<th>ΔAICc</th>
<th>ROC</th>
<th>SSp</th>
<th>Evidence Ratio</th>
</tr>
</thead>
<tbody>
<tr>
<td>Growth</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar+lastyeardiam</td>
<td>41.385</td>
<td>0</td>
<td>0.982</td>
<td>0.287</td>
<td>1</td>
</tr>
<tr>
<td>Growth</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar+avg10rba</td>
<td>41.775</td>
<td>0.39</td>
<td>0.981</td>
<td>0.426</td>
<td>1.22</td>
</tr>
<tr>
<td>Growth</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar+lastyearRba</td>
<td>42.136</td>
<td>0.751</td>
<td>0.982</td>
<td>0.568</td>
<td>1.46</td>
</tr>
<tr>
<td>Growth</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar+avg10</td>
<td>42.208</td>
<td>0.823</td>
<td>0.98</td>
<td>0.194</td>
<td>1.51</td>
</tr>
<tr>
<td>Fire</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar</td>
<td>43.459</td>
<td>2.074</td>
<td>0.977</td>
<td>0.310</td>
<td>2.82</td>
</tr>
<tr>
<td>Health</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar+CrwnHlthR</td>
<td>43.633</td>
<td>2.248</td>
<td>0.985</td>
<td>0.790</td>
<td>3.08</td>
</tr>
<tr>
<td>Health</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar+LCrwnR</td>
<td>44.476</td>
<td>3.091</td>
<td>0.977</td>
<td>0.395</td>
<td>4.69</td>
</tr>
<tr>
<td>Health</td>
<td>Status02~DBH+PerCrwnVolSc+PercBoleCircChar+BRStatus</td>
<td>44.875</td>
<td>3.490</td>
<td>9.976</td>
<td>0.675</td>
<td>5.73</td>
</tr>
</tbody>
</table>
Table 3.3. Coefficient estimates and standard errors for logistic regression models that included only dbh and fire damage parameters (Fire) and models that also included measures of tree growth (Growth) to predict immediate post-fire mortality and fire-year post-fire mortality for sugar pine in Sequoia National park.

<table>
<thead>
<tr>
<th></th>
<th>Estimation</th>
<th>Std. Error</th>
<th>Estimation</th>
<th>Std. Error</th>
</tr>
</thead>
<tbody>
<tr>
<td>Immediate post-</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>fire mortality</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>intercept</td>
<td>-1.057</td>
<td>1.191</td>
<td>-2.237</td>
<td>1.429</td>
</tr>
<tr>
<td>DBH</td>
<td>0.122</td>
<td>0.049</td>
<td>0.113</td>
<td>2.183</td>
</tr>
<tr>
<td>PercrwnVolSc</td>
<td>-0.146</td>
<td>0.051</td>
<td>-0.157</td>
<td>0.054</td>
</tr>
<tr>
<td>PercBoleCircChar</td>
<td>0.114</td>
<td>0.050</td>
<td>0.126</td>
<td>0.052</td>
</tr>
<tr>
<td>lastyeardiam</td>
<td>-</td>
<td>-</td>
<td>1.33</td>
<td>0.703</td>
</tr>
<tr>
<td>Five-year post-</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>fire mortality</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>intercept</td>
<td>-0.250</td>
<td>0.426</td>
<td>0.317</td>
<td>0.531</td>
</tr>
<tr>
<td>DBH</td>
<td>0.028</td>
<td>0.009</td>
<td>0.024</td>
<td>0.010</td>
</tr>
<tr>
<td>slope30ba</td>
<td>-0.027</td>
<td>0.007</td>
<td>-0.035</td>
<td>0.009</td>
</tr>
<tr>
<td>decline30ba</td>
<td>-</td>
<td>-</td>
<td>-1.105</td>
<td>0.393</td>
</tr>
</tbody>
</table>

Models of immediate post-fire mortality that included measures of pre-fire tree health and growth in addition to measures of fire effects displayed only marginal improvement in AICc compared to models based only on measures of fire effects (Table 3.2). A total of 36 different models using different measures of growth produced models within two AICc. Of these 36, four included only one additional measure (the other models included multiple growth parameters) of growth along with measures of fire effects and dbh (Table 3.2). Each model had a ROC of 0.98 and included dbh, percent crown volume scorched, basal char, and then either last year growth, measured as radial increment or relative basal area, or average ten-year growth measured as radial increment or relative basal area. Evidence ratios for these models ranged between 1 and 1.51, compared to 2.82 for the fire only model, suggesting only weak evidence of improvement over the fire effects only model. The model that included last year radial growth predicted 94% of all trees correctly (94% of live trees and 95% of dead trees), using an optimal cut point of 0.74 survival probability. This model was the only one of the four equivalent best models that performed better than the fire effects only model based on AICc, though only marginally so (ΔAICc = 2.07). The coefficient estimate of last year radial increment was positive (Table 3.3) indicating that trees with faster growth tended to survive immediately post fire, though this result was not significant (p = 0.057). Overall, the addition of tree health variables to the fire only model did not substantially improve model fit (Table 3.2).
When delayed mortality was accounted for, models that included measures of fire effects as well as measures of tree health performed substantially better than models that included only measures of fire damage to predict five year post-fire mortality (Table 3.4). The best model based only on measures of fire effects included dbh and crown scorch as the explanatory variables and had a ROC of 0.84 (Table 3.4). Increasing tree size had a positive effect on post-fire survival and increasing crown scorch had a negative effect (Table 3.3). The evidence ratio for this model was >500,000 when compared to the best model that also included measures of pre-fire tree growth, indicating very strong evidence that the model including growth measures was a better fitting model. The fire effects only model classified 78% of all trees correctly using an optimal cut point of 0.54 survival probability and correctly classified 76% of the live trees and 80% of the dead trees (Figure 3.3).

**Figure 3.3.** Predicted probability of survival for sugar pine immediately and five years post-fire in Sequoia National Park, CA.
Table 3.4. Corrected Akaike information criterion (AICc), receiver operating characteristic (ROC), sum of squares $p$-value (SSp), and Evidence Ratio (Evidence) of models for predicting sugar pine mortality in Sequoia National Park five years post-fire. Model Type refers to models that used only fire variables (Fire) or that included measures of pre-fire tree health in addition to fire damage variables (Combination). Status07 is a binary variable indicating whether a tree was alive or dead six-years post-fire. Explanatory variables included diameter at breast height (DBH), percent crown volume scorched (PerCrwnVolSc), average 30 year growth trend measured as basal area increment (slope30ba), and the number of sharp declines in growth over a 30 year period measured in radial increment (decline30) or basal area (decline30ba).

<table>
<thead>
<tr>
<th>Model Type</th>
<th>Model</th>
<th>AICc</th>
<th>ΔAICc</th>
<th>ROC</th>
<th>SSp</th>
<th>Evidence Ratio</th>
</tr>
</thead>
<tbody>
<tr>
<td>Growth</td>
<td>Status07~DBH+PerCrwnVolSc+slope30ba+decline30ba</td>
<td>88.511</td>
<td>0</td>
<td>0.913</td>
<td>0.810</td>
<td>1</td>
</tr>
<tr>
<td>Growth</td>
<td>Status07~DBH+PerCrwnVolSc+slope30ba+decline30</td>
<td>89.952</td>
<td>1.441</td>
<td>0.908</td>
<td>0.682</td>
<td>2.06</td>
</tr>
<tr>
<td>Health</td>
<td>Status07~DBH+PerCrwnVolSc+CrwnHlthR</td>
<td>111.574</td>
<td>23.063</td>
<td>0.859</td>
<td>0.332</td>
<td>101,874.80</td>
</tr>
<tr>
<td>Fire</td>
<td>Status07~DBH+PerCrwnVolSc</td>
<td>114.886</td>
<td>26.375</td>
<td>0.840</td>
<td>&lt;0.001</td>
<td>533,652.42</td>
</tr>
<tr>
<td>Health</td>
<td>Status07~DBH+PerCrwnVolSc+LCrwnR</td>
<td>115.423</td>
<td>26.912</td>
<td>0.839</td>
<td>&lt;0.001</td>
<td>698,017.90</td>
</tr>
<tr>
<td>Health</td>
<td>Status07~DBH+PerCrwnVolSc+BRStatus</td>
<td>116.153</td>
<td>27.642</td>
<td>0.845</td>
<td>&lt;0.001</td>
<td>1,005,505.00</td>
</tr>
</tbody>
</table>
There were eight equivalent models that best predicted five year post-fire mortality using measures of tree health as well as fire effects, two of which included only two growth parameters in addition to dbh and crown scorch. The growth parameters in these two models were 30 year slope measured as basal area, and the number of rapid declines within the 30 year period, measured as either basal area or radial increment (Table 3.4). Both models had a ROC of 0.91 and the evidence ratios for these models were 1 and 2.06, respectively. The model that included rapid declines measured in basal area classified 85% of all trees correctly (84% of live trees and 86% of dead trees), using an optimal cut point of 0.33 survival probability. The coefficient estimate of growth trend was positive, indicating that trees with increasing growth rates were more likely to survive following fire whereas the coefficient for sharp declines was negative indicating that trees with more sharp declines were less likely to survive (Table 3.3). Regardless of what measure of growth was used, almost all models that contained some measure of growth and fire effects performed significantly better at predicting delayed mortality than models with only fire effects data. In addition, including visual crown health rating substantially improved model fit compared to the model based on fire effects only, though not as much as the models based on tree growth (Table 3.4). The inclusion of blister rust status or live crown ratio did not improve model fit compared to the fire only model, however (Table 3.4).

Discussion

Trees which are exposed to long-term stressors are more likely to die when confronted with additional stressors, such as drought or defoliating insects (Manion 1981, Franklin et al. 1987). This concept has been supported by others who have found that trees experiencing a long-term stress, such as competition, are more susceptible to short-term stresses like drought (Linares et al. 2010). In addition, the effect of previous stress on tree health appears cumulative as past events, even if in the distant past, can make trees more susceptible to mortality (Petersen 1998, Das et al. 2007). This study has shown that trees that are less healthy will be more susceptible to mortality when confronted with the additional stress of fire as mortality models that included measures of both fire effects and pre-fire tree health performed much better than models that only included measures of fire effects.

The models of post-fire mortality of sugar pine based on measures of fire effects alone produced similar results to other studies of the effects of fire on sugar pine mortality (Stephens and Finney 2002, Hood et al. 2007, Hood et al. 2010). The model based on dbh, crown volume scorch, and basal char correctly classified 92% of the sugar pine as live or dead immediately post-fire. Crown volume scorched and basal char were also found to be strong predictors of sugar pine mortality three years post-fire by Stephens and Finney (2002), but they found less evidence that dbh was a significant factor controlling probability of mortality. Dbh and crown volume scorched were also powerful predictors of five year post-fire mortality, though the percent of trees classified as live or dead correctly (78%) was lower than the model based on immediate post-fire mortality data.

When evaluating which variables best predict post-fire mortality, it is important to account for delayed mortality following fire because this additional mortality can alter
model predictions and accuracy dramatically. The inclusion of pre-fire health variables in models of five year post-fire mortality substantially improved model fit ($\Delta$AICc = 26.4) and model discrimination ($\Delta$ROC = 0.073) compared to models that only included measures of fire effects, while the inclusion of pre-health variables only marginally improved model fit ($\Delta$AICc = 2.07) and did not alter model discrimination substantially ($\Delta$ROC = 0.005) of immediate post-fire mortality. This indicates that fire effects are very good predictors of immediate post-fire mortality but do not correlate as well to factors which control delayed mortality (i.e., the mortality risk of those trees which survive the initial effects of the burn). There appear to be other factors besides fire effects influencing delayed mortality that are better captured when measures of pre-fire tree health are included. van Mantgem et al. (2003) also found that models that included both measures of growth and fire effects performed better than models based on fire effects alone. However, they only used average growth five years post-fire, in addition to crown scorch. In this study, long-term measures of growth based on 30 years of growth performed substantially better than models based on only five year or ten year growth at predicting five year post-fire mortality. The best measures of tree health for predicting five year post fire mortality were 30 year growth trend, and the count of sharp declines in growth over the last 30 years. These were quite similar to the measures of tree growth that best predicted sugar pine mortality by Das et al. (2007), though their best models were based on 40 year averages instead of 30. It is interesting to note that similar growth variables were selected for the best models to predict both background mortality rates in unburned sites by Das et al. (2007) as well as post-fire mortality rates in this study. In addition, while models that included measures of fire effects and long-term growth provided the best estimates of delayed post-fire mortality, models that used visual crown health rating as a substitute for measures of growth also out-performed models based on crown scorch (though tree-ring models substantially out-performed these models).

Sugar pine in particular may be at higher risk of mortality following fire due to multiple factors such as blister rust, increased fuel loads, especially around the base of larger trees, and changes in climate (van Mantgem et al. 2004, Tomback and Achuff 2010, Nesmith et al. 2010). Prescribed fire has become one of the main silvicultural tools used for management and restoration of sugar pine (Tomback and Achuff 2010). A better understanding of long-term post-fire mortality is critically important for achieving management objectives for this species given local population declines that are expected to continue (van Mantgem et al. 2004). The inclusion of blister rust status in models to predict both immediate and delayed post-fire mortality did not improve model fit, however. This may be due to the small number of infected trees within the study area, making the detection of statistical differences in mortality between infected and uninfected trees difficult.

Conclusions
Predicting mortality following fire is an important goal for forest managers. Models based on tree size and various measures of fire effects are most commonly used to accomplish this. While special attention is often paid to fuel loads and fire weather during prescribed fires to control post-fire mortality rates, the potential effects of pre-fire forest health are rarely considered. This study shows that while immediate post-fire
mortality can be accurately predicted from fire effects, incorporating measures of pre-fire tree health into the model, especially measures of long-term growth, can substantially improve long-term mortality predictions. In addition, the measures of tree health that best predict post-fire mortality change based on how long after fire mortality is measured. For example, crown scorch, dbh and two 30-year measurements are best for predicting mortality five years after fire compared to crown scorch, basal char and dbh for predicting immediate post-fire mortality. This implies that multiple processes, in addition to fire effects, are functioning to control delayed mortality and including measures of pre-fire tree health can improve prediction of long-term post-fire mortality and should be considered when the accuracy of post-fire mortality predictions is important.

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References


Chapter 4

The effects of raking on sugar pine mortality following prescribed fire in Sequoia and Kings Canyon National Parks

Jonathan C.B. Nesmith

Prescribed fire is an important tool for fuel reduction, the control of competing vegetation, or forest restoration. The accumulated fuels associated with historical fire exclusion can cause undesirably high tree mortality rates following prescribed fires and wildfires. This is especially true for sugar pine, which is already negatively affected by the introduced pathogen white pine blister rust. I tested the efficacy of raking away fuels around the base of sugar pine to reduce mortality following prescribed fire in Sequoia and Kings Canyon National Parks, CA. This study was conducted in three prescribed fires and included 457 trees, half of which were raked to mineral soil to 0.5 m away from the stem. Fire effects were assessed and tree mortality was followed for three years after prescribed fires. Overall, raking had no detectable effect on mortality as raked trees averaged 30 % mortality compared to 36 % for unraked trees. There was a significant interaction, however, between raking and average pre-treatment forest floor fuel depth: The predicted probability of survival of a 50 cm dbh tree was 0.94 vs. 0.96 when average pre-treatment fuel depth was 0 cm for a raked and unraked tree, respectively. When average pre-treatment forest floor fuel depth was 30 cm, the predicted probability of survival for a raked 50 cm tree was 0.60 compared to only 0.07 for an unraked tree. Raking did not affect mortality when fire intensity, measured as percent crown volume scorched, was very low (0 % scorch) or very high (>80 % scorch), but the raking treatment significantly increased the proportion of trees that survived by 9.6 % for trees that burned under moderate fire intensity (1 % to 80 % scorch). Raking significantly reduced the likelihood of bole charring and bark beetle activity three years post-fire. Fuel depth and anticipated fire intensity need to be accounted for when deciding whether to implement raking treatments to maximize their effectiveness. Raking is an important management option to reduce tree mortality from prescribed fire, but is most effective under specific fuel and burning conditions.

Introduction

A century of fire exclusion has dramatically shifted forest structure and fuel availability so that many western US forests are more prone to severe fires than in the past (Keeley and Stephenson 2000, Agee and Skinner 2005, Westerling et al. 2006,
Major efforts have been proposed on a national level to reduce forest fuels to levels closer to what they were prior to large scale fire suppression to reduce the risk of large, severe wildfires (Keeley and Stephenson 2000, Agee and Skinner 2005, Sun 2006). Prescribed fire has been used by land managers as one of the primary tools to achieve these goals (Parsons and Botti 1996, Stephens and Ruth 2005).

Prescribed fire has been used in many different fire-dependent ecosystems and regions as an important tool for forest restoration. Prescribed fire has been used extensively in the ponderosa pine (*Pinus ponderosa* Dougl. ex P. & C. Laws.) forests of the southwestern US (Allen et al. 2002, Kolb et al. 2007), longleaf pine (*Pinus palustris* Mill.) forests of the southeastern US (Varner et al. 2005, Lavoie et al. 2010), mixed conifer forests of the Pacific Northwest (Agee 1993, Busse et al. 2000), and many others. Prescribed fire can alter ecosystem properties and functions including the reduction of fuel loads, altered structure and composition of communities, and enhanced nutrient cycling (Vose 2000, Neary et al. 2005). By restoring natural forest processes through prescribed fire, managers can increase forest health, reduce fire hazard, and potentially make forests more resilient to climate change (Mutch and Cook 1996, Covington et al. 1997, Millar et al. 2007). However, a common concern with prescribed fire, especially after long periods of fire exclusion, is that fire behavior will be outside the historical range due to high fuel loads and changes to forest structure that have occurred during the previous period of fire exclusion (McHugh and Kolb 2003, Varner et al. 2005, Kolb et al. 2007). Methods to mitigate possible undesired effects of the re-introduction of fire may therefore be warranted.

Among western US conifers, sugar pine (*Pinus lambertiana* Dougl.) is a species that has experienced an apparent shift in health due to the change in forest structure and dynamics relating to fire suppression and the invasive pathogen white pine blister rust (*Cronartium ribicola* J.C. Fisch. ex Raben). Specifically, following >100 years of fire exclusion, sugar pine experiences high mortality in response to fire, a trend displayed by most tree species within the mixed conifer forest type (Mutch and Parsons 1998, van Mantgem et al. 2004). Post-fire death of sugar pine is a management concern because the effects of blister rust could be compounded by fire or beetle outbreaks (Paine et al. 1998). There is potential for widespread mortality as observed in other white pine species because sugar pine displays moderate to low levels of genetic resistance to this pathogen, which is also continuing to evolve and overcome host resistance (Kinloch and Comstock 1981, Kinloch 1992, Kendall and Keane 2001). When populations are already weakened by blister rust, additional fire-caused mortality could potentially contribute to extirpation, especially in the face of changing climate and fire regimes (van Mantgem et al. 2004, Bigler et al. 2005, Parker et al. 2006). Positive correlations between blister rust and bark beetle activity have also raised concerns that sugar pine may be experiencing elevated levels of beetle activity prior to prescribed fire and that interactions between these pathogens may lead to elevated mortality following fire (Thomas and Agee 1986, van Mantgem et al. 2004, Kulakowski and Veblen 2007).

Managers have expressed a need to develop strategies to protect sugar pine from short-term negative effects of fire. Tree mortality from fire can occur from direct effects of the fire such as crown scorching and cambium injury as well as indirect effects such as increased beetle activity (Hood 2010). While crown scorch is often ascribed as being the best predictor of tree mortality (Fowler and Seig 2004), cambial injury is also an
important contributing factor to mortality (Peterson and Ryan 1986, Hood et al. 2007a, Kolb et al. 2007). Cambial tissues are killed when temperatures reach 60 °C (Dickinson and Johnson 2004). Consumption of fuel at the base of the tree, and duff in particular due to its potential prolonged release of heat, has been found to be an accurate predictor of cambium mortality for ponderosa pine (Ryan and Frandsen 1991). Tree mortality from cambium injury is of particular concern in areas that have experienced long periods of fire exclusion allowing the depth of forest floor fuels to exceed historical ranges.

One management option is the localized removal of forest floor fuels around the base of individual trees via raking. Raking can alter the effects of fire for individual trees and reduce mortality by decreasing the amount of fine root damage and cambial death that can occur (Swezy and Agee 1991, Haase and Sackett 1998, van Mantgem and Schwartz 2004). Reduced mortality may occur directly as a result of diminished fire injury, or indirectly by decreasing the chance of beetle attack of injured trees following fire (Thomas and Agee 1986, Perrakis and Agee 2006, Breece et al. 2008).

Raking has been used to reduce localized fuels in many different forest types and regions including mixed conifer forests in the Pacific Northwest (Swezy and Agee 1991, Perrakis and Agee 2006), ponderosa pine forests in the southwestern US (Fulé et al. 2002, Jerman et al. 2004, Fowler et al. 2010a), and longleaf pine forests in the southeastern US (Williams et al. 2006). Raking was first mentioned as a way to counter undesirably high rates of mortality that occurred following prescribed fires in ponderosa pine forests, especially for large trees (Thomas and Agee 1986, Sackett et al. 1996). Raking and other methods including leaf blowing are being used more frequently in the western United States with mixed results (Hood 2010). Raking was found to successfully reduce mortality in young ponderosa pine stands by van Mantgem and Schwartz (2004) and in both large and small pine trees by Laudenslayer et al. (2008). However, several other studies have found raking to have little effect on tree mortality or stem charring (Fulé et al. 2002, Fowler et al. 2010b).

There are several reasons why the removal of fuels by raking may have produced varied results in previous studies. There has not been a standard depth or width at which raking treatments are applied among studies. Some studies have raked litter and duff away from the base of the stem as little as 0.3 m (Fulé et al. 2002), while others have cleared away fuels over continuous large areas up to 0.2 ha (Feeney et al. 1998). The depth of raking has also differed dramatically across studies with some removing all fuels down to mineral soil (Laudenslayer et al. 2008), others cleared away only litter (Jerman et al. 2004), while others removed the duff, but then replaced the litter layer (Covington et al. 1997, Kolb et al. 2001). At least one study conducted by Williams et al. (2006) compared several different treatments, including different methods of fuels removal, and found that all treatments significantly reduced mortality of longleaf pine compared to a control, but did not differ from each other.

Heterogeneous fuels, forest type, and season of burning also affect mortality and the effectiveness of fuels removal treatments (Kaufmann and Covington 2001, Busse et al. 2005, Perrakis and Agee 2006). The timing of post-fire mortality assessment has differed between studies with some studies revisiting trees only one year post-fire, while others have assessed post-fire mortality over several years. Often the cause of mortality is not reported and it can be unclear whether death was caused directly by the fire or another agent such as beetles that invaded after fire. In addition, many of these studies suffer
from low sample sizes or include only one replicate per treatment. Studies have also tended to focus on low severity burns and rarely encompass the range of burn severities that can occur during prescribed fires (Hood 2010).

In this study I tested whether raking away fuels from the base of sugar pine trees could be an effective management approach for reducing mortality following prescribed fire and if so, under what conditions was raking most effective. Specifically, I tested the effect of raking on tree mortality rates, from direct effects including crown scorch, litter and duff consumption, and stem charring, and indirect effects of post-fire beetle activity. This study examined over 450 trees across three sites within Sequoia and Kings Canyon National Parks, CA. These parks are an ideal location to examine these questions because of their long history of prescribed burning and sites that contain large numbers of sugar pine across a wide range of sizes and ages, i.e. old-growth forest conditions.

Methods

Study Sites

The study was conducted in Sequoia and Kings Canyon National Parks in the Sierra Nevada of California. This area is characterized by a Mediterranean climate of wet winters and dry summers with most precipitation falling as snow (Stephenson 1988). Mean annual temperatures within the parks range from a low of 0 °C during the winter to 18 °C during the summer. Soils are generally coarse loams derived from decomposed granite (Huntington and Akeson 1987). Three sites, Cabin Creek, Redwood Canyon, and Wall Spring, were located in old-growth mixed conifer forest spanning an elevation of 1800 m at Redwood Canyon to 2300 m at Cabin Creek.

Cabin Creek is located in Sequoia National Park (36° 37’ N, 118° 50’ W). Tree species include white fir (Abies concolor (Gord. & Glend.) Lindl. ex Hildebr.), red fir (Abies magnifica A. Murr), Jeffrey pine (Pinus jeffreyi Grev. & Balf.), sugar pine, and California black oak (Quercus kelloggi Newb.). Fuels were categorized as a mix of fuel models 5, 8, 9, and 10 (Anderson 1982). During November 8-10, 2006, 178 ha (435 ac) were prescribe burned using drip torch and aerial ignitions. Weather and ignition patterns produced a mixture of backing and heading fires. This was the first time this site had burned in at least 60 years. This fire occurred after the first precipitation event of the fall, resulting in relatively high fuel moisture. Winds ranged from 0-5 mph, with gust up to 7 mph. Minimum relative humidity was 33% and maximum temperature was 64 °F.

Redwood Canyon is in a giant sequoia-mixed conifer forest located in Kings Canyon National Park (36° 42’ N, 118° 55’ W) and had last burned in 1970 via a prescribed fire (Kilgore 1973). Tree species include giant sequoia (Sequoiadendron giganteum (Lindl.) J. Buchholz), white fir, sugar pine, incense cedar (Calocedrus decurrens (Torr.) Florin), ponderosa pine, and California black oak. Fuels were categorized as a mix of fuel models 8, 9, 10, and 14 (Anderson 1982). 1000 hour fuel moisture three days prior to ignition ranged from 16 % to 31 %. A 250 ha (619 ac) prescribed fire was ignited July 5-9, 2006. Ignition was accomplished through a combination of drip torch and aerial ignitions with mixture of both backing and heading fires. During the fire, winds ranged from 0-5 mph, minimum relative humidity ranged between 26 % and 75 %, and maximum temperature reached 80 °F.
Wall Spring is located in Giant Forest, Sequoia National Park (36° 33’ N, 118° 46’ W). This site is in a giant sequoia-mixed conifer forest and the area had not burned in at least 100 years (Caprio and Swetnam 1995). Tree species include white fir, giant sequoia, sugar pine, incense cedar, red fir, ponderosa pine, and California black oak. The site contained fuel models 5, 8, 9, 10, and 14 (Anderson 1982). The site was prescribed burned September 30 through October 2, 2007 using drip torches and mixture of backing and heading fires. Winds ranged from 2-7 mph, minimum relative humidity ranged between 22 % and 44 %, and maximum temperature reached 68 °F.

**Experimental design**

One ha plots were randomly located within Cabin Creek (10 plots), Redwood Canyon (10 plots), and Wall Spring (7 plots). Plots in Redwood Canyon were located within an 84 ha area of the 250 ha burn unit to exclude areas with low sugar pine density. Prior to each fire, sugar pine with a diameter at breast height (dbh) ≥ 10 cm were tagged and measured within each plot. Tree size (dbh and height), tree vigor, measured as a visual assessment of crown health following methods developed by Salman and Bongberg (1942), blister rust infection status (present or absent), and beetle activity (present or absent) were recorded. Average litter and duff depth was calculated for each tree by averaging the depth of litter and duff at each cardinal direction at the base of the tree. To reduce soil disturbance and expedite data collection, the maximum duff layer depth was censored at 30 cm. The censoring of the duff depth measurements caused average duff depth to be underestimated in 1 % of the trees. These trees with censored data were evenly split between raked and unraked trees, however, so any bias due to the censoring of the data would be similar between raked and unraked trees. Average forest floor fuel depth was also calculated by averaging the total depth of litter, duff, and surface fuels in each cardinal direction.

Raking treatments were assigned by randomly selecting a treatment for the first tagged tree within a plot and then alternating treatments for each successive tree to ensure an equal number of raked and unraked trees within each plot. Raked trees had all the forest floor fuels removed down to mineral soil for a 0.5 m radius around the base of the tree (Figure 4.1). The forest floor fuels that were removed were scattered broadly in the general vicinity. Trees that were within 0.5 m of either other trees or logs or stumps that could not be removed using fire rakes and loppers were not assigned a treatment and were excluded from the analysis. If other sugar pine were within 1 m of a previously tagged tree, they were not assigned a treatment and excluded from the analysis.
Figure 4.1. A raked tree prior to the prescribed burn in Redwood Canyon, Kings Canyon National Park, CA. Forest floor fuels were removed from the base of the tree to mineral soil using hand rakes and loppers to a distance of 0.5 m.

Within a month following each prescribed fire, crown scorch (percent of crown volume scorched, maximum scorch height), stem char (percent of basal stem circumference charred, max stem char height), and consumption of litter and duff were measured. Scorch and char heights were measured using an Impulse ® handheld laser rangefinder and percent crown volume scorch was visually estimated. Litter and duff consumption was measured using a duff pin placed at the uphill side of the unraked trees at the top of the litter layer prior to fire and then measuring from the top of the pin down to any unconsumed litter or duff or mineral soil following the fire. Duff pins were not installed for raked trees because all of the litter and duff at the base of the tree had been removed. The vigor of each tree and beetle activity were monitored for three summers following the fires. There were several areas within each burn unit that did not burn. Trees within these areas were excluded from the analysis reducing the final sample size to 58, 153, and 246 trees in Cabin Creek, Redwood Canyon, and Wall Spring, respectively for a total of 457 trees.

Data Analysis

Data were analyzed using the lme4 package (Bates and Maechler 2009) within the R statistical package version 2.11.1 (http://www.r-project.org). Given the hierarchal nature of the data, a generalized linear mixed effects model (GLMM) approach was used. For each model, I first compared the mixed effects model to a generalized linear model containing only fixed effects to evaluate the need for using the more complex GLMM structure. Akaike Information Criterion (AIC) was used to evaluate whether including random effects in the model was necessary and a change in AIC > 2 was used as the cut-off to indicate a meaningful change in model fit (Burnham and Anderson 2002). For all models where individual trees were used as the sample unit, a GLMM structure where plot was treated as a random effect nested within burn unit consistently had a lower AIC score than a general linear model that ignored the spatial autocorrelation of trees within plots.
The analysis focused on several questions including the effect of raking and site conditions on tree mortality, the influence of fire intensity on the effectiveness of the raking treatments, and the effect of raking on post-fire beetle activity. First, I evaluated the effects of raking and pre-fire variables including blister rust infection status, beetle activity, tree vigor, tree height, dbh, average litter, average duff, average forest floor fuel depth, and slope on post-fire tree mortality. An interaction term for each fixed effect with treatment was included in the initial model to test whether the efficacy of the raking treatment was dependent on the state of the other variables. Only pre-treatment variables were included in this model to evaluate under which fuel and forest structure conditions raking would be most effective, independent of fire effects. Change in AIC > 2 was used to evaluate whether the removal of any of the fixed effects terms and their interactions substantially reduced model fit.

The effect of fire intensity on the efficacy of the raking treatment was investigated using mixed effects logistic regression. Fire intensity is defined as “the energy released during various phases of the fire” (Keeley 2009) and crown scorch is a commonly used proxy for fire intensity (Van Wagner 1973, Williams et al. 1998, Twidwell et al. 2009). I used percent crown volume scorched instead of maximum crown scorch height or relative scorch height as our measure of fire intensity because it is viewed as a more accurate assessment of crown scorch (Peterson 1985, Twidwell et al. 2009). The effect of fire intensity on raking treatment efficacy was tested by including an interaction between raking and fire intensity category in a GLMM model. Low intensity was defined as trees whose crown’s were not scorched (n = 172), moderate intensity was defined as trees where percent crown volume scorched was between 1% and 80% (n = 181), and high intensity was defined as trees with a percent crown volume scorched > 80% (n = 104). A crown scorch value of 80% was chosen for the high intensity cut-off because this is within the range (75% - 90%) often cited as causing death in conifers (McHugh and Kolb 2003, Fowler et al. 2010b). A crown scorch value of 0% was chosen for the low intensity category cut-off because this includes all trees that burned under conditions that did not produce enough heat to damage the crown.

I investigated the effect of raking on several measures of fire effects including percent of basal stem circumference charred, maximum stem char height, and post-fire beetle activity. The linear mixed effects model residuals of percent of basal stem circumference charred displayed a highly non-normal distribution so this modeling approach was not appropriate for these data. The distribution of percent of basal stem circumference charred was u-shaped with many uncharred trees and many trees whose boles were completely charred at the base. Therefore, the average plot level percent of basal stem circumference charred was calculated and plot was used as the sample unit for this analysis instead of individual trees. A log (x + 1) transformation was applied to meet assumptions of normality. The effect of treatment on percent of the circumference of the bole that was charred was then modeled using multiple linear regression.

Maximum stem char height data displayed a highly skewed distribution with many uncharred trees. I analyzed these data in two steps where maximum stem char height was first modeled as binary (charred or uncharred) using a logistic GLMM model. The effect of raking on average maximum stem char height was then examined separately using a linear mixed effects model for only those trees that were charred (n = 273). This is an approach suggested by Fletcher et al. (2005) to model skewed data with many zeros.
The effect of the raking treatment on presence or absence of beetles was also analyzed with GLMM methods. Fixed effects in the initial model included all the pre-fire variables plus percent crown volume scorched, and an interaction of each with treatment. A drop in AIC > 2 was again used to compare model fit between alternative models and select the final model.

Results

Of the 457 sugar pine trees in the study, 150 (33 %) were dead three years following prescribed burning. Mortality of raked trees averaged 11 %, 46 %, and 24 % compared to 10 %, 43 %, and 38 % for unraked trees at Cabin Creek, Redwood Canyon, and Wall Spring, respectively. Most mortality occurred in the first two years following fire and average annual mortality had dropped to 2.3 % and 2.9 % for raked and unraked trees, respectively three years post-fire (Figure 4.2). Pre-fire tree vigor, dbh, height, blister rust infection status, beetle activity, duff depth, and litter depth were all similar between treatments (Table 4.1). The pre-fire variables that best predicted tree mortality three years post-fire included burn unit, dbh, average forest floor fuel depth, and treatment (Table 4.2). Other factors including slope, pre-fire beetle activity, pre-fire tree vigor, and presence of blister rust did not improve model fit. The best fitting model indicated that probability of mortality decreased with increasing dbh and increased with increasing average forest floor fuel depth and that there was a significant interaction between raking and average forest floor fuel depth (Table 4.3). Raking had no detectable effect on mortality when average forest floor fuel depth was low, but it significantly reduced mortality when average forest floor fuel depth was high (Figure 4.3). The predicted probability of survival of a 50 cm dbh tree at Wall Spring with an average forest floor fuel depth of 0 cm was virtually the same for raked vs. unraked trees (0.94 compared to 0.96, respectively). In contrast, when average forest floor fuel depth was 30 cm the predicted probability of survival for raked trees was 0.60 compared to 0.07 for unraked trees.
Figure 4.2. Average annual mortality of sugar pine following prescribed fire in Sequoia and Kings Canyon National Parks, CA. Health status was assessed one month, one, two, and three years following a prescribed burn. Whiskers indicate standard errors.
Table 4.1. Summary of pre-treatment conditions prior to three prescribed fires for raked and unraked sugar pine trees in Sequoia and Kings Canyon National Parks. N is sample size, Vigor is average crown vigor score, Infection is number of trees that displayed symptoms of blister rust, Beetles is number of trees that showed signs of beetle activity, Litter and Duff is the average depth of litter and duff measured at four locations at the base of each tree, respectively. No statistically significant differences were found between raked and unraked trees. Numbers in brackets are standard deviations.

<table>
<thead>
<tr>
<th>Burn</th>
<th>N</th>
<th>Treat</th>
<th>dbh (cm)</th>
<th>Height</th>
<th>Vigor</th>
<th>Infection</th>
<th>Beetles</th>
<th>Litter</th>
<th>Duff</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cabin Creek</td>
<td>28</td>
<td>Raked</td>
<td>61.7 (43.6)</td>
<td>20.8 (14.1)</td>
<td>1.4</td>
<td>5</td>
<td>0</td>
<td>4.6 (2.8)</td>
<td>5.7 (5.1)</td>
</tr>
<tr>
<td>Cabin Creek</td>
<td>30</td>
<td>Unraked</td>
<td>59.4 (47.8)</td>
<td>18.6 (13.6)</td>
<td>1.6</td>
<td>3</td>
<td>1</td>
<td>4.4 (3.0)</td>
<td>4.8 (5.6)</td>
</tr>
<tr>
<td>Redwood</td>
<td>76</td>
<td>Raked</td>
<td>58.7 (44.5)</td>
<td>25.9 (16.0)</td>
<td>1.7</td>
<td>22</td>
<td>12</td>
<td>4.1 (2.9)</td>
<td>3.4 (2.3)</td>
</tr>
<tr>
<td>Redwood</td>
<td>77</td>
<td>Unraked</td>
<td>57.6 (40.9)</td>
<td>26.2 (15.9)</td>
<td>1.5</td>
<td>22</td>
<td>8</td>
<td>3.3 (1.7)</td>
<td>3.3 (2.2)</td>
</tr>
<tr>
<td>Wall Spring</td>
<td>122</td>
<td>Raked</td>
<td>34.7 (33.8)</td>
<td>17.5 (14.7)</td>
<td>1.6</td>
<td>33</td>
<td>9</td>
<td>4.7 (3.4)</td>
<td>4.1 (2.8)</td>
</tr>
<tr>
<td>Wall Spring</td>
<td>124</td>
<td>Unraked</td>
<td>31.2 (25.0)</td>
<td>16.4 (12.5)</td>
<td>1.6</td>
<td>32</td>
<td>9</td>
<td>4.6 (3.0)</td>
<td>4.3 (3.0)</td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td>75</td>
<td>Raked</td>
<td>51.7 (14.8)</td>
<td>21.4 (4.2)</td>
<td>1.6</td>
<td>20.0</td>
<td>7.0</td>
<td>4.5 (0.3)</td>
<td>4.4 (1.2)</td>
</tr>
<tr>
<td><strong>Average</strong></td>
<td>77</td>
<td>Unraked</td>
<td>49.4 (15.8)</td>
<td>20.4 (5.1)</td>
<td>1.6</td>
<td>19.0</td>
<td>6.0</td>
<td>4.1 (0.7)</td>
<td>4.1 (0.8)</td>
</tr>
</tbody>
</table>
Table 4.2. A sample of the models tested to predict three year post-fire mortality of sugar pine using pre-fire variables. All models were mixed effects logistic regression models where the random effect was plot. Fixed effects varied by model and included raking treatment (Treat), pre-fire blister rust infection status (InfStatus), pre-fire beetle activity (BeetAct), pre-fire tree vigor (Vigor), burn unit (BurnUnit), average litter depth (Avg.Litter), average duff depth (Avg.Duff), average forest floor fuel depth (Avg.Fuel), dbh, and slope. The model in bold was selected as the final model based on AIC. Log L is the Log likelihood value, K is the number of parameters, and AIC is the Akaike Information Criterion score for each model.

<table>
<thead>
<tr>
<th>Fixed effects variables</th>
<th>Log L</th>
<th>K</th>
<th>AIC</th>
<th>Δ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Treat×(InfStatus+BeetAct+Vigor+BurnUnit+Avg.Litter+Avg.Duff+DBH+Slope)</td>
<td>-212.6</td>
<td>24</td>
<td>475.2</td>
<td>22.0</td>
</tr>
<tr>
<td>Treat×(InfStatus+BeetAct+Vigor+BurnUnit+Avg.Fuel+DBH+Slope)</td>
<td>-212.3</td>
<td>22</td>
<td>470.7</td>
<td>17.5</td>
</tr>
<tr>
<td>Treat×(Avg.Litter+Avg.Duff)+BurnUnit+DBH+InfStatus+Vigor+BeetAct</td>
<td>-218.2</td>
<td>14</td>
<td>466.3</td>
<td>13.1</td>
</tr>
<tr>
<td>Treat×Avg.Fuel+BurnUnit+DBH+InfStatus+Vigor+BeetAct</td>
<td>-217.8</td>
<td>12</td>
<td>461.7</td>
<td>8.5</td>
</tr>
<tr>
<td>Treat×(Avg.Litter+Avg.Duff)+BurnUnit+DBH</td>
<td>-219.2</td>
<td>9</td>
<td>458.4</td>
<td>5.2</td>
</tr>
<tr>
<td>Treat×Avg.Duff+Avg.Litter+BurnUnit+DBH</td>
<td>-220.6</td>
<td>8</td>
<td>459.2</td>
<td>6.0</td>
</tr>
<tr>
<td>Treat×Avg.Duff+Avg.Litter+BurnUnit+DBH</td>
<td>-221.5</td>
<td>7</td>
<td>459.0</td>
<td>5.8</td>
</tr>
<tr>
<td><strong>Treat×Avg.Fuel+BurnUnit+DBH</strong></td>
<td><strong>-218.6</strong></td>
<td>7</td>
<td><strong>453.2</strong></td>
<td>-</td>
</tr>
<tr>
<td>Treat+Avg.Fuel+BurnUnit+DBH</td>
<td>-220.8</td>
<td>6</td>
<td>455.7</td>
<td>2.5</td>
</tr>
</tbody>
</table>
When only large trees with a dbh ≥ 50 cm were considered, the parameters that best predicted three year post-fire mortality were slightly different and included dbh, average litter depth, and average duff depth. When all trees were used, models that included average forest floor fuel depth consistently had lower AIC scores than models with separate measures of litter and duff depth when the other fixed effects were the same (Table 4.2). For large trees, however, substituting average duff depth for average forest floor fuel depth in the final model resulted in a lower AIC score (ΔAIC = 2.84).

**Table 4.3.** Mixed effects logistic regression of factors that influenced sugar pine mortality three years post fire in Sequoia and Kings Canyon National Parks, CA. Fixed effects were burn unit (BurnUnit), dbh, average forest floor fuel depth (Avg.Fuel), raking treatment (Unraked), and the interaction between average forest floor fuel depth and raking treatment. Plot was treated as a random effect. Estimate is the coefficient from the GLMM, with associated standard error (SE) and p-value.

<table>
<thead>
<tr>
<th>Fixed effects</th>
<th>Estimate</th>
<th>SE</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>(Intercept)</td>
<td>2.191</td>
<td>0.829</td>
<td>0.008</td>
</tr>
<tr>
<td>Redwood Canyon</td>
<td>-3.170</td>
<td>0.853</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Wall Spring</td>
<td>-1.237</td>
<td>0.868</td>
<td>0.154</td>
</tr>
<tr>
<td>Avg.Fuel</td>
<td>-0.076</td>
<td>0.040</td>
<td>0.055</td>
</tr>
<tr>
<td>Unraked</td>
<td>0.543</td>
<td>0.523</td>
<td>0.300</td>
</tr>
<tr>
<td>DBH</td>
<td>0.034</td>
<td>0.005</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Avg.Fuel:Unraked</td>
<td>-0.119</td>
<td>0.056</td>
<td>0.034</td>
</tr>
</tbody>
</table>
Figure 4.3. Effect of average forest floor fuel depth on predicted probability of survival of raked vs. unraked sugar pine three years following prescribed fire in Sequoia and Kings Canyon National Parks, CA using logistic mixed effects regression. Predicted probabilities assume site is Wall Spring and dbh is 50 cm.

Raking was more effective at reducing mortality under conditions of moderate fire intensity compared to trees that experienced low or high intensity fire (Table 4.4). Raking significantly increased the proportion of trees that survived by 9.6 % for trees that burned under moderate intensity (1 % - 80 % crown volume scorched) compared to a small, insignificant increase of 2.9 % when fire intensity was low (0 % crown volume scorched), or even a reduction of 4.2 % in survival when fire intensity was high (81 % - 100 % crown volume scorched).
Table 4.4. Effect of fire intensity on raking treatment efficacy for sugar pine in Sequoia and Kings Canyon National Parks, CA. Three year post-fire mortality was modeled using logistic mixed effects regression. Trees were assigned to a fire intensity category based on percent crown volume scorched. Low Intensity = Scorch% (percent crown volume scorched) = 0 (n = 172), Moderate Intensity = Scorch% between 1 and 79 (n = 181), or High Intensity = Scorch% > 80% (n = 104). Fixed effects in the models were burn unit, raking treatment, and fire intensity category, and an interaction between raking and fire intensity category. Plot was included as a random effect. Estimate is the coefficient from the GLMM, with associated standard error (SE) and p-value.

<table>
<thead>
<tr>
<th>Fixed Effects</th>
<th>Estimate</th>
<th>SE</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>-1.917</td>
<td>0.906</td>
<td>0.034</td>
</tr>
<tr>
<td>Redwood Canyon</td>
<td>-1.101</td>
<td>0.672</td>
<td>0.101</td>
</tr>
<tr>
<td>Wall Spring</td>
<td>-1.941</td>
<td>0.654</td>
<td>0.003</td>
</tr>
<tr>
<td>Unraked</td>
<td>1.141</td>
<td>0.854</td>
<td>0.181</td>
</tr>
<tr>
<td>Low Intensity</td>
<td>6.563</td>
<td>0.920</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Moderate Intensity</td>
<td>4.861</td>
<td>0.795</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Unraked:Low Intensity</td>
<td>-1.393</td>
<td>1.100</td>
<td>0.205</td>
</tr>
<tr>
<td>Unraked:Moderate Intensity</td>
<td>-1.940</td>
<td>0.922</td>
<td>0.035</td>
</tr>
</tbody>
</table>

Raking significantly reduced the occurrence of stem charring, but the effectiveness of raking was dependent on average forest floor fuel depth (Figure 4.4). The occurrence of stem charring was also dependent on burn unit, dbh, and white pine blister rust infection status (Table 4.5). Interestingly, when only trees that were charred were considered, there was no significant difference between char heights due to raking, or any other factors (linear mixed effects regression, raking coefficient = -0.195, SE = 0.184, p = 0.292), illustrating that the primary effect of raking on stem char was to reduce the likelihood of charring, but not the maximum height of the charring if it did occur. While char height was not affected by raking, average percent of basal stem circumference charred was significantly lower in raked trees (19.4 %) compared to unraked trees (59.2 %; linear regression, raking coefficient = 1.034, SE = 0.401 p = 0.013).

Post-fire beetle activity was strongly associated with pre-fire beetle activity, burn unit, and raking and was significantly lower in raked trees compared to unraked trees (Table 4.6). Beetle activity increased every year after fire, though at a decreasing rate (Figure 4.5). Raking resulted in a 25 %, 22 % and 29 % drop in beetle activity one, two, and three years post-fire, respectively. Redwood Canyon displayed significantly higher beetle activity two years post-fire but beetle activity was similar among burn units one and three years post-fire (Table 4.6).
Figure 4.4. Effect of average forest floor fuel depth on the likelihood of stem char of raked vs. unraked sugar pine three years following prescribed fire in Sequoia and Kings Canyon National Parks, CA using logistic mixed effects regression. Predicted probabilities assume site is Cabin Creek, dbh is 50 cm, and trees displayed no signs of blister rust infection.
Table 4.5. Mixed effects logistic regression results of factors that influenced the occurrence of stem charring on sugar pine three years post-fire in Sequoia and Kings Canyon National Parks, CA. Fixed effects in the model were burn unit (BurnUnit), white pine blister rust infection status (InfStatus), raking (Unraked), average forest floor fuel depth (Avg.Fuel), dbh, an interaction between raking and forest floor fuel depth, raking and burn unit, and raking and white pine blister rust infection. Plot was treated as a random effect. Estimate is the coefficient from the GLMM, with associated standard error (SE) and $p$-value.

<table>
<thead>
<tr>
<th>Fixed Effects</th>
<th>Estimate</th>
<th>SE</th>
<th>$p$-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>-1.732</td>
<td>0.779</td>
<td>0.026</td>
</tr>
<tr>
<td>Redwood Canyon</td>
<td>0.694</td>
<td>0.847</td>
<td>0.413</td>
</tr>
<tr>
<td>Wall Spring</td>
<td>-0.648</td>
<td>0.902</td>
<td>0.472</td>
</tr>
<tr>
<td>Unraked</td>
<td>-1.264</td>
<td>0.859</td>
<td>0.141</td>
</tr>
<tr>
<td>Avg.Fuel</td>
<td>0.031</td>
<td>0.035</td>
<td>0.376</td>
</tr>
<tr>
<td>DBH</td>
<td>0.012</td>
<td>0.004</td>
<td>0.003</td>
</tr>
<tr>
<td>InfStatus</td>
<td>0.286</td>
<td>0.400</td>
<td>0.475</td>
</tr>
<tr>
<td>Unraked:Redwood Canyon</td>
<td>2.622</td>
<td>0.846</td>
<td>0.002</td>
</tr>
<tr>
<td>Unraked:Wall Spring</td>
<td>4.525</td>
<td>0.909</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Unraked:Avg.Fuel</td>
<td>0.254</td>
<td>0.074</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Unraked:InfStatus</td>
<td>-1.953</td>
<td>0.657</td>
<td>0.003</td>
</tr>
</tbody>
</table>
Figure 4.5. Average proportion of raked and unraked sugar pine trees that displayed signs of beetle activity following prescribed fire in Sequoia and Kings Canyon National Parks, CA. Whiskers indicate standard errors.
Table 4.6. Effect of raking treatment on beetle activity one, two, and three years post-fire for sugar pine in Sequoia and Kings Canyon National Parks, CA. Beetle Activity was modeled using mixed effects logistic regression. Fixed effects were burn unit (BurnUnit), raking treatment (Unraked), and pre-fire beetle activity (Beetles). Plot was treated as a random effect. Estimate is the coefficient from the GLMM, with associated standard error (SE) and p-value.

<table>
<thead>
<tr>
<th>Fixed Effects</th>
<th>Year 1</th>
<th>Year 2</th>
<th>Year 3</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Estimat</td>
<td>SE</td>
<td>p-value</td>
</tr>
<tr>
<td>Intercept</td>
<td>-1.436</td>
<td>0.427</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Redwood Canyon</td>
<td>0.669</td>
<td>0.503</td>
<td>0.183</td>
</tr>
<tr>
<td>Wall Spring</td>
<td>-0.599</td>
<td>0.520</td>
<td>0.249</td>
</tr>
<tr>
<td>Unraked</td>
<td>0.552</td>
<td>0.231</td>
<td>0.017</td>
</tr>
<tr>
<td>Beetles</td>
<td>2.198</td>
<td>0.406</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>
Discussion

Raking did not detectably reduce mortality when all trees were considered; however, it was effective at reducing mortality under specific conditions. Raking was most beneficial in areas where high levels of forest floor fuels had accumulated. This is consistent with previous studies that found manipulating fuel depths, particularly duff depth, was an effective way for reducing cambial injury following prescribed fire (Hood et al. 2007b, Laudenslayer et al. 2008). While consumption of the duff layer has been shown to be an important factor controlling tree mortality due to its sustained release of heat at the base of the tree (Ryan and Frandsen 1991, Varner et al. 2007), I found a better relationship between mortality and average forest floor fuel depth (combined average depth of duff, litter, and surface fuels) than average duff depth alone. One possible explanation for this is that there were many small trees in our study as our minimum dbh threshold for raking was 10 cm. Therefore the combustion of deep litter layers and small surface fuels, as well as duff, may have created enough heat to damage the cambium of smaller trees with thinner bark, increasing the importance of the litter layer and small surface fuels compared to studies that focused only on large trees. Indeed, when only large trees (dbh ≥ 50 cm) were considered, average duff depth was a better predictor of mortality than average forest floor fuel depth. Some of the effect of average forest floor fuel depth on mortality may have been better explained by larger spatial scale variation in fuel loads and ladder fuels that were not captured at the individual tree scale. However, fuel loads varied widely among plots, even within the same burn unit, and some of this plot level variation in fuel loads would have been captured as plot level random effects in the models.

Along with average forest floor fuel depth, the effectiveness of the raking treatment varied by site. Each site was different in respect to topography, exposure, time since last burn, and timing of the burn. These differences led to burn severities that varied substantially among sites and greatly influenced sugar pine mortality rates. Where fire intensity was high, as at Redwood Canyon, the usefulness of the raking treatment was reduced, resulting in little difference in mortality between raked and unraked trees. The main source of mortality following fire is often attributed to crown scorch (Ryan and Reinhardt 1988, Keyser et al. 2006), and raking does little to mitigate this. When fire intensity was low, most trees survived, regardless of whether the duff and litter were removed from the base and the raking treatment provided little benefit, as at Cabin Creek. Several studies that used fuels treatments under low intensity burns also found no difference in mortality rates between treated and untreated trees (Swezy and Agee 1991, Fulé et al. 2002). The largest difference in mortality between raked and unraked trees in this study was at Wall Spring, which displayed moderate fire intensity compared to the other two sites. The interaction between treatment efficacy and fire intensity provides a mechanism to explain the mixed results of the effect of raking on reducing tree mortality that have been reported in the past. By measuring multiple fires that differed in fire intensity, this study demonstrated that treatment effects on mortality are not uniform across different fire intensities and it is important to account for this factor when assessing the effects of raking treatments on mortality.

There were no significant interactions between the raking treatment and factors associated to pre-fire tree health, including beetle activity, blister rust infection status, and crown vigor in terms of the effect of raking on post-fire tree mortality. Therefore, the effectiveness of the raking treatment did not vary significantly due to pre-fire tree health. Other research, however, has found that tree health, measured as annual growth rate, was a significant predictor of mortality following fire (van Mantgem et al. 2003). More research is needed to identify the extent to which
pre-fire tree health controls post-fire mortality and how this may influence the effectiveness of raking.

Raking significantly reduced the occurrence of stem charring, and the magnitude of the effect of the raking treatment was influenced by blister rust infection status, average forest floor fuel depth, and burn unit. One of the symptoms of blister rust is cankers that often exude large quantities of pitch. The interaction of blister rust infection status and raking may be caused by unraked infected trees having highly flammable pitch on the main bole leading to a higher likelihood of stem charring. By reducing the occurrence of stem charring, raking can reduce mortality rates directly by reducing cambium necrosis as well as indirectly by reducing a tree’s susceptibility to beetle attacks (Ferrell 1996, Bradley and Tueller 2001). Stem charring may facilitate beetle attacks by reducing tree defenses such as bark thickness and resin production immediately following fire (Wallin et al. 2003, Yongblood et al. 2009, Fettig et al. 2010). While most beetle activity that was observed occurred in the charred areas of the lower bole, possibly by red turpentine beetle (Dendroctinus valens LeConte), which is rarely a direct cause of mortality, beetle attack often occurs by multiple species simultaneously (Breece et al. 2008, Wallin et al. 2008) and the presence of D. valens has been found to be associated with western pine beetle (Dendroctinus brevicomis LeConte) in ponderosa pine (Perrakis and Agee 2006).

Raking reduced beetle activity significantly following fire, which may lead to further differences in mortality between raked and unraked trees in the future as trees recently attacked by beetles continue to die.

Raking away forest floor fuels at the base of sugar pine reduced the occurrence of stem charring and beetle activity following prescribed fire. These factors contributed to a reduction in mortality associated with raking treatments. Raking was particularly effective when forest floor fuel depth was high. While dbh was a significant predictor of average duff depth, in our study, the correlation between them was low (Adjusted \( R^2 = 0.27, 0.03, \) and 0.06 at Cabin Creek, Redwood Canyon, and Wall Spring, respectively). It is often the large trees that accumulate these deep duff and litter layers at the base without the occurrence of fire because of the large amount of bark and needles that they drop annually. These are also the trees that managers are often most concerned about losing during prescribed fire. Targeting large trees for raking treatments may therefore be a simple and effective way to increase probability of survival.

While raking can be used on the scale of some prescribed burns, the time and resources that would be needed to apply it at a landscape scale are not available in most situations and other methods of reducing sugar pine mortality should be considered. Also, there are situations where burn severity is expected to be relatively high, or fuel levels are already low, reducing the effectiveness of raking, and alternative treatment options would be more appropriate. Other possible treatment options include pruning (Brown et al. 2004), thinning (Stephens and Moghaddas 2005), modifying ignition techniques (McCallum 2009), changing season of burn (Swezy and Agee 1991), and shortening the time interval between burns (He and Mladenoff 1999). Pruning raises the height to live crown, which can reduce the potential for crown fire spread (Keyes and O’Hara 2002). Reducing fuels through thinning treatments when combined with prescribed fire has been well documented as an effective method for reducing mortality (Harrod et al. 2009, Schwilk et al. 2009, Stephens et al. 2009). However this method may not be an option in many areas like national parks and wilderness areas.

Thinning, pruning, and raking all require a substantial investment of resources to carry out and will often not be realistic options at the landscape level. Therefore it is important to also consider simple changes in the timing and methods of prescribed fire ignition as potential
methods of reducing mortality at a larger scale without required additional treatments. Burning during the winter when trees are dormant has been shown to reduce the effects of crown scorch and burning under wetter fuel conditions can reduce fire intensity (Swezy and Agee 1991, Harrington 1993, Thies et al. 2005). However, there may be unintended ecological consequences for burning outside of the historical fire season, though these effects are often minor when compared to the alternative of no action at all (Knapp et al. 2009). In addition, burning under conditions when duff has high moisture content can result in low duff consumption, so that fuel reduction goals are not met (Hille and Stephens 2005). For restoration purposes, the benefits of reintroducing fire often outweigh any potential negative impacts for burning outside of the normal fire season. Changing firing patterns offers some promise for reducing mortality as well. Backing fires often burn at lower intensity than heading fires (Finney 2001). However, backing fires may produce higher levels of duff and litter consumption than heading fires due to a slower rate of spread leading to a possible trade-off between fire intensity and fire severity depending on ignition method. Once fire has been re-introduced to an area, maintaining a shorter fire return interval can also reduce fire intensity and result in increased survival, especially of the larger trees.

Raking treatments can be an efficient and relatively quick method for decreasing sugar pine mortality following prescribed fire. Raking rarely took longer than five minutes per tree and can be performed with the same tools that are commonly used to prepare the site for a prescribed fire. Raking time increased with increasing tree size and fuel depth. Hood et al. (2007b) found that raking took 16 minutes/person on average for large Jeffrey pine in California and raking time increased with duff depth. Raking may not be appropriate under all conditions and Swezy and Agee (1991) and Kolb et al. (2007) have suggested that raking may negatively affect tree health by killing fine roots within the duff layer. To avoid this, raking has sometimes been carried out at least a year prior to the fire to allow trees to recover from the loss of fine roots removed by raking and develop new roots deeper in the soil (Hood 2010). This may be less of a concern in ponderosa and Jeffrey pine forests where Noonan-Wright et al. (2010) found no negative effects of raking on growth rate and mortality in the absence of fire.

With relatively little additional cost or training of crews, raking may be a viable option for managers concerned about sugar pine survival following prescribed fire. However, is the potential increase in survival worth the additional effort expended or increased cost? In this study, the average size of the three prescribed fires was 166 ha and averaged 18 treatable (>10 cm dbh, could be raked) sugar pine per ha resulting in an estimated 3000 sugar pine per fire. If it is assumed that average forest floor fuel depth must be at least 15 cm for raking to cause a meaningful change in probability of survival, this limits the pool of trees that would be considered for treatment. I found raked trees with average forest floor fuel depth of 15 cm were 25 % more likely to survive than unraked trees and roughly 10 % of the trees in our study met this criterion. The average fuel depth of these trees was 18.2 cm. Fire intensity is inherently patchy, however, and trees often burn under very low or very high fire intensity, where the effect of the treatment does not affect mortality. For example, 228 of our 457 trees (50 %) had crown volume scorched values of either 0 % or 100 %. If we assume that half of the estimated 300 trees being considered for raking will burn under moderate fire severity, we can predict that the treatment will be effective for 150 of them. Our model predicts a 50 % greater probability of survival for raked vs. unraked trees (85 % vs. 35 %) when average fuel depth is 20 cm. This implies survival of 128 of 150 trees compared to only 53 surviving without the raking treatment. Therefore, of the 300 trees that we would have chosen to treat, we can estimate raking will
prevent the death of 75. It will depend on the specific objectives of the manager to decide whether this potential increase in survival is worth the added cost and effort of raking. Managers should also consider the specific conditions of their site and the expected fire severity to evaluate the merits of raking. Do to the host of interacting factors that influence the efficacy of raking on reducing mortality, the impact of raking on post-fire mortality will likely continue to be mixed, depending on the specific site conditions and fire behavior patterns under which raking is applied (Fowler et al. 2010a, Hood 2010).

Conclusions

Removing fuels from the base of the tree was an effective means of reducing sugar pine mortality following prescribed fire under specific conditions. Raking had little effect on mortality when average forest floor fuel depth was low, but significantly reduced mortality when it was high. In addition, raking had little effect on mortality when fire intensity was low, as most trees survived regardless of raking. When fire intensity was high, raking did little to mitigate mortality. However, when fire intensity fell between these two extremes, raking was an effective method for reducing post-fire mortality. By measuring the effects of raking on mortality across a wide range of burn severities over multiple prescribed fires, I was able to demonstrate this interaction between fire intensity and the efficacy of raking treatments and provide a mechanism to explain the contradictory results of previous studies. Along with the direct effect of raking on reducing stem charring, it may also help prevent additional post-fire mortality from beetle attacks as beetle activity was significantly lower in raked trees compared to unraked trees following fire. The differences in mortality between raked and unraked trees may therefore become even larger over the next few years as beetle attacked trees continue to die (Agee 2003).

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I would like to thank N. Stephenson, J. van Wagtendonk, and two anonymous reviewers for the insightful comments they provided on earlier drafts. I would also like to thank the National Park Service for providing logistical support, training, and use of their facilities for this research. Funding for this work was provided by the USGS Park Oriented Biological Support Grant 06-72. This work is a contribution from the Western Mountain Initiative (a USGS global change research project), and from the Cordillera Forest Dynamics Network (CORFOR). Any use of trade names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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