DISSERTATION

MAINTAINING FUEL TREATMENTS WITH PRESCRIBED FIRE IN PONDEROSA PINE FORESTS OF THE BLACK HILLS, SOUTH DAKOTA

Submitted by

Michael A. Battaglia

Department of Forest, Rangeland, and Watershed Stewardship

In partial fulfillment of the requirements
For the Degree of Doctor of Philosophy
Colorado State University
Fort Collins, Colorado
Summer 2007

COLORADO STATE UNIVERSITY

April 27, 2007

WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY MICHAEL A. BATTAGLIA ENTITLED MAINTAINING FUEL TREATMENTS WITH PRESCRIBED FIRE IN PONDEROSA PINE FORESTS OF THE BLACK HILLS, SOUTH DAKOTA BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

Committee on Graduate Work

Monique E. Rocca

Mary Stromberger

Adviser: Frederick W. Smith

Co-Adviser: Wayne D. Shepperd

Department Head: Melinda Laituri

ABSTRACT OF DISSERTATION

MAINTAINING FUEL TREATMENTS WITH PRESCRIBED FIRE IN PONDEROSA PINE FORESTS OF THE BLACK HILLS, SOUTH DAKOTA

Recent wildfires in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) forests have increased efforts to create forest structures that reduce the risk of crown fire. In the Black Hills, these fuel-reduction treatments often result in a new cohort of ponderosa pine regeneration. If no action is taken, the efficacy of these fuel treatments eventually diminishes as the regeneration grows and creates a ladder fuel complex. In this dissertation, I examine the utility of using prescribed fire to control this regeneration. I also explore if restoration of historical forest structure would result in reduced crown fire risk.

Models that predict ponderosa pine regeneration mortality after dormant season fires in the Black Hills were developed. The models showed that tree size, crown damage, ground char severity, and basal char severity were important factors in predicting mortality. With these models, I examined the temporal susceptibility of regeneration to prescribed fire. I then examined the use of prescribed fire to maintain fuel treatments by evaluating the temporal dynamics of regeneration, surface fuel accumulation, and associated fire behavior under prescribed fire conditions for a 120-year *time since fire* chronosequence. Current fire prescriptions and fuel loads were adequate to maintain low densities of seedlings (<90 cm tall) if burns occurred every 10 to 15 years, but bigger regeneration would not be susceptible. Burn intervals exceeding 15 years allowed regeneration to attain sizes that reduced susceptibility to fire. Flame lengths exceeding 1-2 meters were required to obtain significant mortality. Achieving these flame lengths will require burning under drier weather conditions or fuel load augmentation.

Finally, I explore the idea that restoration of Black Hills historical forest structure results in reduced crown fire risk. Potential fire behavior of these forest structures was modeled under various weather conditions and fire return intervals. Simulation results suggested that the historical Black Hills landscape was shaped by a mixed-severity fire regime, indicating that restoration of some historical Black Hills forest structures would be in conflict with the goals of fuel-reduction treatments. Furthermore, the simulations highlighted the importance of maintaining low densities of regeneration to limit crown fire initiation during a wildfire.

Michael A. Battaglia
Department of Forest, Rangeland,
and Watershed Stewardship
Colorado State University
Fort Collins, Colorado 80523
Summer 2007

ACKNOWLEDGEMENTS

It is hard to put into the words the gratitude that I feel toward the many people that helped me along the past several years. I am especially grateful to Skip Smith and Wayne Shepperd for guiding me through my dissertation. Both of them gave me guidance when needed, but also allowed me to figure things out on my own. Their mentoring allowed me to grow confident in my abilities and knowledge. It was a privilege to work with them both. I also thank my committee members, Monique Rocca and Mary Stromberger for providing insightful feedback and encouragement throughout my dissertation work.

I thank the staff of the Mystic Ranger District of the Black Hills National Forest for supporting this dissertation research. Without their willingness to allow me to put research plots on their burn units, this dissertation would never succeeded. I appreciate the time they took to show me potential sites, provide me with maps, call me when they burned, and for burning the research plots. I am especially grateful to Terry Tompkins, Jay Kurth, Gale Gire, and Gwen Lipp for all of the time and support they provided this study. I also want to thank Blaine Cook for his continued interest in this project and other research activities on the Black Hills National Forest. Cody Wienk and and Andy Thorstenson of Wind Cave National Park were also very helpful in providing sites on National Park lands.

I thank Mike Peterkin, Jay Aylward, and Vicky Williams for their help during the summer field seasons. I am also grateful for all the help Lance Asherin provided me throughout my time in Fort Collins and especially for his help during the prescribed burn monitoring. Both Brian Henley and Zach Glover helped with the monitoring setup of the prescribed fires. I am thankful to Charity Weaver who helped me when she had time during the summers and in the lab during the school year.

I owe my officemate Tara Keyser so much appreciation. Her diligence kept me motivated. I will miss our daily discussions and stories. She provided me with insightful feedback and often encouraged me when I needed it most. I thank my friends on the A-wing of NESB for their constant support, encouragement, and lunch time discussions. I especially thank Molly Cavaleri for all of her many talents: SAS, SigmaPlot, powerpoint color schemes, and her ability to provide insightful comments on my research even though fire was outside her field.

I thank the staff of the Forest Vegetation Simulator for their excellent help line. I especially thank Stephanie Rebain and Chad Keyser for their patience and helpfulness when FVS would give me trouble.

Last but not least, I thank my wife Chrissy and daughter Ella. Words can not even begin to express the gratitude I have for Chrissy's sacrifice to essentially be a single-mom during my absence during the summer months. Without her support, I would not be where I am today. I especially thank both of them for putting up with me for the last several months of this journey. It was a hard time for all of us and I am encouraged that we survived it.

DEDICATION

To Chrissy and Ella

TABLE OF CONTENTS

CHAPTER 1: INTRODUCTION

The increase in the number and acres burned by wildland fires in the western United States over the past decade has prompted the federal government to develop a comprehensive strategy to reduce wildland fire risks to communities and the environment (Western Governors Association 2002, Stephens and Ruth 2005). Part of this strategy is 1) to reduce the risk of unintended wildland fire by reducing hazardous fuels; and 2) to restore fire-adapted ecosystems in a way that will provide sustainable environmental, social, and economic benefits. To implement this strategy, the National Fire Plan requires that sound science be used as a basis for management. Therefore, the first goal of my dissertation was to provide managers with scientifically based data that would promote the use of prescribed fire to maintain fuel treatment effectiveness in ponderosa pine (*Pinus ponderosa* var. *scopulorum* Dougl. ex Laws.) forests of the Black Hills. My second goal was to identify historical stand structure characteristics that could be incorporated into fuel treatment prescriptions that would reduce crown fire risk and overstory mortality while concurrently restoring some elements of historical forest structure.

The reintroduction of fire into the Black Hills is an important first step in the restoration of this fire-adapted ecosystem. Historically, fire was a keystone ecological process that shaped the structure and composition of ponderosa pine communities in the Black Hills of South Dakota. On average, these fires burned along the ponderosa pine-prairie ecotone every 10 to 20 years and every 20 to 35 years in the interior ponderosa pine forests (Brown and Sieg 1996, 1999, Brown *et al.* 2000, Brown 2003). These periodic fires reduced surface fuels, regulated forest densities and age structure, and shaped tree morphology by pruning lower branches. However, since EuroAmerican settlement in the 1870's, most fire has been excluded and suppressed.

Fire suppression combined with the extensive management of the Black Hills with shelterwood harvesting has resulted in large contiguous areas of dense and young even-aged forests (DeBlander 2002). The increase in the densities of smaller size classes has decreased the canopy base height of current forests and has increased the canopy bulk density. More available fuel in the overstory at lower heights has led to a situation where surface fires can be transported into the overstory trees and initiate crown fire at lower wind speeds. Similar changes have been quantified in ponderosa pine forests of Arizona and have resulted in a shift in the predominance of fire behavior from surface fires to crown fires (Fulé *et al.* 2002, Fulé *et al.* 2004). This increase in crown fire behavior in ponderosa pine forests has prompted managers to establish fuel-reduction treatments in an effort to reduce the risk of crown fire.

Recent treatments to reduce crown fire hazard in the Black Hills National Forest include mechanical treatments and prescribed fire to reduce surface fuel loadings and crown fire potential by eliminating ladder fuels and reducing canopy bulk density. The typical treatment is to thin the overstory basal areas to around 11 to 14 m^2 ha^{-1} and prescribe burn the area in the dormant season to reduce the surface fuels that have accumulated. While the initial treatment may reduce crown fire risk in the short term, the long-term outcome of these treatments is unknown. Under the proposed treatment conditions, prolific ponderosa pine natural regeneration is likely, up to several thousand seedlings per hectare, due to abundant seed crops and favorable climatic conditions during the growing season (Shepperd and Battaglia 2002). This regeneration will quickly grow and increase canopy bulk density, decrease canopy base height, and subsequently increase the risk of crown fire. Therefore, sustainability of forest stand structures that are more resilient to crown fire will require the control of ponderosa pine regeneration.

The successful use of prescribed fire to control ponderosa pine regeneration in the Black Hills will require specific knowledge about the levels of fire-related damage that is associated with mortality. Current ponderosa pine mortality models were developed using data collected from trees > 5 cm dbh (diameter at breast height; 1.37 m) with datasets that had average diameters exceeding 20 cm dbh (Wyant *et al.* 1986, Saveland and Neuenschwander 1990, Regelbrugge and Conard 1993, Stephens and Finney 2002, McHugh and Kolb 2003, Keyser *et al.* 2006, Sieg *et al.* 2006, Thies *et al.* 2006). Use of these models to predict seedling (<137 cm tall) and sapling (0.25 to 10 cm dbh) mortality under prescribed burning conditions may produce erroneous predictions. In **Chapter 2**, I develop logistic regression models that predict post-fire mortality of ponderosa pine seedlings and saplings based on regeneration size and direct fire effects for the Black Hills. This process further allowed me to identify factors that contribute to fire-induced mortality of these small trees.

With information about the susceptibility of ponderosa pine seedling and saplings to fire, I examined the utility of using prescribed fire as a technique to control regeneration densities to promote fuel treatment effectiveness over time in **Chapter 3**. First, I quantified the time it took for a seedling or sapling to reach a certain height and/or diameter. This information was important for two reasons. First, knowledge of height growth rates allows managers to determine how long it takes before regeneration connects the surface fuel complex to the overstory fuels. Second, since the susceptibility of ponderosa pine to fire may change with tree size, it is important to know when to use fire to effectively cause mortality. My second objective in **Chapter 3** was to quantify the post-fire fuel accumulation rates over a 120-year chronosequence in the Black Hills. Fires cannot burn without fuel, so understanding when there is enough fuel to carry a fire and produce the fire intensity required to cause regeneration mortality is important. With the fuel accumulation data, I then used BehavePlus (Andrews *et al.* 2005) to estimate the potential flame lengths that would occur over time. These flame lengths were combined with the seedling/sapling susceptibility data to estimate when prescribed fire could be used to cause mortality in ponderosa pine regeneration and when other treatments will be required to successfully maintain fuel treatment effectiveness.

In **Chapter 4**, I explore the idea that restoration of Black Hills historical forest structure results in reduced crown fire risk. In southwestern ponderosa pine ecosystems, restoration of historical ponderosa pine forest structure has been demonstrated to be compatible with fuel reduction treatment goals (Fulé *et al.* 2001, Fulé *et al.* 2002, Fulé *et al.* 2004). However, the appropriateness of applying the southwestern restoration model to ponderosa pine forests of other geographical regions has been questioned (Shinneman and Baker 1997, Baker and Ehle 2001, Shepperd and Battaglia 2002, Brown 2003, Veblen 2003, Baker *et al.* 2006) because of the differences in fire frequency, fire behavior, and historical forest structure between the regions. In **Chapter 4**, I model the potential fire behavior of published reconstructed early settlement Black Hills ponderosa pine forest structures (Brown and Cook 2006) under various weather conditions and fire return intervals. I use this information to explore how fire frequency and subsequent fire behavior affect stand development. I also use this information to identify historical stand characteristics that were associated with low crown fire risk. If any do exist, these characteristics could be incorporated into fuel treatment prescriptions to reduce the risk of crown fire while concurrently restoring some elements of historical forest structure.

LITERATURE CITED

Andrews, P.L., Bevins, C.D., Seli, R.C., 2005. BehavePlus fire modeling system, version 3.0: User's Guide. Gen. Tech. Rep. RMRS-GTR-106WWW Revised. Ogden, UT: Department of Agriculture, Forest Service, Rocky Mountain Research Station. 144 p.

Baker, W.L., Veblen, T.T., Sherriff, R.L. 2007. Fire, fuels and restoration of ponderosa pine-Douglas fir forests in the Rocky Mountains, USA. Journal of Biogeography 34(2): 251-269.

Baker, W.L., Ehle, D. 2001. Uncertainty in surface-fire history: The case of ponderosa pine forests in the western United States. Can. J. For. Res. 31: 1205-1226.

Brown, R.T., Agee, J.K., Franklin, J.F. 2004. Forest restoration and Fire: Principles in the Context of Place. Conservation Biology 18(4): 903-912.

Brown, P.M. and Cook, B. 2006. Early settlement forest structure in Black Hills ponderosa pine forests. Forest Ecology and Management 223: 284-290.

Brown, P.M. 2003. Fire, climate, and forest structure in Black Hills ponderosa pine forests. Dissertation. Colorado State University, Ft. Collins, CO. 103 p.

Brown, P. M.; Ryan, M. G.; Andrews, T. G. 2000. Historical surface fire frequency in ponderosa pine stands in research natural areas, Central Rocky Mountains and Black Hills, USA. Natural Areas Journal. 20 (2): 133-139.

Brown, P. M.; Sieg, C. H. 1999. Historical variability in fire at the ponderosa pine-Northern Great Plains prairie ecotone, southeastern Black Hills, South Dakota. Ecoscience. 6(4): 539-547.

Brown, P. M.; Sieg, C. H. 1996. Fire history in interior ponderosa pine forests of the Black Hills, South Dakota, USA. International Journal of Wildland Fire. 6: 97-105.

DeBlander. L.T. 2002. Forest resources of the Black Hills National Forest. USDA Forest Service, Rocky Mountain Research Station. 16 p.

Fulé, P.Z., Crouse, J.E., Cocke, A.E., Moore, M.M., Covington, W.W. 2004. Changes in canopy fuels and potential fire behavior 1880-2040: Grand Canyon, Arizona. Ecol. Mod. 175: 231-248.

Fulé, P.Z., Covington, W.W., Smith, H.B., Springer, J.D., Heinlein, T.A., Huisinga, K.D., Moore, M.M. 2002. Comparing ecological restoration alternatives: Grand Canyon, Arizona. For. Ecol. Mgmt. 170: 19-41.

Fulé, P.Z., Waltz, A.E.M., Covington, W.W., Heinlein, T.A. 2001. Measuring forest restoration effectiveness in hazardous fuels reduction. J. For. 99(11): 24-29.

Keyser T.L., Smith, F.W., Lentile L.B., Shepperd, W.D., 2006. Modeling postfire mortality of ponderosa pine following a mixed-severity wildfire in the Black Hills: The role of tree morphology and direct fire effects. For Sci. 52, 530-539.

McHugh, C.W., Kolb, T.E., 2003. Ponderosa pine mortality following fire in northern Arizona. Int. J. Wild. Fire 12, 7-22.

Regelbrugge, J.C., Conrad, S.G., 1993. Modeling tree mortality following wildfire in *Pinus ponderosa* forests in the central Sierra Nevada of California. Int J Wildland Fire 3 (3), 139-148.

Saveland, J.M., Neuenschwander, L.F. 1990. A signal detection framework to evaluate models of tree mortality following fire damage. For. Sci. 36, 66-76.

Shepperd, Wayne D., Battaglia, Michael A. 2002. Ecology, silviculture, and management of Black Hills ponderosa pine. Gen. Tech. Rep. RMRS-GTR-97. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 110 p.

Shinneman, D. J.; Baker, W. L. 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in Ponderosa pine landscapes of the Black Hills. Conservation Biology. 11(6): 1276-1288.

Sieg, C.H., McMillin, J.D., Fowler, J.F., Allen, K.K., Negron, J.F., Wadleigh, L.L., Anhold, J.A., Gibson, K.E., 2006. Best predictors for postfire mortality of ponderosa pine trees in the Intermountain West. For. Sci. 52(6), 718-728.

Stephens, S.L., Ruth, L.W., 2005. Federal forest-fire policy in the United States. Ecol. Appl. 15(2): 532-542.

Stephens, S.L., Finney, M.A., 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. For. Ecol. Manage. 162, 261-271.

Thies, W.G., Westling, D.J., Loewen, M., Brenner, G., 2006. Prediction of delayed mortality of fire-damaged ponderosa pine following prescribed fires in eastern Oregon, USA. Int. J. Wildland Fire 15, 19-29.

Veblen, T.T. 2003. Key issues in fire regime research for fuels management and ecological restoration. In: USDA Forest Service Proceedings RMRS-P-29. Pgs. 259-276.

Western Governors Association. 2002. A Collaborative Approach for Reducing Wildland Fire Risks to Communities and the Environment: 10-Year Comprehensive Strategy Implementation Plan.

Wyant, J.G., Omi, P.N., Laven, R.D., 1986. Fire induced tree mortality in a Colorado Ponderosa Pine/Douglas-fir stand. For. Sci. 32(1), 49-59.

CHAPTER 2: PREDICTING MORTALITY OF PONDEROSA PINE REGENERATION AFTER PRESCRIBED FIRE IN THE BLACK HILLS, SOUTH DAKOTA

Abstract

Reduction of crown fire hazard in ponderosa pine (*Pinus ponderosa* var. *scopulorum* Dougl. ex Laws.) forests in the Black Hills of South Dakota often focuses on the removal of overstory trees to reduce canopy bulk density. Within several years after treatment, dense ponderosa pine regeneration establishes and eventually increases crown fire risk. The use of prescribed fire to control ponderosa pine regeneration density is hampered by the lack of knowledge of fire-induced mortality threshold values for seedlings (<1.4 m tall) and saplings (0.1 to 5 cm diameter at breast height (dbh)). This study was initiated to assess fire-related mortality factors of ponderosa pine seedlings and saplings on several prescribed burns across the Black Hills. Plots were established in several burn units after the first postfire growing season to measure crown volume scorch/consumption, basal scorch, and ground char for each seedling/sapling. Logistic regression was used to model the probability of ponderosa pine seedling/sapling mortality and to determine threshold values based on tree size and direct fire effects. Tree size, crown damage, ground char severity, and basal char severity were all important factors in the prediction of mortality. Observed mortality was high for seedlings but was only 30 to 50% for sapling-sized trees. This low level of sapling mortality suggests that managers need to alter their burning windows and burn under drier conditions if sapling mortality is

an objective. Alternatively, managers can add more activity fuels to create a more intense fire in order to use fire as a tool to reduce ponderosa pine sapling densities.

Introduction

Reduction of crown fire hazard in ponderosa pine (*Pinus ponderosa* var. *scopulorum* Dougl. ex Laws.) forests in the Black Hills of South Dakota often focuses on the removal of overstory trees to reduce canopy bulk density. While the initial treatment may reduce crown fire risk in the short-term, the long-term outcome of these treatments is unknown. Prolific ponderosa pine natural regeneration at densities up to several thousand seedlings per hectare is likely to occur in the Black Hills within several years post-treatment (Shepperd and Battaglia 2002, Battaglia 2007), which will subsequently increase the risk of crown fire. Therefore, sustainability of forest stand structures that are more resilient to crown fire will require the control of ponderosa pine seedling regeneration at low densities. Prescribed fire may be used to achieve this goal; however, the severity of direct fire effects needed to induce certain levels of mortality is unknown. While the ability to predict individual tree mortality in large ponderosa pine trees has been well studied (Wyant *et al.* 1986, Saveland and Neuenschwander 1990, Regelbrugge and Conard 1993, Stephens and Finney 2002, McHugh and Kolb 2003, Keyser *et al.* 2006, Sieg *et al.* 2006, Thies *et al.* 2006), fire-induced mortality threshold values for ponderosa pine seedlings (<1.4 m tall) and saplings (0.25 to 10 cm dbh) are limited (Harrington 1987, 1993, van Mantgem and Schwartz 2004). In this paper we explore the utility of developing logistic regression models based on postfire observations to predict first year postfire mortality of individual ponderosa pine seedlings and saplings.

Reducing the density of regeneration to reduce wildfire hazard and promote residual tree vigor is often desired in pine ecosystems where natural regeneration is abundant. Mechanical thinning of regeneration in these stands is often not economically feasible in the Black Hills. However, the application of prescribed fire could potentially be used to achieve this management objective since it has been used to thin dense young stands in several pine ecosystems (Morris and Mowat 1958, Wooldridge and Weaver 1965, Waldrop and Lloyd 1988, Harrington 1987, 1993).

Ponderosa pine seedlings and saplings are thought to be more susceptible to low-intensity prescribed fires because they have thinner bark and lower crowns than their larger diameter counterparts. These two factors increase the probability of multiple fire-induced injuries. It is the cumulative effects of these injuries to the foliage and cambium that contributes to small tree mortality (van Mantgem and Schwartz 2004).

Low crown base heights increase the probability of crown scorch during a fire due to the proximity of the foliage to the flame (van Wagtendonk 1983). Several studies in ponderosa pine trees > 5 cm dbh have demonstrated a strong positive relationship between crown damage by fire and subsequent mortality (Wyant *et al.* 1986, Harrington 1987, 1993, Regelbrugge and Conrad 1993, Stephens and Finney 2002, McHugh and Kolb 2003, van Mantgem and Schwartz 2004, Thies *et al.* 2006, Keyser *et al.* 2006, Sieg *et al.* 2006). Other studies have shown that in general, larger diameter ponderosa pine trees can survive proportionally greater crown damage than smaller trees (Lynch 1959, Wyant *et al.* 1986, Harrington 1987, 1993, Stephens and Finney 2002, McHugh and Kolb 2003, Keyser *et al.* 2006, Sieg *et al.* 2006). The crown damage mortality thresholds in these studies were often dependent on tree diameter or bark thickness and the presence/absence of other fire-related injuries.

As the live crown reaches heights beyond the reach of flames, heat damage from flaming and smoldering combustion to basal cambial tissue also becomes an important factor to consider (Peterson and Arbaugh 1986, 1989, Ryan *et al.* 1988, Ryan and Frandsen 1991, Stephens and Finney 2002). Bark thickness and thermal resistance combined with fire intensity and duration determines the extent of cambium damage (Ryan and Frandsen 1991, van Mantgem and Schwarz 2003). Although small diameter (<15 cm dbh) ponderosa pine trees possess some bark thermal resistance (van Mantgem and Schwarz 2003), fire-related cambial damage in these small trees has been associated with postfire mortality (Lynch 1959, Wyant *et al.* 1986, Stephens and Finney 2002, McHugh and Kolb 2003, van Mantgem and Schwartz 2004, Thies *et al.* 2006, Keyser *et al.* 2006). Cambial damage by itself has not been shown to predict mortality of ponderosa pine trees >5 cm dbh in prescribed burns; however, the inclusion of this variable with crown injury has improved mortality models (Wyant *et al.* 1986, Stephens and Finney 2002, McHugh and Kolb 2003, Thies *et al.* 2006).

Heating of the soil from surface fuel and forest floor consumption can damage the fine root system (Frandsen and Ryan 1986, Dumm 2003, Smith *et al.* 2004, Hart *et al.* 2005) and contribute to tree mortality (Swezy and Agee 1991, Stephens and Finney 2002). In areas where duff is completely consumed by fire, heat pulses into the upper layers of the mineral soil can reach temperatures lethal to roots and soil biota (Hartford and Frandsen 1992). Because root damage is inherently difficult to assess without excavation (Ryan 1983, Swezy and Agee 1991), the amount of charred ground is often used as a surrogate measure (Ryan and Noste 1985). Measures of ground char severity or fuel consumption in conjunction with other damage variables (crown and cambial damage) has been shown to both enhance (Swezy and Agee 1991, Stephens and Finney 2002, Sieg *et al.* 2006) or be insignificant (McHugh and Kolb 2003, Thies *et al.* 2006) in the prediction of mortality in ponderosa pine trees >5 cm dbh.

The successful use of prescribed fire to reduce ponderosa pine seedling and sapling densities requires specific knowledge of fire-induced mortality thresholds and predictive models developed with empirical data from these size classes. Most ponderosa pine mortality models were developed using data collected from trees > 5 cm dbh with datasets that had average diameters exceeding 20 cm dbh. Use of these models to predict seedling (<137 cm tall) and sapling (0.25 to 10 cm dbh) mortality under prescribed burning conditions may produce erroneous predictions. Giving managers the ability to accurately predict regeneration mortality will allow the development of burn prescriptions to meet specific density reduction objectives and greatly enhance the ability to maintain specific forest conditions through time. The specific purpose of this paper was to develop logistic regression models to predict postfire mortality of ponderosa pine seedlings and saplings in the Black Hills. This process further allowed the identification of factors that contribute to fire-induced mortality of these small trees. Providing these tools will greatly enhance the application of prescribed fire in the Black Hills ponderosa pine ecosystem and potentially save critical funds currently devoted to mechanical treatment of fuels and stocking control.

Methods

Logistic regression models were developed to model the probability of seedling or sapling mortality within the first postfire growing season. Logistic regression models are useful for predicting the probability of an occurrence, (*i.e.* live or dead tree) based on predictor variables such as crown damage or tree size. Also, logistic regression analysis does not require normally distributed values. Logistic regression analysis is an accepted technique to model fire-related mortality and has been successfully used in prescribed burn (Ryan and Reinhardt 1988, Ryan and Frandsen 1991, Harrington 1993, Stephens and Finney 2002, McHugh and Kolb 2003, Thies *et al.* 2006) and wildfires (Regelbrugge and Conrad 1993, Keyser *et al.* 2006, Sieg *et al.* 2006).

Study sites

The Black Hills is an isolated forested geologic uplift located in southwest South Dakota and northeast Wyoming that extends 200 km from north to south and 100 km from east to west (Shepperd and Battaglia 2002). Elevations in the Black Hills range from 1,000 m to 2,207 m and are approximately 300 to 1,200 m above the surrounding Great Plains. The increased elevation results in an orographically induced microclimate that increases precipitation. Annual average precipitation ranges from 41 cm in the south to 74 cm in the north. Most of the precipitation occurs from April to August, with May and June receiving 33% of annual precipitation (Driscoll *et al.* 2000). Annual average temperature ranges from 2.9° to 9° C. The frequent rain showers during the early growing season in combination with warm temperatures are conducive to prolific natural ponderosa pine regeneration (Shepperd and Battaglia 2002). Ponderosa pine dominates 85% of the forested land base (DeBlander 2002) and is found at all elevations, soil types, and aspects. It can be found mixed with white spruce (*Picea glauca* [Moench] Voss) and aspen (*Populus tremuloides* Michx.) in the moister, higher elevations.

This study was conducted within the Black Hills on National Forest, Wind Cave National Park, and Mt. Rushmore National Monument public lands. Postfire mortality of ponderosa pine seedlings and saplings were measured on four dormant season prescribed fires and one dormant season wildfire. Sampling was limited to dormant season fires for several reasons. Harrington (1987, 1993) documented higher mortality in growing season fires than in dormant season fires for similar levels of crown scorch. These differences in mortality were attributed to the physiological state of the trees. Therefore, models to predict mortality during prescribed fire conditions should occur when trees are in a dormant physiological state. We also limited our sampling to dormant season fires because that is the time period when prescribed fire is applied in the Black Hills.

Sampling was limited to fires that occurred within a year of the following growing season to ensure fire-related damage measurements were captured before needles abscised or understory vegetation recovery made it difficult to determine ground severity. Because of this time limit and logistical constraints, our sampling was limited to the few prescribed fires that occurred within the Black Hills from Fall 2005 to Spring 2006. The only exception was the Medicine prescribed burn which occurred in the Fall of 2004 and sampled in summer 2005.

Medicine prescribed fire

The Medicine prescribed fire was located on the Mystic Ranger District, Black Hills National Forest about 10 km west southwest of Hill City, South Dakota (approximately 43° 56' 17" N, 103° 42' 28" W) within the central crystalline core

(Shepperd and Battaglia 2002). The Medicine burn unit was 728 ha in size. The vegetation was dominated by ponderosa pine stands with overstories ranging from 15 to 41 cm dbh with substantial densities of seedlings (<137 cm tall). Surface fuel models consisted of a spatial mosaic of fuel model 1, 2, and 9 (Anderson 1982). The Medicine prescribed fire was burned on two separate dates: October 19, 2004 and November 18, 2004. Burns were carried out within the burn prescription and were representative of operational burns (Table 2.1).

Rankin Tower prescribed fire

The Rankin Ridge prescribed burn was located in the northwestern portion of Wind Cave National Park (approximately 43° 37' 50" N, 103° 28' 56" W) within the red valley geomorphological feature (Shepperd and Battaglia 2002). The burn unit was 500 hectares and consisted of ponderosa pine forest and mixed-grass prairie. Surface fuel models consisted of a spatial mosaic of fuel model 1 and 2 (Anderson 1982). The burn occurred on October 25, 2005 and was carried out within the burn prescription representative of operational burns (Table 2.1).

Horse Nugget prescribed fire

The Horse Nugget prescribed fire was located on the Mystic Ranger District, Black Hills National Forest about 9 to 13 km north of Hill City, South Dakota (approximately 44° 01' 16" to 44° 03' 11" N, 103° 34' 16" to 103° 34' 40" W) within the central crystalline core geomorphological feature (Shepperd and Battaglia 2002). The burn unit was approximately 600 ha in size. The vegetation was dominated by ponderosa pine stands with overstories ranging from 15 to 41 cm dbh with substantial densities of seedlings (<137 cm tall) and saplings (0.25 to 10 cm dbh). Surface fuel models consisted of a spatial mosaic of fuel models 2, 9, and 11 (Anderson 1982). The Horse Nugget prescribed fire was ignited on three separate dates: October 28, 2005, November 1, 2005, and November 2, 2005. Burns were carried out within the burn prescription and were representative of operational burns (Table 2.1).

Bullock prescribed fire

The Bullock prescribed fire was located on the Mystic Ranger District, Black Hills National Forest about 10 km northeast of Hill City, South Dakota approximately 44° 00' 47" N, 103° 32' 26" W) within the central crystalline core geomorphological feature (Shepperd and Battaglia 2002). The Bullock burn unit was 1,000 ha in size. The area was a mix of ponderosa pine stands with overstories ranging from 15 to 41 cm dbh with limited regeneration and large open areas with ponderosa pine seedling/sapling encroachment. Surface fuel models consisted of a spatial mosaic of fuel model 1 and 9 (Anderson 1982). The Bullock prescribed fire was ignited on January 24, 2006 and carried out within the burn prescription and was representative of operational burns (Table 2.1).

Lafferty Gulch wildfire

The Lafferty Gulch (Mount Rushmore 1) wildfire was located on 39 ha in the northeastern portion of Mount Rushmore National Monument (approximately 43° 53' 20" N, 103° 26' 19" W) within the central crystalline core geomorphological feature (Shepperd and Battaglia 2002). The wildfire was ignited as a result of a pile burn operation on February 26, 2006 and burned until March 3, 2006. Although it was classified as a wildfire, weather conditions during the wildfire were similar to those of other prescribed burns (Table 2.1) and bole char heights up to 7.5 m were also similar.

The overstory was dominated by ponderosa pine with the majority of diameters from 15 to 41 cm dbh. The areas also had some large (30 to 60.5 cm dbh) diameter ponderosa pines. The majority of the area burned had been mechanically treated in 2003 to remove ponderosa pine saplings < 15 cm dbh and resulted in a residual density of 91 saplings ha^{-1}. Surface fuels in the area consisted of a spatial mosaic of fuel model 2 and 9 (Anderson 1982).

Plot setup

Although sample design and layout differed between the sites, plot sampling procedures were identical. The design and layout differed because the Medicine burn, which occurred in the Fall 2004, was part of a separate study to investigate the relationship between fire behavior and seedling mortality. Since fire behavior was to be measured, random plots were established before the prescribed burn. Two other sites were included in that study but due to logistical problems, they were not burned. The other four sites measured in the current study were measured with transects with a random starting point.

Medicine prescribed fire

Using aerial photographs taken in 2002 of the Medicine burn unit, potential study sites were identified based on overstory coverage and presence of ponderosa pine regeneration. Each potential site was visited for ground truthing and delineated with GPS, ensuring that ponderosa pine regeneration was present and the site was homogeneous based on overstory coverage, regeneration, and fuel model. Areas less than 0.5 hectare were not considered. Five study sites, ranging from 0.5 to 2 hectares in size were established within the 728 ha burn unit. Elevations of these study sites ranged from 1900 to 1975 m. Five randomly located plots were established within each of the study sites. Fire behavior was monitored during the October 19, 2004 prescribed fire using a video camera and flame heights were estimated for 15 of the 25 plots. At each plot, a post with 0.15 m intervals was used as a reference point to measure flame height. Flame height (H, m) was converted to flame length (L, m) using the following equation:

$$L = \{H [\sin (90 - \beta)]\}/[\sin (\theta - \beta) \quad [eq. 2]$$

where, β is the measured slope angle and θ is the angle of the flame from the horizontal. A mean flame angle of 50° was used for the conversion in eq. 2 following Kobziar *et al.* (2006).

Rankin Tower, Bullock, and Horse Nugget prescribed fires

Transects 200 m long were established on each burn unit in the 2006 growing season. The starting point of each transect was randomly generated within the Bullock (n=1) and Horse Nugget (n=5) prescribed burn perimeters. Elevations for the Bullock transect was approximately 1600 m and ranged between 1630 to 1770 m for the Horse Nugget transects. In the Rankin Tower prescribed burn, the starting point of two transects were placed on permanent monitoring plots that were already established by Wind Cave National Park staff, while the other three transects were randomly established. Elevations for the Rankin Tower transects ranged from 1400 to 1480 m. The direction of each transect was randomly chosen and a plot (n=5 per transect) was established every 50 m along each transect.

Lafferty Gulch wildfire

Before the wildfire, National Park Service staff had set up randomly selected plots within the Lafferty Gulch mechanical fuel reduction project for monitoring purposes.

Several of these monitoring plots burned in the wildfire. The starting point of four transects were placed on an original fuel reduction project monitoring plot. The direction of each transect was randomly chosen and a plot (n=5 per transect) was established every 50 m along each transect. An additional seven fire mortality plots were established at the location of other original fuel reduction project monitoring plots within the burn perimeter. Elevations for transects and monitoring plots ranged between 1400 and 1450 m.

Data collection

A nested plot design consisting of a 2-m radius circular plot to sample ponderosa pine seedlings (trees<137 cm tall) located within a larger circular plot to sample saplings (tree 0.25 to 10 cm dbh) was used on all sites. The larger sapling plot was 5-m in radius on the Medicine site and 15-m radius on the Rankin Tower, Bullock, Horse Nugget, and Lafferty Gulch study sites. We increased the radius size for the Rankin Tower, Bullock, Horse Nugget, and Lafferty Gulch because it was found that a 5 m radius on the Medicine burn was not large enough to measure enough saplings. Ponderosa pine seedlings were categorized into 5 height size classes (0.1 to 15.2 cm; 15.2 to 45.7 cm; 45.7 to 76.2 cm; 76.2 to 106.7 cm; and 106.7 to 137 cm). Seedling density, mortality, and fire-induced mortality were measured on the 2-m radius plot for each study site. Ponderosa pine saplings (0.25 to 10 cm dbh) were categorized into 4 diameter size classes (0.25 to 2.54 cm; 2.54 to 5.1 cm; 5.1 to 7.6 cm; and 7.6 to 10.2 cm). Sapling density, mortality, and fire-induced damage were assessed on each of the larger sapling plots.

Fire damage variables were measured for each seedling and sapling on a plot. Percent of pre-fire live crown volume scorched (CVS) and percent of live crown volume consumed (CVC) were visually assessed to the nearest 5% by viewing the tree from all sides (Peterson 1985). Crown scorch was defined as foliage that experienced color change as a result of fire (Keyser *et al.* 2006), but which was not consumed (Ryan 1982). This included singed foliage as defined by McHugh and Kolb (2003). Crown consumption was defined as foliage consumed by active combustion and was determined by the presence of needle fascicles on small branches to indicate that branches had supported live foliage before the fire (McHugh and Kolb 2003, Sieg *et al.* 2006). Total crown volume damaged (CVD) was calculated by adding CVS+CVC (McHugh and Kolb 2003, Sieg *et al.* 2006).

Basal char severity was used to estimate cambial damage following criteria established by Ryan (1983) and utilized by several recent studies (McHugh and Kolb 2003, Sieg *et al.* 2006, Thies *et al.* 2006, Hood and Bentz *in press*). The percentage of the basal circumference on the first 5 cm above the ground was assessed with the following criteria: 0 = none, no evidence of flame having contacted the bole and no charring or darkening of the bole; 1 = light, light scorch or char on edges of bark plates; 2 = moderate, bark is uniformly black with the possible exception of the inner depths of the prominent fissures, but bark character is still discernable; 3 = heavy, bark is deeply charred, but not necessarily to the wood and surface characteristics have been lost. We assumed that heat sufficient to cause moderate or heavy basal char severity would kill the cambium (Thies *et al.* 2006). A basal char severity class was calculated based on the summation of the percentage of the bole that had moderate or heavy basal char severity.

Basal char severity classes were: 0 if the summation of moderate and heavy basal char = 0; 1 if the summation of moderate and heavy basal char was >0 and <25%; 2 if the summation of moderate and heavy basal char was >25 and <50%; 3 if the summation of moderate and heavy basal char was >50 and <75%; and 4 if the summation of moderate and heavy basal char was >75%.

Ground surface char severity was used to indirectly estimate damage to root systems (Swezy and Agee 1991, McHugh and Kolb 2003, Sieg *et al.* 2006, Thies *et al.* 2006, Hood and Bentz *in press*). Ground char was measured under the dripline of the crown of each seedling and sapling using the criteria developed by Ryan (1983): 0 = none, no visible effect on soil; 1 = light, surface of litter and duff layers scorched or charred; 2 = medium, litter completely consumed and duff deeply charred or consumed, but the underlying mineral soil not visibly altered; 3 = high, litter and duff completely consumed.

Data analysis

Wilcoxon rank sum tests were used to test for differences (α=0.05) between live and dead trees morphological and fire-damage characteristics across all fires using PROC NPAR1WAY in SAS (SAS Institute 2001).

Logistic regression models were developed to model the probability of seedling or sapling mortality within the first postfire growing season. The logistic regression equation used to model tree mortality has the form:

$$P_m = 1/[1+\exp(-(\beta_0+\beta_1 X_1+\ldots+\beta_n X_n))], \text{ (1)}$$

Where P_m is the probability of tree mortality, β_0, β_1, and β_n, are regression coefficients, and X_1 and X_n are representative independent variables. The logistic function provides a continuous estimate of the probability of mortality between 0 and 1, with 1 indicating a dead tree. We used a cutoff of 0.5 to signal mortality (Saveland and Neuenschwander 1990, Keyser *et al.* 2006). If $P_m < 0.5$ then the tree was predicted live, otherwise it was predicted dead.

We used the tree morphological and fire-damage variables to predict tree mortality using PROC LOGISTIC in SAS (SAS Institute 2001). Only variables that were significantly different between live and dead trees ($P<0.05$) and were not strongly correlated ($r \leq 0.50$) with other independent variables were used in the development of logistic regression models (McHugh and Kolb 2003, Thies *et al.* 2006). Separate models were developed for seedlings and saplings because the tree morphological measurement that differed by convention defines the two size classes (*i.e.* seedlings=height and saplings=dbh). Models were built using a random sample of ~75% of the full data set [calibration data set (n=1280 for seedling models and n=1768 for saplings models)]. Predictability of significant models was assessed using the remaining ~25% of the full data set to validate the models [validation data set (n=336 for seedling models and n=573 for saplings models)] (Regelbrugge and Conard 1993, Keyser *et al.* 2006).

Exploratory models were developed using score and stepwise selection options (Hosmer and Lemeshow 2000). All independent variables and any biologically relevant interactions were included in the full model. The generalized Wald statistic with a χ^2 distribution was used to test if model coefficients were different from zero ($\alpha=0.05$). Only variables and interactions that were found to have a P-value < 0.05 were kept in the model. The Hosmer-Lemeshow goodness of fit test was used to evaluate the fit of the final models (Hosmer and Lemeshow 2000). If a model is good, the Hosmer-Lemeshow test statistic will have a p-value>0.05 indicating that the model prediction does not significantly differ from the observed data. We also used the receiver operations characteristic (ROC) curve analysis (Saveland and Neuenschwander 1990, Hosmer and Lemeshow 2000) to evaluate the model's ability to discriminate between dead and live trees. Models with ROC values between 0.70 and 0.80 have acceptable discrimination, ROC values between 0.80 and 0.90 have excellent discrimination, and ROC values $\geq$ 0.90 have outstanding discrimination (Hosmer and Lemeshow 2000). For each model, we used the validation data set to assess the model's prediction to the observed status of the tree and calculated the percentage of correctly classified trees.

Results

Seedlings

Of the 1310 seedlings surveyed, 86% were classified dead after the first growing season following fire. The percentage of seedlings that died decreased with seedling height (Figure 2.1a). Mortality of seedlings less than 46 cm tall was above 90% (Figure 2.1a). Mortality of seedlings that were 107 to 137 cm tall was 56% (Figure 2.1a).

Dead seedlings were significantly shorter than the surviving seedlings (Table 2.2). Dead and live seedlings had similar crown volume scorch values, but dead seedlings had significantly higher crown volume consumption and total crown damage than live seedlings (Table 2.2). Cambial injury, as measured by basal char severity, was significantly higher for dead seedlings (Table 2.2). Damage to roots, as measured by ground char severity, was significantly higher around dead seedlings (Table 2.2).

Observed mortality was around 10% for seedlings at lower levels of crown damage, but a threshold was reached around 40% crown damage and mortality increased linearly (Figure 2.2a). Seedlings experiencing no ground char severity still had 60% mortality, but when ground char was present, observed mortality was >90% (Figure 2.2b). Observed seedling mortality was >80% with no severe basal char and >95% with high levels of severe basal char (Figure 2.2c).

***Logistic models for seedlings*:**

Multivariate models that included seedling height, multiple fire damage variables, and flame height predicted mortality with the highest ROC values and accuracy (Table 2.3). In contrast to the sapling models, basal char severity and ground char severity were not highly correlated ($r=0.42$; $p<0.0001$), so both were included together during model development. Because there was no significant difference in crown volume scorch between live and dead seedlings (Table 2.2) we limited our model development to use only total crown volume damage (CVD) as our crown damage variable. Many one or two variable models that managers could use more readily in the field were attempted, but the majority of them failed to pass the Hosmer-Lemeshow goodness of fit test. For each of the significant models, coefficients for seedling height were negative indicating that shorter seedlings have a significant increase in the predicted probability of mortality. Coefficients for total crown damage (CVD), basal char (BC) severity, and ground char (GC) severity were all positive suggesting that increases in direct fire effects results in an increase in predicted probability of mortality.

The full model (four variables) using seedling height, CVD, GC severity, and BC severity explained the most variation (ROC=0.97) and had the highest accuracy (93%) of all the models developed (Table 2.3). This model suggests that seedling size, crown damage, impacts to surface roots, and cambial injury are all important factors in predicting mortality. To illustrate the relationship of these factors, two levels of BC severity: 0 to 25% and >75% were chosen to consider further. In general, as seedlings decreased in height, less crown damage, lower BC severity, and lower GC severity were associated with mortality (Figure 2.3). Levels of GC and BC severity influenced the amount of crown damage that was in turn associated with mortality (Figure 2.3). Predicted mortality was more sensitive to GC severity than BC severity. For example, at similar levels of BC severity, the amount of CVD associated with dead trees decreased substantially with an increase in GC severity (Figure 2.3). At similar levels of GC severity, the amount of CVD required to predict mortality also decreases with an increase in BC severity, but at a slower rate (Figure 2.3).

Using the fire behavior data from the 15 of the 25 plots on the Medicine prescribed burn, we developed a model that predicted seedling mortality based on observed flame height. This model had a ROC value of 0.90 and an accuracy of 92% (Table 2.3). The predicted probability of seedling mortality increased with flame height (Figure 2.4). Dead seedlings less than 60 cm tall were associated with flame heights of 5 cm (Figure 2.4). Seedlings that were 90 cm tall required flame heights > 25 cm and seedlings that were 120 cm tall required flame heights > 45 cm to predict mortality (Figure 2.4).

Many one variable models were tested to predict seedling mortality. These often had acceptable ROC values, but failed the Hosmer-Lemeshow test for lack-of-fit. Only one model to predict seedling mortality was significant, had acceptable ROC values, and passed the lack-of-fit test. The GC severity model had an ROC of 0.72 and 78% accuracy (Table 2.3). This model predicted that the probability of seedling mortality increased as GC increased. However, this model predicted some seedling mortality when no ground char occurred, suggesting that other factors play a role in seedling mortality.

Saplings

Of the 2,341 saplings surveyed, 48% were classified as dead in the first growing season following fire. The percentage of saplings that died ranged from 32% to 59%, with lower mortality occurring at larger diameters (Figure 2.1b).

Dead saplings were significantly smaller in diameter than the surviving saplings (Table 2.2). Dead saplings also had significantly higher crown scorch, crown consumption, and total crown damage than live saplings (Table 2.2). Cambial injury, as measured by basal char severity, was significantly higher for dead saplings (Table 2.2). Damage to roots, as measured by ground char severity, was significantly higher around dead saplings (Table 2.2).

Observed sapling mortality was low up to about 65% crown damage and then mortality increased sharply (Figure 2.2a). Sapling mortality increased also with increased ground char severity (Figure 2.2b) and with basal char severity. However, observed sapling mortality was <40% until basal char severity >75% of the basal circumference and then mortality increased to 80% (Figure 2.2c).

Logistic models for saplings:

Multivariate models that included sapling dbh, multiple fire damage variables, and sapling density predicted mortality with the highest ROC values and accuracy (Table 2.4). Since BC severity and GC severity were correlated ($r=0.66$; $p<0.0001$) separate regression models were created for each variable. As with seedlings, one or two variable models that managers could use more readily in the field were attempted, but again, the majority of them also failed to pass the Hosmer-Lemeshow goodness of fit test. For each of the significant models, coefficients for dbh were negative indicating there is a significant increase in the predicted probability of mortality of saplings of smaller diameters. Coefficients for crown damage (CVD or CVC), basal char (BC) severity, and ground char (GC) severity were all positive suggesting that increases in these direct fire effects results in a greater likelihood of mortality. Sapling density was also shown to be an important factor in the full model. Higher sapling density resulted in an increase in predicted probability of mortality.

The full model (four variables) using dbh, CVD, GC severity, and sapling density explained the most variation (ROC=0.97) and had the highest accuracy (92%) of all the models developed (Table 2.4). This model suggests that the effect of fire on ponderosa pine saplings is strongly linked to sapling size, crown damage, impacts to surface roots, and sapling density. The amount of CVD required to predict sapling mortality decreased as GC severity increased (Figure 2.5). Thresholds of CVD differed for each GC severity and sapling dbh (Figure 2.5). Sapling density also impacted these thresholds, but differences in the amount of CVD within a GC severity were within 5% (data not shown). Median sapling density (1160 TPH) across the study sites was chosen to illustrate the differences in damage required to predict mortality. For example, an area that experienced light GC required 85% CVD to predict a dead 2.5 cm dbh sapling and 95% CVD to predict a dead 5.0 cm dbh sapling, but saplings 7.5 and 10 cm dbh would still be predicted to survive with 100% CVD (Figure 2.5a and 2.5b). Predicted mortality for the 7.5 cm and 10 cm dbh saplings required at least moderate levels of GC combined with 85 to 95% CVD, respectively (Figure 2.5c and 2.5d). Mortality in saplings 2.5 to 5 cm dbh experiencing moderate GC required 75 to 80% CVD (Figure 2.5a and 2.5b), respectively.

The three-variable model using dbh, CVD, and BC severity class had a high ROC value (0.96) and an accuracy of 91% (Table 2.4), suggesting that cambial damage is an important predictor of mortality. In general, the amount of CVD required to predict mortality decreased as BC severity increased (Figure 2.6). Thresholds of CVD differed for the BC severity class and sapling dbh (Figure 2.6). Mortality of smaller dbh saplings required lower values of crown damage and cambial damage than larger dbh saplings. However, it still required at least 95% CVD with no BC severity and 70% CVD in conjunction with >75% BC to predict mortality in the 2.5 cm dbh saplings (Figure 2.6a). For saplings 10 cm dbh, a minimum of 90% CVD in conjunction with BC of 50 to 75% was required to predict sapling mortality (Figure 2.6d).

The multivariate model that included sapling dbh and CVC had a ROC of 0.71 with an accuracy of 68% (Table 2.4). While this model was moderately successful in correctly predicting mortality, the ease of measuring these two variables in the field warrants their attention. The estimated probability of sapling mortality increases with an increase in CVC, but mortality thresholds at CVC<10% differ somewhat with dbh. However, if 40% of the crown is consumed, mortality is predicted for all sapling sizes.

The only significant univariate model developed was the GC severity model. The GC severity model had a ROC of 0.76 and an accuracy of 72% (Table 2.4). This model showed that saplings experiencing moderate to high GC had increased predicted probability of sapling mortality.

Discussion

A survey of several prescribed fires and one dormant season wildfire in the Black Hills indicates that dormant season fire can be used to control ponderosa pine regeneration density. Fire-induced mortality of ponderosa pine seedlings (<137 cm tall) and saplings (0.25 to 10 cm dbh) requires different levels of damage depending on individual size. Logistic regression analysis indicated that tree size (height or dbh), crown damage, ground char severity, basal char severity, and density were all important factors in predicting mortality. Model accuracy was highest with the inclusion of at least 3 of these variables suggesting that mortality is a result of multiple injuries. A model relating observed flame height to seedling mortality also indicated that shorter seedlings were more susceptible to taller flames. Understanding the mechanisms that promote fire-induced mortality in ponderosa pine regeneration will allow managers to effectively plan prescribed fires and assess their success in meeting burn plan objectives.

The significance of ponderosa pine tree size in predicting the probability of fire-induced mortality in seedlings and saplings in our study was not surprising since most other studies have reported a decrease in mortality with an increase in large tree diameter (Wyant *et al.* 1986, Harrington 1987, Saveland *et al.* 1990, Harrington 1993, Regelbrugge and Conard 1993, Stephens and Finney 2002, McHugh and Kolb 2003, Keyser *et al.* 2006, Sieg *et al.* 2006). Tree size influences the susceptibility of foliage, cambium, and roots to fire damage. In terms of prescribed fires, where flame heights are often designed to be 0.25 to 1 meter tall, foliage of a seedling is highly susceptible. Saplings, with live crowns within the maximum range of prescribed flame heights, are at less risk for crown consumption, but still have foliage highly susceptible to crown scorch.

Smaller trees have thinner bark which influences cambial heat resistance and increases susceptibility to girdling (van Mantgem and Schwartz 2003). A significant amount of fine root biomass is located in the upper 10 cm of the mineral soil (van Haverbeke 1963, Dumm 2003, Smith *et al.* 2004, Hart *et al.* 2005) and increases the susceptibility of seedlings and saplings to root damage and subsequent mortality.

Under prescribed burning conditions, our survey observed that mortality exceeded 90% for ponderosa pine seedlings <46 cm tall, but as seedlings reached 137 cm tall, mortality decreased to 56% (Figure 2.1a). Sapling mortality also decreased as diameter increased, with only 32% of 7.5 to 10 cm dbh saplings succumbing to fire (Figure 2.1b). The higher mortality in the seedlings was in part due to convective heat and direct flame contact with the foliage. The logistic regression model developed from our observations of flame lengths on the Medicine prescribed burn predicted that mortality of a seedling less than 60 cm tall was associated with flames less than 5 cm (Figure 2.4). On average, observed flame lengths were at least 15 to 30 cm tall. This would have resulted in torching fire behavior within the seedling regeneration layer, high levels of crown consumption, and subsequently higher mortality rates. The taller seedlings (107 to 137 cm), which had mortality rates of 56%, required flame lengths of about 45 cm to predict mortality (Figure 2.4). These flames lengths were observed at times during the fire, but likely mortality in these taller seedlings was a combination of crown damage and other fire-related damage (Table 2.3).

Unfortunately, we were unable to monitor flame lengths on the prescribed burns where saplings were sampled for this study, but we hypothesize that similar to the taller seedlings, taller flames would be required for crown consumption. We did observe some crown consumption in the sapling sized trees in our survey. Results from the reduced model that used only crown volume consumption and dbh to predict mortality in saplings suggested that small percentages of crown consumption resulted in higher mortality. Furthermore, the mortality threshold of crown consumption increased slightly with dbh. Accuracy of that model was only 68%, which suggests that while severe crown damage is important in saplings, the consideration of other fire-related damage, especially when crowns are only scorched, is warranted.

As the live crown reaches heights beyond the reach of direct flames, heat damage from flaming and smoldering combustion to cambial tissue also became an important factor in this study. Fire related cambial damage in small trees has been associated with postfire mortality in many studies (Lynch 1959, Wyant *et al.* 1986, Stephens and Finney 2002, McHugh and Kolb 2003, van Mantgem and Schwartz 2004, Thies *et al.* 2006, Keyser *et al.* 2006). We indirectly measured cambial damage by examining the degree of basal char on the lower 5 cm of the seedling and sapling bole and calculating the percentage of the circumference that experienced moderate to high severity char (BC severity). While indirect measures of cambial damage by itself has not been shown to predict mortality of ponderosa pine trees >5 cm dbh in prescribed burns; the inclusion of this variable with crown injury was shown to improve model predictions in other studies (Wyant *et al.* 1986, Stephens and Finney 2002, McHugh and Kolb 2003, Thies *et al.* 2006) as well as this study (Table 2.3 and 2.4). For saplings, the amount of crown volume damage required to predict mortality decreased as BC severity increased and thresholds decreased with sapling dbh (Figure 2.6). Predicting mortality of smaller saplings required lower values of crown damage and cambial damage than larger saplings. For example, predicting mortality in a 2.5 cm dbh sapling, required at least 95% CVD with no BC severity compared to 70% CVD with >75% BC (Figure 2.6a). For saplings 10 cm dbh, a minimum of 90% CVD in conjunction with BC of 50 to 75% was required for predicting sapling mortality (Figure 2.6d). These results support the hypothesis put forth by van Mantgem and Schwartz (2004) that it is the additive effect of damage to different tree organs that results in tree death.

Our observation of increased seedling and sapling mortality with an increase in ground char severity also supports the multiple-injury hypothesis. Ground char severity is a surrogate measure of duff consumption and subsequent fine root mortality (Ryan and Noste 1985, Swezy and Agee 1991). Seedlings in our study with no ground char had mortality levels of about 60% (Figure 2.2b), most likely due to crown damage. However, when there was ground char present, observed seedling mortality increased to above 90%, regardless if ground char was light or high (Figure 2.2b). Sapling mortality also increased with an increase in ground char severity, but the severity of ground char determined the amount of mortality (Figure 2.2b).

Differences in seedling and sapling response to ground char severity may be a function of biomass allocation to roots. Ponderosa pine seedlings allocate over 50% of their biomass to roots whereas saplings only allocate about 11% (Grunkle and Retzlaff 2001). Most of the fine root biomass is located in the upper 10 cm of the mineral soil (van Haverbeke 1963, Dumm 2003, Smith *et al.* 2004, Hart *et al.* 2005) and it is at this depth that fine root and associated mycorrhizae biomass significantly decrease after low intensity prescribed fires (Swezy and Agee 1991, Dumm 2003, Smith *et al.* 2004, Hart *et al.* 2005). The combination of crown damage and some root mortality was enough to presumably impact the seedling's water/nutrient uptake ability. However, it took higher amounts of ground char to impact the saplings (Figure 2.2b).

Measures of ground char severity or fuel consumption in conjunction with other damage variables (crown and cambial damage) has been shown to enhance prediction of mortality in ponderosa pine trees > 5 cm dbh in some studies (Swezy and Agee 1991, Stephens and Finney 2002, Sieg *et al.* 2006) but not in others (McHugh and Kolb 2003, Thies *et al.* 2006). In our study, in both the seedling and sapling mortality models, ground char severity was shown to be an important factor by itself or in conjunction with other damage variables in predicting mortality (Table 2.3 and 2.4). Mortality model with ground char severity alone had acceptable discrimination and accuracy of 72-78% (Table 2.3 and 2.4). However, combining ground char severity with other fire damage variables and tree size increased the accuracy to over 90% (Table 2.3 and 2.4).

When ground char severity was combined with the other variables to predict seedling and sapling mortality, we found that ground char and basal char severity were highly correlated in the sapling observations, but not for the seedlings. This discrepancy in correlation between seedlings and saplings is likely a result of the fuel bed. Seedlings were often found in open gaps within the forest matrix or under low density overstories. Low density overstories typically have an herbaceous fuel bed (Shepperd and Battaglia 2002). In contrast, saplings were found under a gradient of overstory conditions with fuel beds that ranged from herbaceous fuels to needle litter and woody surface fuels. In areas with an herbaceous fuel bed, fires would move quickly with lower impacts on the surface, while still scorching the bole. In contrast, in areas with a needle and woody fuel bed, fires would move slower and have an impact on the ground and the bole simultaneously.

Because of the high correlation in the saplings, basal char severity was not included in any of the ground char severity sapling models. Without basal char in the sapling model, the stepwise regression procedure determined that sapling density was a significant variable along with dbh, CVD, and ground char severity. Higher predicted probability of mortality with an increase in sapling density is likely due to a combination of lower tree vigor (Swezy and Agee 1991, van Mantgem *et al.* 2003) and higher needle litter fuel loadings. The higher needle fuel loads would increase the duration of fire and subsequently kill more roots and foliage.

Ground char and basal char severity were not strongly correlated in the seedling measurements so they were modeled together. In general, less damage was required to predict mortality as seedlings decreased in height (Figure 2.3 and 2.4). Levels of GC and BC severity also influenced the amount of crown damage required to predict mortality (Figure 2.3 and 2.4). These changes in thresholds highlight the susceptibility of seedlings to different damage pathways and give managers several options when designing burn prescriptions.

Conclusions

The sustainability of fuel reduction treatments in the Black Hills will require the control of ponderosa pine regeneration densities. Because mechanical thinning of small trees is not economical in the Black Hills, there is interest in using prescribed fire to maintain low regeneration densities. Traditionally, prescribed burns are written with the premise of limiting mortality. However, the use of prescribed burn to maintain low densities will require managers to plan burns that encourage mortality of small trees while limiting mortality of larger trees. Fire managers are looking for ways to achieve this goal.

The models developed in this study will aid managers in the planning stages of their prescribed burns. Our model that relates flame length and seedling height to mortality provides managers with benchmark flame lengths needed to predict mortality in seedlings (<137 cm tall). Future research is planned to identify fire behavior thresholds required to induce mortality in small diameter saplings. Observed mortality in the saplings 2.5 to 10 cm dbh in this study was 30-50%. This low level of mortality suggests that to kill saplings, managers need to alter their burning windows and burn under drier conditions. Alternatively, managers can add more activity fuels to create a more intense fire in order to use fire as a tool to reduce ponderosa pine sapling densities.

Literature cited

Anderson, H.E., 1982. Aids to determining fuel models for estimating fire behavior. USDA Forest Service, General Technical Report, INT-122.

Battaglia, M.A. 2007. Maintaining fuel treatments with prescribed fire in ponderosa pine (*Pinus ponderosa*) forests of the Black Hills, South Dakota. Ph.D. Dissertation. Colorado State University, Fort Collins, Colorado.

Dumm, G. 2003. Fire effects on fine roots and ectomycorrhizae of ponderosa pine and Douglas-fir following a prescribed burn in a central Idaho forest. M.S. Thesis. Univ of Idaho, Moscow, Idaho.

Frandsen, W.H., Ryan, K.C., 1986. Soil moisture reduces belowground heat flux and soil temperatures under a burning fuel pile. Can. J. For. Res. 16, 244-248.

Grunkle, N.E., Retzlaff, W.A. 2001. Changes in physiological attributes of ponderosa pine from seedling to mature tree. Tree Phys. 21, 275-286.

Harrington, M. G., 1993. Predicting Pinus ponderosa mortality from dormant season and growing season injury. Int. J. Wildland Fire 3 (2), 65-72.

Harrington, M.G., 1987. Ponderosa pine mortality from spring, summer, and fall crown scorching. WJAF 2 (1), 14-16.

Hart, S.C., Classen, A.T., Wright, R.J., 2005. Long-term interval burning alters fine root and mycorrhizal dynamics in a ponderosa pine forest. J. Applied Ecology. 42, 752-761.

Hartford, R.A., Frandsen, W.H., 1992. When it's hot,…., its hot or maybe it's not! (surface flaming may not portend extensive soil heating). Int. J. Wildland Fire 2 (3), 139-144.

Hood S, Bentz B., (*in press*). Predicting post-fire Douglas-fir beetle attacks and tree mortality in the Northern Rocky Mountains. *Can. J. For. Res.*

Hosmer D., Lemeshow, S,, 2000. Applied logistic regression. John Wiley and Sons, New York.

Keyser T.L., Smith, F.W., Lentile L.B., Shepperd, W.D., 2006. Modeling postfire mortality of ponderosa pine following a mixed-severity wildfire in the Black Hills: The role of tree morphology and direct fire effects. For Sci. 52, 530-539.

Kobziar, L., Moghaddas, J., Stephens, S.L, 2006. Tree mortality patterns following prescribed fires in a mixed conifer forest. Can. J. For. Res. 36, 3222-3238.

Lynch, D.W., 1959. Effects of wildfire on mortality and growth of young ponderosa pine trees. USDA Forest Service Research Note, INT-66.

Morris, W.G., Mowat, E.L., 1958, Some effects of thinning a ponderosa pine thicket with a prescribed fire. J. For. 56, 203-209.

McHugh, C.W., Kolb, T.E., 2003. Ponderosa pine mortality following fire in northern Arizona. Int. J. Wild. Fire 12, 7-22.

Peterson, D.L., 1985. Crown scorch volume and scorch height: estimates of post-fire tree condition. Can. J. For. Res. 15, 596-598.

Peterson, D.L., Arbaugh, M.J., 1986. Postfire survival of Douglas-fir and lodgepole pine: Comparing the effects of crown and bole damage. Can. J. For. Res. 16(6), 1175-1179.

Peterson, D.L., Arbaugh, M.J., 1989. Estimating post-fire survival of Douglas-fir in the Cascade Range. Can. J. For. Res. 19, 530-533.

Regelbrugge, J.C., Conrad, S.G., 1993. Modeling tree mortality following wildfire in *Pinus ponderosa* forests in the central Sierra Nevada of California. Int J Wildland Fire 3 (3), 139-148.

Ryan, K.C., 1982. Evaluating potential tree mortality from prescribed burning. In: Baumgartner, D.M. (Ed.), Site preparation and fuels management on Steppe terrain. Washington State University Cooperative Extension, Pullman, WA, pp. 167-179.

Ryan K.C., 1983. Techniques for assessing fire damage to trees. In: Lotan, J.E. (Ed.), Proceedings of the symposium: fire its field effects. Intermountain Fire Council: Missoula, MT, pp. 2-10.

Ryan, K.C., Noste, N.V., 1985. Evaluating prescribed fires. In: Lotan, J.E., Kilgore, B.M., Fischer, W.C., Mutch, R.W., (Tech. Coords.), Proceedings of the symposium and workshop on wilderness fire. USDA Forest Service General Technical Report, INT-82.

Ryan, K.C., Reinhardt, E.D., 1988. Predicting post-fire mortality of seven western conifers. Can. J. For. Res. 18, 1291-1297.

Ryan, K.C., Peterson, D.L., Reinhardt, E.D., 1988. Modeling long-term fire caused mortality of Douglas-fir. For. Sci. 34 (1), 190-199.

Ryan, K.C., Frandsen, W.D., 1991. Basal injury from smoldering fires in mature *Pinus ponderosa* Laws. Int J Wildland Fire 1(2), 107-118.

SAS Institute, 2001. SAS/STAT software, version 9.1 of the SAS system for Windows 2000. SAS Institute Inc., Cary, NC.

Saveland, J.M., Neuenschwander, L.F. 1990. A signal detection framework to evaluate models of tree mortality following fire damage. For. Sci. 36, 66-76.

Shepperd, W.D., Battaglia, M.A., 2002. Ecology, silviculture, and management of Black Hills ponderosa pine. USDA Forest Service General Technical Report, RMRS-GTR-97.

Sieg, C.H., McMillin, J.D., Fowler, J.F., Allen, K.K., Negron, J.F., Wadleigh, L.L., Anhold, J.A., Gibson, K.E., 2006. Best predictors for postfire mortality of ponderosa pine trees in the Intermountain West. For. Sci. 52(6), 718-728.

Smith, J.E., McKay, D., Niwa, C.G., Thies, W.G., Brenner, G., Spatafora, J.W., 2004. Short-term effects of seasonal prescribed burning on the ectomycorrhizal fungal community and fine root biomass in ponderosa pine stands in the Blue Mountains of Oregon. Can. J. For. Res. 34, 2477-2491.

Stephens, S.L., Finney, M.A., 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. For. Ecol. Manage. 162, 261-271.

Swezy, M.D., Agee, J.K., 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. Can. J. For. Res. 21, 626-634.

Thies, W.G., Westling, D.J., Loewen, M., Brenner, G., 2006. Prediction of delayed mortality of fire-damaged ponderosa pine following prescribed fires in eastern Oregon, USA. Int. J. Wildland Fire 15, 19-29.

van Haverbeke, D.F., 1963. Root development of ponderosa pine seedlings in the Black Hills. Ecology. 44, 161-165.

van Mantgem, P., Schwarz, M., 2003. Bark heat resistance of small trees in Californian mixed conifer forests: Testing some model assumptions. For. Ecol. Manage. 178 (3), 341-352.

van Mantgem, P., Schwarz, M., 2004. An experimental demonstration of stem damage as a predictor of fire-caused mortality for ponderosa pine. Can. J. For. Res. 34(6), 1343-1347.

van Wagtendonk, J.W. 1983. Prescribed fire effects on forest understory mortality. In: Seventh Conference on Forest Fire and Meteorology, 25-28 Apr. 1983. Fort Collins, CO. American Meteorological Society, Boston. Pp. 136-138.

Waldrop, T.A., Lloyd, F.T., 1988. Precommercial thinning of a sapling-sized loblolly pine stand with fire. SJAF. 12, 203-207.

Wyant, J.G., Omi, P.N., Laven, R.D., 1986. Fire induced tree mortality in a Colorado Ponderosa Pine/Douglas-fir stand. For. Sci. 32(1), 49-59.

Wooldridge, D.D., Weaver, H., 1965. Some effects of thinning a ponderosa pine thicket with a prescribed fire, II. J. For. 63, 92-95.

Table 2.1: Ranges of fire weather data for each day of the prescribed fires and wildfire obtained from spot forecast or local remote access weather stations (RAWS).

	Medicine	Rankin Tower	Horse Nugget	Bullock	Lafferty Gulch
Date	10/19/04; 11/18/04	10/25/05	10/28/05; 11/1-2/05	1/24/06	2/26/06 to 3/3/06
Air Temperature (°C)	5.5 to 11	15.5 to 18.8	13.3 to 19.4	7.2 to 10.5	-0.5 to 8.9
Relative Humidity (%)	29 to 48	21 to 30	24 to 40	26 to 33	31 to 48
Wind speed (km/hr)	4.8 to 11.2	1.6 to 8.1	0 to 12.9	3.2 to 4.8	1.6 to 8.1
Wind gusts (km/hr)	16.1 to 24.1	n/a	9.7 to 38.6	27.4	25.8
10-hr fuel moisture (%)	9 to 12	6 to 8	8 to 12	9 to 10	6 to 7
Source	Spot forecast	WICA RAWS[1]	BKF3 portable RAWS[2]	BKF3 portable RAWS[2]	MORU RAWS[3]

[1]43.5575N, -103.4914W
[2]44.0519N, -103.5778W
[3]43.8764N, -103.4575W

Table 2.2: Mean characteristics of live and dead ponderosa pine seedlings (<137 cm tall) and saplings (0.25 to 10 cm diameter at breast height) for prescribed fires in the Black Hills of South Dakota. The P-value shows results of Wilcoxon rank sum test.

Variable	Seedlings (n=1310)			Saplings (n=2341)		
	Live	Dead	P-value	Live	Dead	P-value
Height (cm)	66.9	43.2	<0.0001	N/A	N/A	N/A
dbh (cm)	N/A	N/A	N/A	4.6	3.1	<0.0001
Crown Volume Scorched (%)	45	46.6	0.1625	38.1	80.0	<0.0001
Crown Volume Consumed (%)	0.56	51	<0.0001	0.18	16.4	<0.0001
Crown Volume Damaged (%)	45.6	97.5	<0.0001	38.3	96.4	<0.0001
Basal Char class	0.37	1.25	<0.0001	1.9	3.5	<0.0001
Ground Char severity class	0.35	0.9	<0.0001	1.4	2.3	<0.0001

Basal char severity (%moderate + %high) class rating (0, 0% Basal char severity; 1, >0% and <25% basal char severity; 2, >25% and <50% basal char severity; 3, >50% and <75% basal char severity; or 4, 75% basal char severity); Ground char severity class rating (0, unburned; 1, light; 2, moderate; 3, high).

Table 2.3: Logistic regression coefficients, -2 Log Likelihood ratio statistic (-2LL), receiver operating characteristic curve value (ROC), and overall rate of correctly predicting mortality for ponderosa pine seedlings (< 137 cm tall) following prescribed fires during the dormant season in the Black Hills of South Dakota. Model coefficients: HGT (cm) is seedling height; CVD, crown volume damage (scorch+consumption); CVC, crown volume consumption; GC, ground char severity class rating (0, unburned; 1, light; 2, moderate; 3, high);); BC, basal char severity (%moderate + %high) class rating (0, 0% Basal char severity; 1, >0% and <25% basal char severity; 2, >25% and <50% basal char severity; 3, >50% and <75% basal char severity; or 4, 75% basal char severity). Regression coefficients are significant at *α=<0.0001, ‡ α=0.0003, § α=0.0001, and † α=0.02.

Model	Intercept	HGT (cm)	CVD (%)	GC	BC	Flame length (cm)	-2LL	H-L	ROC	% correct
1. HGT, CVD, GC, BC	-2.7988*	-0.0344*	0.0637*	1.668*	0.3609‡		475.11	0.38	0.97	93
2. HGT, Flame length	2.3757*	-0.0434*				0.0634*	n/a	0.35	0.90	92
3. GC	0.2365†			1.563*			1179.11	0.70	0.72	78

Table 2.4: Logistic regression coefficients, -2 Log Likelihood ratio statistic (-2LL), receiver operating characteristic curve value (ROC), and overall rate of correctly predicting mortality for ponderosa pine saplings (0.25 to 10 cm dbh) following prescribed fires during the dormant season in the Black Hills of South Dakota. Model coefficients: dbh (cm) is diameter at breast height; CVD, crown volume damage (scorch+consumption); CVC, crown volume consumption; GC, ground char severity class rating (0, unburned; 1, light; 2, moderate; 3, high); BC, basal char severity (%moderate + %high) class rating (0, 0% Basal char severity; 1, >0% and <25% basal char severity; 2, >25% and <50% basal char severity; 3, >50% and <75% basal char severity; or 4, 75% basal char severity); Sapling density (trees per hectare). Regression coefficients are significant at *α=<0.0001.

Model	Intercept	dbh (cm)	CVD (%)	CVC (%)	GC	BC	Sapling TPH	-2LL	H-L	ROC	% correct
1. DBH, CVD, GC, Sapling density	-10.299*	-0.2984*	0.1012*		1.433*		0.00057*	802.0	0.61	0.97	92
2. DBH, CVD, BC	-8.742*	-0.2304*	0.0985*			0.6247*		864.9	0.14	0.96	91
3. DBH, CVC	0.0847	-0.1157*		0.1247*				2085.4	0.06	0.71	68
4. GC	-2.196*				1.139*			2029.7	0.49	0.76	72

Figure captions

Figure 2.1: Observed percent mortality of ponderosa pine (a) seedlings (<137 cm tall) and (b) saplings after dormant season fire by size class.

Figure 2.2: Observed percent mortality of ponderosa pine (open circles) seedlings (<137 cm tall) and (closed circles) saplings after dormant season fire by a) total crown volume damage (scorch + consumption), b) Ground char severity, and c) Basal char severity.

Figure 2.3: Estimated mortality of ponderosa pine seedlings (<137 cm tall) predicted by crown volume damage, ground char severity, and basal char severity. Figures depict several combinations: (a and b) unburned ground char with (a) 0 to 25% basal char severity and (b) >75% basal char severity; (c and d) light ground char with (c) 0 to 25% basal char severity and (d) >75% basal char severity; (e and f) moderate ground char with (e) 0 to 25% basal char severity and (f) >75% basal char severity; (g and h) high ground char with (g) 0 to 25% basal char severity and (h)>75% basal char severity. See Table 2.3 for model form and statistics.

Figure 2.4: Estimated mortality of ponderosa pine seedlings (<137 cm tall) predicted by flame length (cm). See Table 2.3 for model form and statistics.

Figure 2.5: Estimated mortality of ponderosa pine saplings (a) 2.5 cm dbh, (b) 5 cm dbh, (c) 7.5 cm dbh, and (d) 10 cm dbh predicted by crown volume damage, ground char

severity (unburned, light, moderate, and high), and sapling density of 1160 trees per ha-1. See Table 2.4 for model form and statistics.

Figure 2.6: Estimated mortality of ponderosa pine saplings (a) 2.5 cm dbh, (b) 5 cm dbh, (c) 7.5 cm dbh, and (d) 10 cm dbh predicted by crown volume damage, and basal char severity (0%, 0 to 25%, 25 to 50%, 50 to 75%, and >75%). See Table 2.4 for model form and statistics.

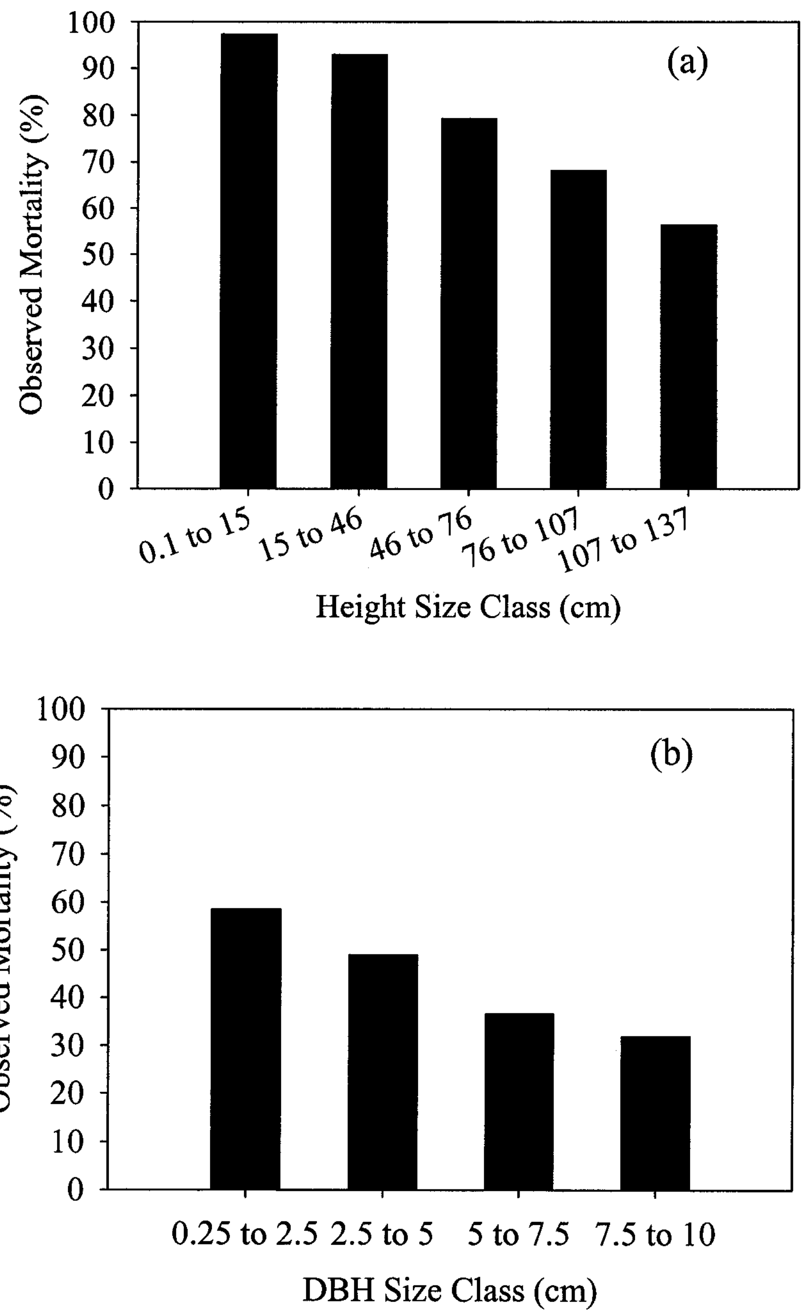

Figure 2.1: Observed percent mortality of ponderosa pine (a) seedlings (<137 cm tall) and (b) saplings after dormant season fire by size class.

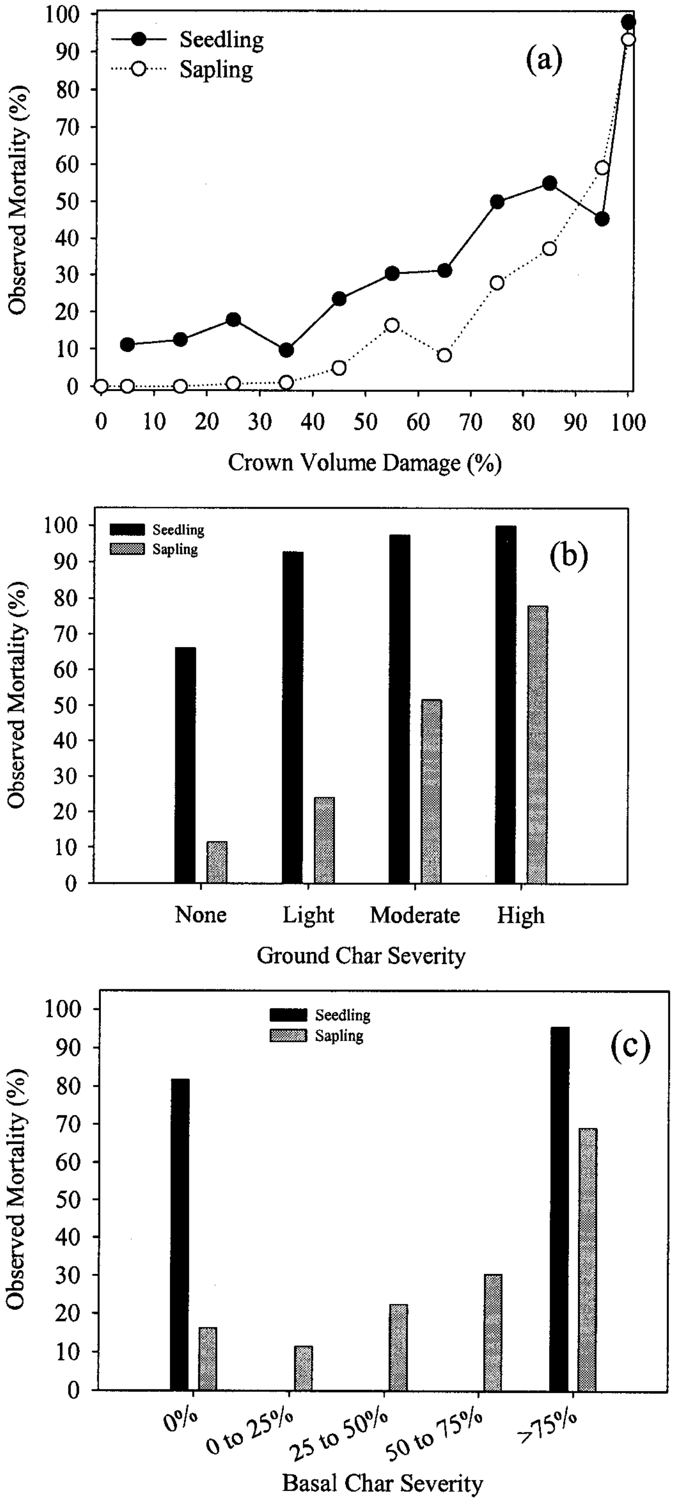

Figure 2.2. Observed percent mortality of ponderosa pine (open circles) seedlings (<137 cm tall) and (closed circles) saplings after dormant season fire by a) total crown volume damage (scorch + consumption), b) Ground char severity, and c) Basal char severity.

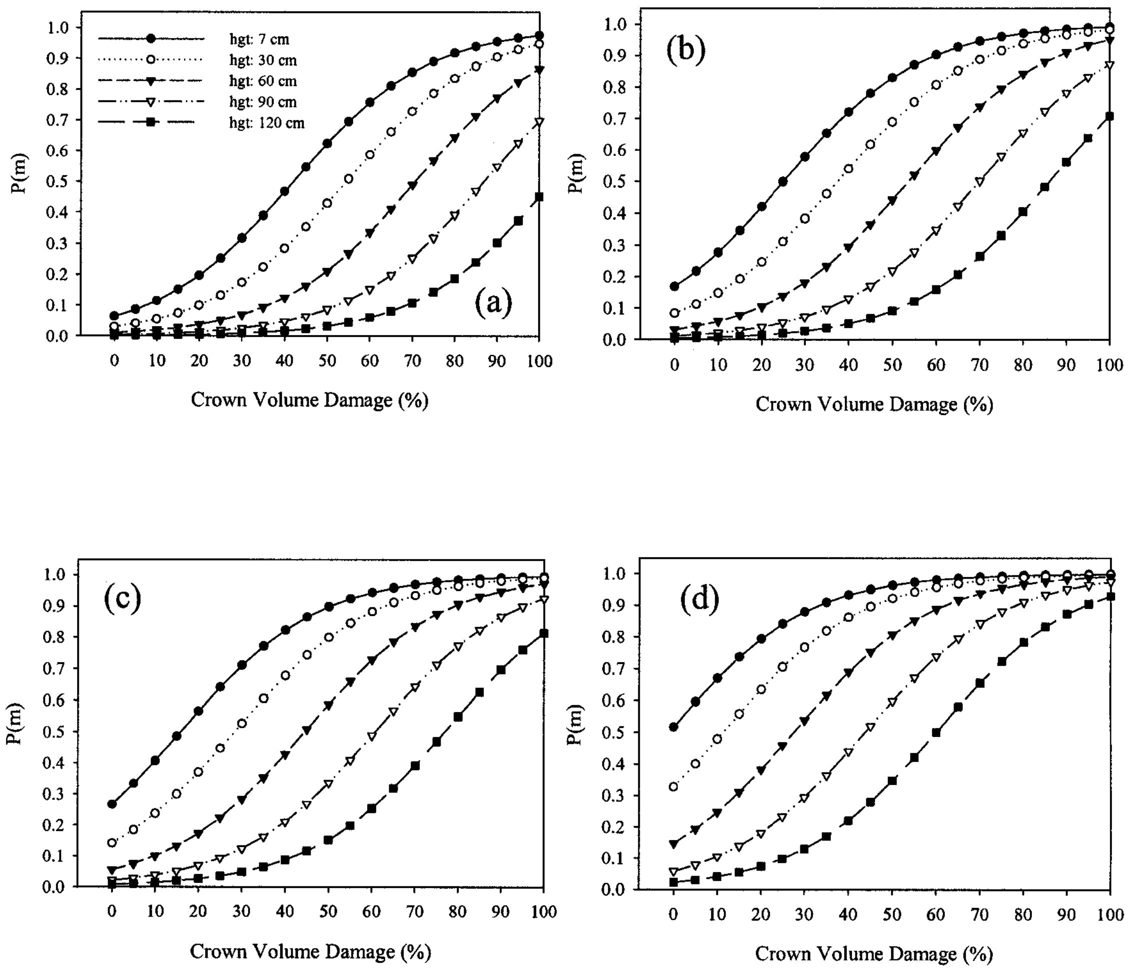

Figure 2.3: Estimated mortality of ponderosa pine seedlings (<137 cm tall) predicted by crown volume damage, ground char severity, and basal char severity. Figures depict several combinations: (a and b) unburned ground char with (a) 0 to 25% basal char severity and (b) >75% basal char severity; (c and d) light ground char with (c) 0 to 25% basal char severity and (d) >75% basal char severity;

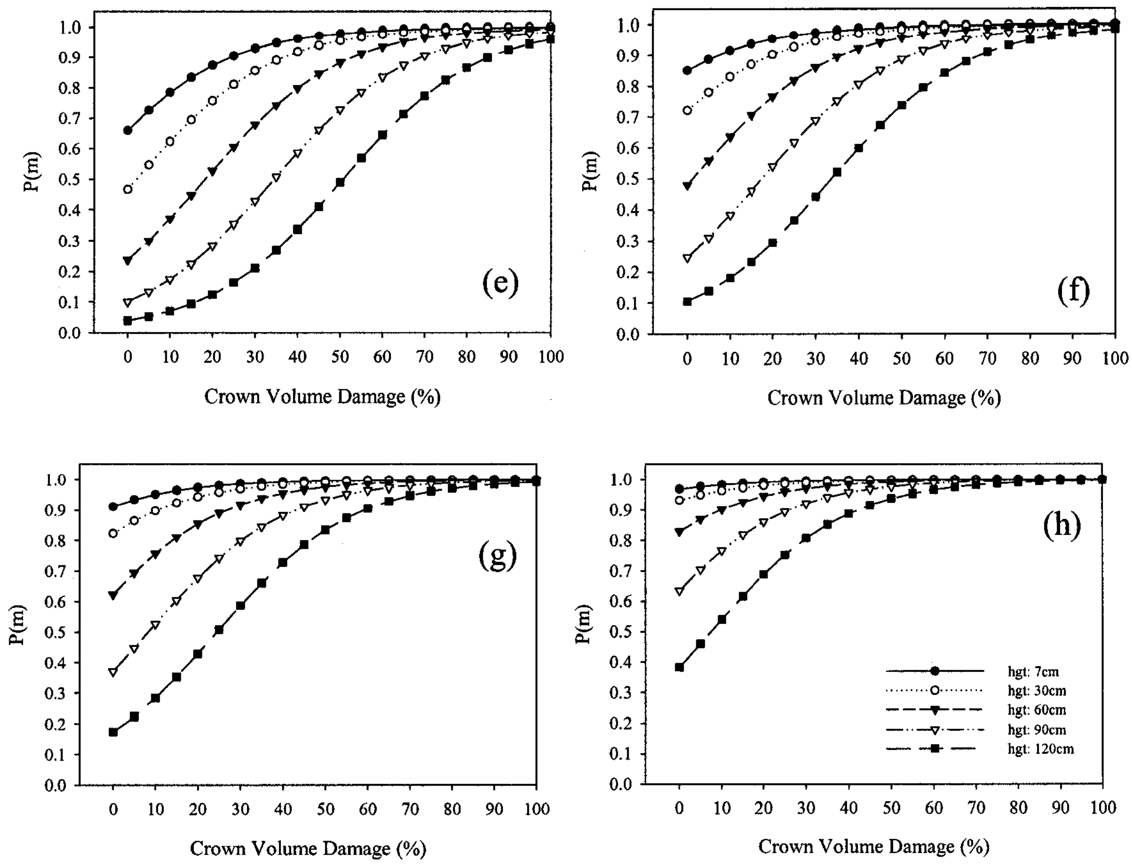

Figure 2.3 cont:. Estimated mortality of ponderosa pine seedlings (<137 cm tall) predicted by crown volume damage, ground char severity, and basal char severity. Figures depict several combinations: (e and f) moderate ground char with (e) 0 to 25% basal char severity and (f) >75% basal char severity; (g and h) high ground char with (g) 0 to 25% basal char severity and (h)>75% basal char severity. See Table 2.3 for model form and statistics.

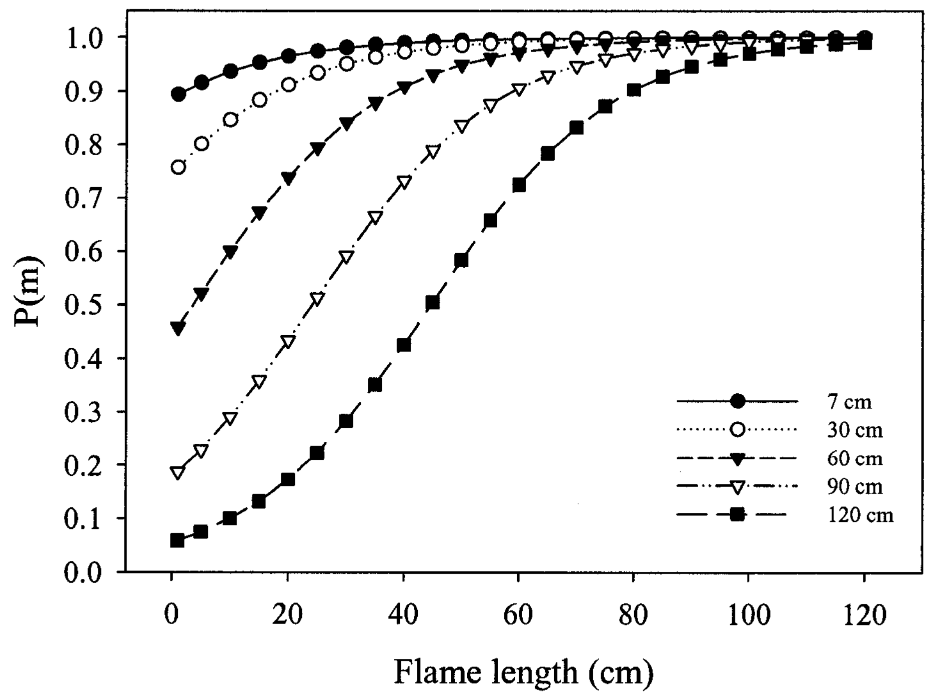

Figure 2.4: Estimated mortality of ponderosa pine seedlings (<137 cm tall) predicted by flame length (cm). See Table 2.3 for model form and statistics.

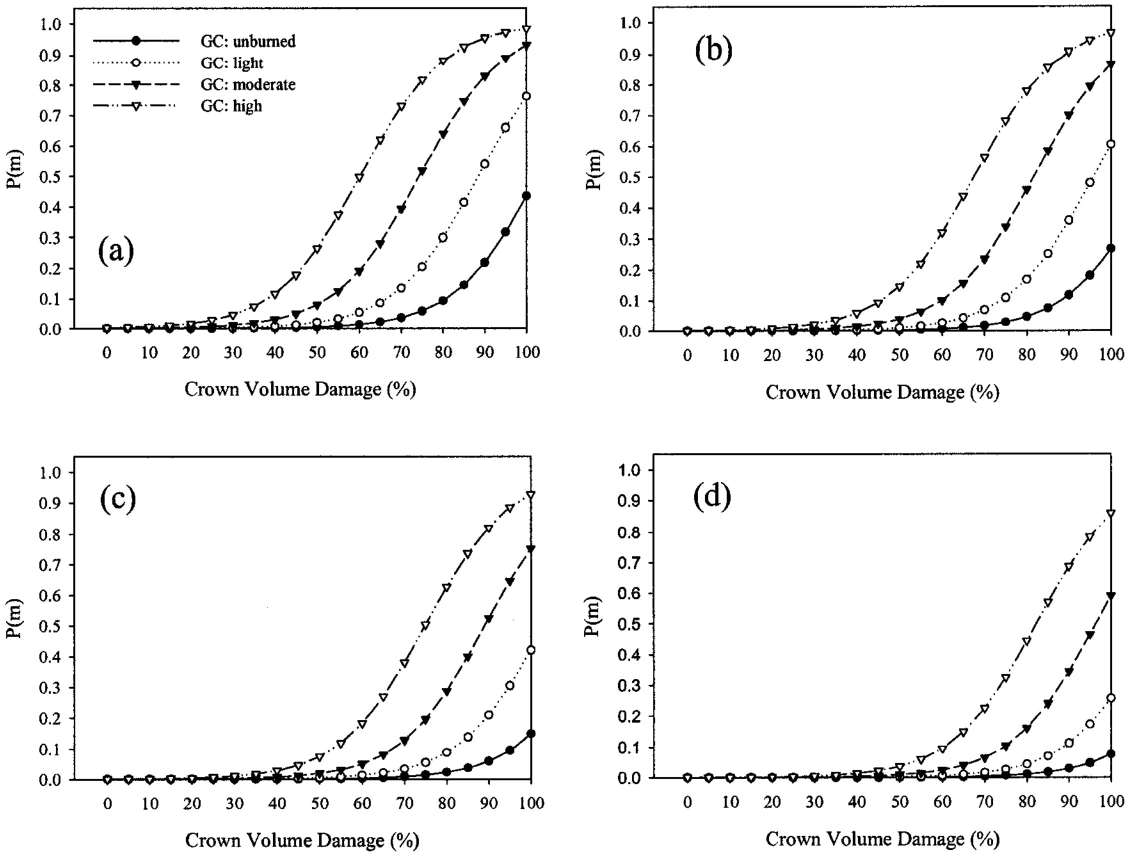

Figure 2.5: Estimated mortality of ponderosa pine saplings (a) 2.5 cm dbh, (b) 5 cm dbh, (c) 7.5 cm dbh, and (d) 10 cm dbh predicted by crown volume damage, ground char severity (unburned, light, moderate, and high), and sapling density of 1160 trees per ha-1. See Table 2.4 for model form and statistics.

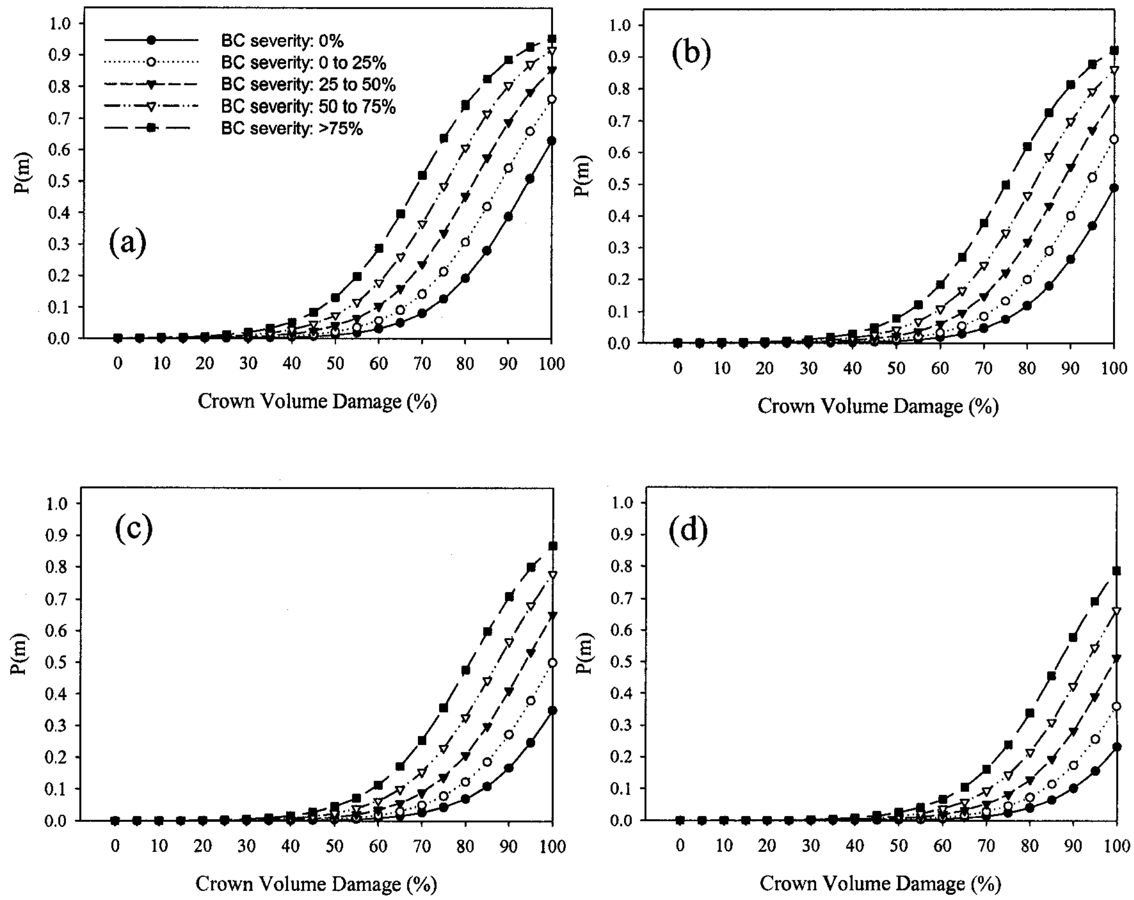

Figure 2.6: Estimated mortality of ponderosa pine saplings (a) 2.5 cm dbh, (b) 5 cm dbh, (c) 7.5 cm dbh, and (d) 10 cm dbh predicted by crown volume damage, and basal char severity (0%, 0 to 25%, 25 to 50%, 50 to 75%, and >75%). See Table 2.4 for model form and statistics.

Acknowledgements

This research was supported by Joint Venture Agreement #03-JV-11221616-232 with the USFS Rocky Mountain Research Station, National Fire Plan project 01.RMS.C.4, and by McIntire-Stennis funding to the Colorado State University Department of Forest, Rangeland, and Watershed Stewardship. The authors would like to thank Gale Gire, Jay Kurth, Gwen Lipp, and Terry Tompkins from the Mystic Ranger District of the Black Hills National Forest and Andy Thorstenson and Cody Wienk from Wind Cave National Park for their help in locating prescribed burns to sample. The authors also thank Lance Asherin, Jay Aylward, Mike Peterkin, Charity Weaver, Andy Whelan, and Vicky Williams for their assistance in field data collection and entry. We also thank Rudy King for statistical consultations and Tara Keyser for valuable suggestions and recommendations during model building.

CHAPTER 3: CAN PRESCRIBED FIRE BE USED TO MAINTAIN FUEL TREATMENT EFFECTIVENESS OVER TIME IN PONDEROSA PINE FORESTS?

Abstract

To maintain fuel-reduction treatment effectiveness in ponderosa pine (*Pinus ponderosa* var. *scopulorum* Dougl. ex Laws.) forests of the Black Hills, the abundant natural regeneration must be controlled. In this paper we examine the utility of using prescribed fire as a technique to control regeneration densities to promote fuel treatment effectiveness over time. The temporal dynamics of ponderosa pine regeneration, surface fuel accumulation, and associated fire behavior under prescribed fire conditions was assessed over a 120-year *time since fire* chronosequence in Black Hills ponderosa pine forests. With this information, the susceptibility of seedlings and saplings to the predicted fire behavior was assessed. Fuel treatments in ponderosa pine forests will lose their effectiveness within 20 to 30 years if regeneration densities are not controlled. The ability to successfully use prescribed fire to maintain low regeneration densities is dependent on seedling/sapling size and the frequency of burns.

Current prescribed fire prescriptions and fuel loads are adequate to maintain low densities of ponderosa pine seedlings (<90 cm tall) if burns occur every 10 to 15 years. If burn intervals exceed 15 years, regeneration can attain sizes that reduce susceptibility to prescribed fire under typical operational weather conditions. After 20 to 25 years, ponderosa pine can reach sizes that will require flame length that exceed 1 to 2 meters to

obtain significant mortality. Achieving these flame lengths will require burning under drier weather conditions to allow coarse woody debris (CWD) consumption to contribute to the fire intensity. Alternatively, managers can augment sites with activity fuels to increase fine fuel loadings to achieve flame lengths that will increase sapling susceptibility to fire, while burning when CWD fuel moistures are high to limit the potential for spotting. While the flames might kill sapling-sized trees, mature overstory trees could still maintain resiliency to fire.

Introduction

The goal of a fuel-reduction treatment is frequently to decrease the chance of crown fire in a stand during a wildfire. Mechanical thinnings are applied to reduce the horizontal and vertical continuity of canopy fuels. Sometimes mechanical treatments are followed by prescribed fire to reduce the surface fuel loadings associated with the thinning treatment. The reduction in canopy and surface fuels results in a lower risk of crown fire initiation and spread (Agee and Skinner 2005, Cram *et al.* 2006). However, fuel-reduction treatments have potential problems. For instance, if treatments that reduce overstory density do not thin the smaller unmerchantable trees, the potential for passive crown fire is still present. Moreover, the partial removal of the overstory can increase growth of advance regeneration and decrease the time needed to connect the canopy and surface fuel complex (Keyes and Varner 2006). Initial applications of prescribed fire used to reduce post-thinning surface fuels can subsequently lead to increased fuel loadings as a result of small tree mortality (Parsons 1978, Thomas and Agee 1986, Agee and Huff 1987, Agee 2000, Stephens and Moghaddas 2005). Also, the partial removal of the overstory and/or the reduction of the forest floor layer with fire are known to promote the initiation of a new ponderosa pine (*Pinus ponderosa* var. *scopulorum* Dougl. ex Laws.) cohort (Shepperd and Battaglia 2002, Bonnet *et al.* 2005, Keyser 2007). New regeneration, acceleration of small tree growth rate and surface fuels may ultimately reduce the efficacy of the fuel reduction treatment. In this paper we explore the utility of using prescribed fire as a technique to maintain low densities of ponderosa pine regeneration to promote fuel treatment effectiveness over time.

In the Black Hills, favorable growing conditions and good seed crops often coincide and result in abundant natural ponderosa pine regeneration establishment, often exceeding 1000 seedlings ha^{-1} (Shepperd and Battaglia 2002). This establishment is often enhanced after mechanical treatments as a result of scarification or fire, both of which provide the mineral soil seedbed favored by ponderosa pine (Oliver and Ryker 1990, Shepperd and Battaglia 2002, Bonnet *et al.* 2005, Shepperd *et al.* 2006). Once established, the time it takes a seedling to reach a height that connects the surface fuel complex to the canopy is important in determining when a fuel treatment loses its ability to reduce crown fire initiation. Furthermore, the susceptibility of ponderosa pine to fire changes with tree size with smaller trees requiring lower values of crown damage, cambial damage, and root damage than larger trees (Battaglia 2007). These temporal differences in seedling and sapling susceptibility to fire highlight the importance of understanding the regeneration dynamics of ponderosa pine in the Black Hills when determining when to use fire to maintain the long-term effectiveness of a fuel treatment.

The ability to predict mortality based on potential fire behavior is important so managers can tailor burn prescriptions to meet their regeneration reduction/maintenance targets objectives (Johnson and Miyanishi 1995). Fire managers frequently use

BehavePlus, a fire behavior simulation program (Andrews *et al.* 2005), to predict flame lengths and fireline intensity under various fuels, weather, and topographic scenarios. While mortality models such as FOFEM (Reinhardt 2003) and those included with BehavePlus do predict tree mortality based on flame length and fireline intensity (Van Wagner 1977), the equations used in these models were developed for trees > 10 cm diameter at breast height (dbh) and often over predict mortality in seedlings and saplings (Battaglia 2007). Direct measures of flame lengths during prescribed burns have been used to model mortality of ponderosa pine seedlings (<137 cm tall) in the Black Hills (Battaglia 2007) and the Sierra Nevada (van Wagtendonk 1983). In both studies, taller flame lengths were needed to increase the probability of seedling mortality as seedlings grew taller. However, models relating observed fire behavior to sapling-sized ponderosa pine mortality are still needed for better mortality estimates.

The amount of fuel available to burn influences the potential flame lengths during a fire. Therefore, documentation of fuel accumulation rates in fuel treatments maintained by fire is needed to determine if an area has enough fuel to re-burn with a prescribed fire to produce the desired flame lengths. Data on fuel accumulation rates after fire for the Black Hills is limited to one recent study investigating the recovery after a mixed-severity wildfire (Keyser 2007). In the low burn severity areas of that study, needle litter mass had recovered to 75% of the unburned areas within 5 years (Keyser 2007). Fine (< 7.62 cm diameter) and coarse (>7.62 cm diameter) woody debris had each recovered to approximately 40% of the unburned areas. Longer term data is needed to determine whether surface fuels continue to accumulate or if they reach an equilibrium state. In addition, fuel accumulation data from a variety of forest densities is needed because certain fuel loadings are known to increase with overstory density (Fulé and Covington 1994). Quantifying the temporal dynamics of fuel accumulation within a fuel treatment maintained by prescribed fire is needed to understand how potential fire behavior and seedling/sapling susceptibility might change over time.

While initial fuel reduction treatments may reduce crown fire risk in the short term, the abundant natural regeneration in the Black Hills poses a future management problem. The successful use of prescribed fire to maintain low regeneration densities will require a combined understanding of regeneration dynamics, surface fuel accumulation, fire behavior, and regeneration size susceptibility to fire. Our objectives were to (*i*) quantify growth rates of ponderosa pine seedling and sapling growth to determine how long it takes to reach a certain size (both height and diameter), (*ii*) quantify post-fire fuel accumulation rates, (*iii*) estimate potential fire behavior associated with the temporal changes in fuel loadings and (*iv)* determine the relationship between fire behavior and seedling/sapling mortality. With this information, we assess the potential of using prescribed fire to control ponderosa pine densities with potential surface fuel loadings and determine if other treatments will be required to successfully maintain fuel treatment effectiveness.

Methods

Study area

The Black Hills is an isolated forested geologic uplift located in southwest South Dakota and northeast Wyoming that extends 200 km from north to south and 100 km from east to west (Shepperd and Battaglia 2002). Elevations in the Black Hills range from 1,000 m to 2,207 m and are approximately 300 to 1,200 m above the surrounding

Great Plains. The increased elevation results in an orographically induced microclimate that increases precipitation. Annual average precipitation ranges from 41 cm in the south to 74 cm in the north. Most of the precipitation occurs from April to August, with May and June receiving 33% of annual precipitation (Driscoll *et al.* 2000). Annual average temperature ranges from 2.9° to 9° C. The frequent rain showers during the early growing season in combination with the warm temperatures are conducive to prolific natural ponderosa pine regeneration (Shepperd and Battaglia 2002). Ponderosa pine dominates 85% of the forested land base (DeBlander 2002) and is found at all elevations, soil types, and aspects. It can be found mixed with white spruce (*Picea glauca* [Moench] Voss) and aspen (*Populus tremuloides* Michx.) in the moister, higher elevations.

Ladder fuel development

Sites within three fuel treatment areas (Buffalo, Lookout, and Medicine) on the Mystic Ranger District of the Black Hills that had canopy coverage < 40% and consisted of a broad range of ponderosa pine seedling and sapling sizes were sampled to determine seedling and sapling growth. Buffalo, a 182 ha unit, was located 21 km northeast of Hill City, South Dakota (approximately 44° 07' N, 103° 32'). Lookout, a 223 ha unit, was located 24 km northwest of Hill City, South Dakota (approximately 44° 02' N, 103° 50'). Medicine, a 728 ha unit, was located 9.7 km west southwest of Hill City, South Dakota east of Medicine Mountain (approximately 43° 55' N, 103° 42'). These areas are representative of the geomorphological portion of the Black Hills known as the central crystalline core (Shepperd and Battaglia 2002).

Ten sampling points within each unit were located using a random number generator based on the UTM coordinates that fell within the sampling area. At each point, regeneration was classified into size classes based on height (0 to 15 cm, 15 to 46 cm, 46 to 76 cm, 76 to 107 cm, and 107 to 137 cm) or diameter at breast height (dbh) if taller than 137 cm (0.25 to 2.54 cm, 2.54 to 5.1 cm, 5.1 to 7.62 cm, 7.62 to 10.2 cm and 10.2 to 12.7) as defined by the Fire Effects Monitoring and Inventory Protocol (Lutes *et al.* 2006).

At each point, the closest ponderosa pine tree to the center of the plot in each size class was destructively harvested. Trees were harvested by cutting the tree as close to the root collar as possible with a bow saw or loppers. Total tree height was measured for each tree. A cross-section of the stem was cut and brought back to the lab for ring analysis to determine tree age. Cross-sections were surfaced and rings were counted to determine age. A total of 10 trees for each size class per treatment unit (n=3) were destructively sampled to determine an age-height relationship resulting in 30 trees per size class.

Fuel accumulation: Time Since Fire

A chronosequence of *22* sites with increasing time since fire (120 years) were selected from previously published fire history studies (Brown *et al.* 2000, Brown 2003) and GIS fire history maps of the Black Hills National Forest (BHNF). Sampling of recent (<15 years) fires were limited to prescribed fires. Locations of prescribed fires were obtained from local Forest Service district staff. Because the prescribed burning program on the BHNF before 1990 was limited, we sampled wildfires >15 years old that burned with low to moderate severity.

Within each fire perimeter, the BHNF resource information system (RIS) database was used to identify potential stands. This database contains information on stand structure, cover type, and habitat structural stage. Habitat structural stage (HSS) is an index of stand diameter (3 = 12.7 to 22.9 cm dbh; 4 = 22.9 to 35.6 cm dbh; 5 = >35.6 inch dbh) and overstory canopy coverage (A=0 to 40%; B = 40 to 70%; C = >70%). Within each fire perimeter, a stand with HSS of 4A (open-canopied) and/or 4C (closed-canopied) was randomly chosen to sample. If there was only one HSS represented within a fire, that stand was measured. In some of the fires, both HSS 4A and 4C were not available so the HSS that was available was measured. In a few cases, a HSS 3C or 4B was measured as a surrogate for a 4C stand because the diameter or canopy coverage was at the upper limit of the classification. Because HSS 5 (mature) stands are limited in the Black Hills, only two were measured. The mature stands were located in the Black Elk Wilderness Area in the Upper Pine Creek Natural Area (Ryan *et al.* 1994) and in the Mount Rushmore National Monument. Selected potential sites were ground-truthed and either accepted or rejected based on evidence of burning and harvesting activity. Evidence of burning was determined by the presence of char on trees and downed logs. Harvesting activity was assessed by the presence of recent stumps. Elevations of the sites ranged from 1450 to 1950 m (Table 3.1). Sites were dominated by ponderosa pine (Table 3.2) with some sporadic aspen and bur oak (*Quercus macrocarpa* Michx.).

Within each *time since fire* stand, fifteen sample points were randomly located using a random number generator based on UTM coordinates that fell within the stand. Surface fuels and forest floor depths were inventoried as described above for the fuel accumulation/overstory density experiment, except transect direction was determined randomly. Five stands were measured in the summer of 2004, three stands were measured in the summer of 2005, and 14 stands were measured in the summer of 2006.

Overstory tree density was measured at all sites with a BAF 10 prism from the center of each sample point. Sapling and seedling densities were also sampled on sites measured in 2006 (n=14) using a nested plot design consisting of a 2 m radius circular plot to sample seedlings (trees < 137 cm tall) located with a larger circular plot to sample saplings (trees 0.25 to 10 cm dbh). Dead trees (snags) were also measured on sites sampled in 2006 using the same measurement protocols.

In five *time since fire* stands, a 0.025 m^2 PVC-frame was used to collect litter and duff layers at the each sampling plane point to aid in the development of Black Hills specific bulk density estimates. Litter and duff layers were bagged separately and oven-dried at 65°C until weight loss stabilized. Litter and duff samples were weighed and then combusted at 400°C to obtain the organic matter weight of each sample. Bulk density of each litter and duff sample was calculated with the following equation:

Bulk density = Organic matter weight / (frame area * layer depth) [Eq. 1],

where, bulk density is in kg m^{-3}, organic matter weight is in kg m^{-2}, layer depth is in m. For Black Hills ponderosa pine forests, needle litter bulk density is 60.71 $\pm$ 3.1 kg m^{-3} and duff bulk density is 102.58 $\pm$ 2.9 kg m^{-3}.

Statistical analysis

Ladder fuel development

Regression analysis was performed with PROC GLM (SAS Institute 2001) to determine a predictive relationship between tree age and height. Tree age and height was natural log transformed to fulfill normality and constant variance assumptions.

Fuel accumulation: Time Since Fire

Time since fire stands were placed into 10-year interval classes and summary statistics were calculated for fuel loadings, snag densities, and ladder fuel densities for each interval within a HSS using PROC MEANS (SAS Institute 2001). Significant differences among the sample means were not tested since stands were not replicated. To estimate fuel accumulation over time, gamma distributions combined with a power function were fit using PROC NLIN (SAS Institute 2001) for the average fine (sum of 1 hour, 10 hour, and 100 hour fuels) fuels, sound >7.62 cm fuels, rotten >7.62 cm fuels, and total fuels (Hall *et al.* 2006). All regression analyses were checked for normality and constant variance assumptions. If assumptions were violated, data was natural log transformed and regression analysis and assumptions were tested.

Fire behavior and mortality: Time Since Fire

BehavePlus 3.0.2 (Andrews *et al.* 2005) was used to estimate fire behavior for the predicted fine (<7.62 cm) fuel loadings over time. To estimate surface fire behavior, BehavePlus requires inputs of 1 hour, 10 hour, and 100 hour fuel loadings. These fuel loadings were estimated by calculating the proportion of each size class that made up the total predicted fine fuel loading over time. These values were entered into BehavePlus to estimate potential flame lengths over time for the open- and closed-canopied forests. Fire behavior was calculated with average prescribed burning weather conditions commonly used on the Black Hills: 1 hour fuel moisture = 7%, 10 hour fuel moisture = 8%, 100 hour fuel moisture = 11%, midflame windspeed = 5 km/hr. Slope was set at 15%.

Results

Ladder fuel development

In the Black Hills, a ponderosa pine seedling can grow up to 15 cm tall in 5 years (Figure 3.1a) and up to 76 cm tall within 7 to 8 years (Figure 3.1a). Within 15 years of establishment, ponderosa pines in the Black Hills can reach the threshold for dbh (Figure 3.1b). Within 20 to 30 years, ponderosa pine saplings can range from 0.25 to 12.7 cm dbh (Figure 3.1b) and 2.2 to 4.3 meters tall (Figure 3.2).

Seedling and sapling densities in relation to *time since fire* were only measured in summer of 2006, so the interpretation for the open-canopied stands is limited to fires that burned greater than 20 years ago, but spans the entire chronosequence for the closed-canopied stands.

Overall, ponderosa pine seedlings and saplings reestablish and contribute to the ladder fuel complex within 10 to 20 years post-fire (Table 3.3). Once established, this ladder fuel complex persists in the absence of fire and shifts from seedling- to sapling-sized trees (Table 3.3). Seedling (<137 cm tall) and sapling (0.25 to 10 cm dbh) densities do vary with time since fire, but in general, total densities exceed several thousand ha^{-1} (Table 3.3).

In open-canopied forests, seedling densities following fires that burned 20 to 49 years ago were approximately 2000 to 2500 ha^{-1} (Table 3.3). Sapling densities were lower, averaging about 400 saplings ha^{-1} (Table 3.3). Seedling densities in the older fires were low (109 ha^{-1}) and sapling density was approximately 300 ha^{-1} (Table 3.3).

Ponderosa pine seedling and sapling densities in the closed-canopied forests were abundant. Within 16 years post-fire, seedling densities exceeded 20,000 ha^{-1} and sapling densities were about 250 ha^{-1} (Table 3.3). Fires that burned 30 to 49 years ago, had seedling densities ranging from 6700 to 8000 ha^{-1} (Table 3.3), but older fires (67 to 89 year old fires) had few seedlings or none at all (Table 3.3). Instead, these older fires had sapling densities ranging from approximately 150 to 600 ha^{-1} (Table 3.3). Sapling densities in the fires 40 to 59 years old were much higher, ranging from 950 to 2100 saplings ha^{-1} (Table 3.3).

The mature stand in Mount Rushmore National Monument, which had not burned in 113 years, had high seedling and sapling densities. Seedling densities exceeded 6000 ha^{-1} and saplings exceeded 2000 ha^{-1} (Table 3.3).

Snags

Since snags contribute to future surface fuel loadings, their presence and size distribution in a stand can impact the long-term effectiveness of a fuel treatment and provide future fuel for prescribed fires. Snag densities were only measured in the *time since fire* stands measured in summer of 2006, so the interpretation for the open-canopied stands is limited to fires that burned greater than 20 years ago, but spans the entire time for the closed-canopied stands.

Snag densities of sapling-sized (0.25 to 10 cm dbh) ponderosa pines varied with time since fire and stand structure (Table 3.2). Sapling snag densities were lowest in the open-canopied forests, with 8 to 9 snags ha^{-1}. Sapling snag densities in closed-canopied stands ranged from 0 to 132 snag ha^{-1} with the highest sapling snag densities in stands with >50 years since the last fire (Table 3.2). The mature stands had the highest sapling snag density with 254 snags ha^{-1} (Table 3.2).

Ponderosa pine snags >10 cm dbh were greatest in stands that burned recently (within the last 30 years) and sustained 10 to 19 snags ha^{-1} >40 years postfire (Table 3.2), regardless of stand structure. Within the first 10 years after fire, snag densities of ponderosa pine>10 cm dbh averaged 164 snags ha^{-1} in closed-canopied forests (Table 3.2). In open-canopied forests, snag densities were greatest 20 to 29 years after the fire event (Table 3.2). However, most of the snags >10 cm dbh were less than 25 cm dbh (Table 3.2). Mature stands had 29 snags ha^{-1} >10 cm dbh (Table 3.2).

Snag densities of large diameter (>25 cm dbh) ponderosa pine trees was highest between 20 and 49 years after fire and were lowest in the most recent fires and older fires (Table 3.2). Snags densities ranged between 0 and 5 snags ha^{-1} in the most recent and oldest fires, but increased to 11 to 17 snags ha^{-1} in the middle-aged fires (Table 3.2). In the baseline forest, densities averaged 12 snags ha^{-1} (Table 3.2).

Surface fuel accumulation

Needle litter is a primary carrier of fire and its presence indicates that fire spread will be slow and flame lengths will be short relative to a grass fuelbed. The amount of needle litter available for consumption can also impact potential tree mortality, so an understanding of how much is present is also important. In general, open-canopied stands had shallower needle litter depth and lower needle litter mass than the closed-canopied stands (Table 3.4). Within the first 20 years post-fire, open-canopied stands had litter depths of 2.2 cm and mass of 12.9 Mg ha^{-1} (Table 3.4). Closed-canopied stands had slower accumulation within the first 10 years, with needle litter depth of 1.4 cm and mass of 8.4 Mg ha^{-1} (Table 3.4). However, after 10 years, needle litter mass in closed-canopied stands stabilized at about 15-16 Mg ha^{-1} and litter depth recovered to approximately 2.5

cm (Table 3.4), both of which are similar to those observed in the mature (HSS 5) stands (Table 3.4).

The amount of duff available to smolder during a fire can influence the amount of root and cambial damage to a tree. In general, as time since fire increased, open-canopied stands had shallower duff depths and lower duff mass than closed-canopied stands (Table 3.4). In the first 29 years post-fire, open-canopied stands had duff depths ranging from 0.36 to 1.1 cm with duff mass ranging from 3.7 to 11.1 Mg ha^{-1} (Table 3.4). Duff depth and mass in the closed-canopied forests in the first 39 years post-fire ranged between 0.73 to 0.80 cm and 7.5 to 10.1 Mg ha^{-1}, respectively (Table 3.4). Open-canopied stands never reached values observed in mature stands, but closed-canopied stands did within 100 years (Table 3.4).

The amount of fine woody fuels available for consumption influences flame lengths, rate of spread, and subsequent tree mortality during a prescribed burn. Fine woody debris patterns varied with stand structure, but time since fire explained between 41 to 93% of the variability (Table 3.5). In open-canopied forests, fine woody fuel accumulation increased rapidly to approximately 11.6 Mg ha^{-1} within the first 25 years after fire and gradually decreased and stabilized around 4 Mg ha^{-1} (Figure 3.3). In contrast, the highest fine woody fuel loadings in the closed canopied forests occurred within 10 years after fire at approximately 6 Mg ha^{-1} (Figure 3.3). Within 19 years after fire, fine fuels stabilized at 4 Mg ha^{-1} (Figure 3.3).

Temporal dynamics of sound coarse woody debris (CWD) showed no significant pattern over time in either the open- or closed-canopied forests (Figure 3.4; Table 3.5). Open-canopied forests had sound CWD fuel loads that ranged from 1.6 to 11.0 Mg ha^{-1}, with the majority of stands having 3 to 4 Mg ha^{-1}. Closed-canopied forests had sound CWD fuels that ranged from 2.5 to 8.2 Mg ha^{-1} (Figure 3.4b), with most stands around 7 to 8 Mg ha^{-1} (Figure 3.4b). In the mature forest, average sound CWD fuel was 15.2 Mg ha^{-1} (Figure 3.4); fuel loads that were much higher than those observed in either the open- or closed canopied forests.

Rotten CWD fuel accumulation did show a significant pattern with time since fire (Table 3.5), although the peak differed with stand structure. In open-canopied forests, rotten CWD peaked between 20 to 29 years after fire (Figure 3.5a). Rotten CWD ranged from 0.38 to 31.8 Mg ha^{-1}, with fuels predicted to stabilize after 65 years at approximately 2.0 Mg ha^{-1} (Figure 3.5a). In closed-canopied forests, rotten CWD showed a gradual increase in fuel accumulation between 20 and 60 years postfire (Figure 3.5b). Rotten CWD in closed-canopied forests were typically low (0 to 2 Mg ha^{-1}), except for the fires between 40 (11.3 Mg ha^{-1}) and 59 (8.1 Mg ha^{-1}) years old (Figure 3.5b). Rotten CWD fuel loads in the mature forest were similar to the other forests with average fuel loadings of 3.3 Mg ha^{-1} (Figure 3.5).

Overall, the sound and rotten CWD fuels made up the majority of total woody fuel loadings and followed the same temporal patterns of the rotten wood component (Figure 3.6). Open-canopied forests had total fuel loadings that ranged from 4.3 to 54.3 Mg ha^{-1}, with the majority of stands ranging from 8.5 to 11 Mg ha^{-1} (Figure 3.6a). Closed-canopied forests had total fuel loadings that ranged from 4.7 to 20.3 Mg ha^{-1}, with the majority of stands between 12 and 13.5 Mg ha^{-1} (Figure 3.6b). Mature forests had higher total fuel loads than the open- or closed-canopied forests, with an average of 23.6 Mg ha^{-1} (Figure 3.6).

Fire behavior over time

Potential surface fire behavior over time during prescribed burns varied for the open- and closed canopied forests due to the differences in post-fire fine fuel accumulation (Figure 3.7). Over time, open-canopied forests had taller predicted flame lengths than the closed-canopied forests; except for the first 10 years post-fire (Figure 3.7). In the first 5 years post-fire, closed-canopied forests had predicted flame lengths about 0.43 m, whereas open-canopied forests had flame lengths of 0.27 m (Figure 3.7). Within 10 years, open- and closed-canopied forests had flame lengths of 0.43 m. After 15 years, flame lengths declined for the closed-canopied forests and stabilized at 0.27 m around 25 years post-fire (Figure 3.7). In contrast, after 15 years, flame lengths continued to increase in open-canopied forests and reached a maximum of 0.64 m at 20 years post-fire. After 20 years, flame lengths declined gradually and stabilized at 0.27 m around 60 years postfire (Figure 3.7).

Discussion

In ponderosa pine forests of the Black Hills, the potential to develop ladder fuels within a couple of decades after a fuel treatment is high if regeneration densities are not regulated. In the absence of mechanical or prescribed fire treatment, this ladder fuel complex remains and shifts from seedling- to sapling-sized trees. As these seedlings and saplings increase in height and diameter, their susceptibility to mortality during prescribed fire decreases. The situation is complicated by the temporal dynamics of surface fuel availability and the low-intensity fire behavior that is typical of prescribed fires. These factors impact the ability to use prescribed fire to regulate ponderosa pine seedling and sapling density. Overall, current prescribed fire prescriptions can be successful in regulating ponderosa pine seedlings, but higher fuel loads or burning when fuel moistures are lower will be needed to reduce high densities of saplings in Black Hills ponderosa pine forests.

The size distribution of ponderosa pine regeneration in fuel treatments over time is influenced by several factors and should be considered in the planning process. Site productivity and stand density can influence the rate at which a seedling or sapling reaches a certain size (Shepperd *et al.* 2006). This age-size differentiation was reflected in our study by the variability in tree age with seedling height and diameter, especially as trees reach 20 to 30 years old (Figure 3.1 and Figure 3.2). Seedling and sapling size distribution over time in a fuel treatment would also be influenced by the immediacy of regeneration establishment after treatment. Furthermore, the presence of any advance regeneration which was not mechanically thinned or killed in an initial prescribed fire would also provide for some size differentiation. These possible differences would be site specific and should be considered when future maintenance activities of fuel treatments are in the planning stage.

The temporal change in regeneration size distribution in fuel treatments will impact the potential for crown fire behavior and overstory mortality. With an assumption that regeneration establishes soon after a fuel treatment is implemented, the majority of regeneration within the first 10 years will be < 1 meter tall (Figure 3.1a, Figure 3.2, Table 3.3). Seedlings of this size would not likely transfer a surface fire to the overstory canopy during a wildfire. However, within 20 years of regeneration establishment, the probability of passive crown fire initiation does increase for two reasons. First, the size distribution of seedling regeneration can now span from <15 cm tall to 137 cm tall

(Figure 3.1b and Figure 3.2; Table 3.3). Second, within 20 years, the seedlings that established immediately post-treatment have now reached diameters of 2.5 cm and are approximately 2 meters tall (Figure 3.1b and Figure 3.2). The presence of a continuous fuel strata from <15 cm to 2 meters would likely allow a surface fire to transfer into the crowns of the saplings and allow flames to heat the foliage of the overstory. Within 30 years of regeneration establishment, the initial fuel treatment's ability to reduce crown fire risk is diminished. Ponderosa pine regeneration size distribution could range from seedlings <15 cm tall to saplings that are 5 meters tall (Figure 3.1 and 3.2; Table 3.3). This continuous fuel stratum would likely allow surface fire to transfer into the overstory and result in passive crown fire behavior under moderate weather conditions, but possibly active crown fire behavior if windspeeds were extreme enough. As fuel treatments get older and are not maintained, regeneration will continuously establish until all growing space is occupied and eventually succumb to density-dependent mortality (Table 3.2) as observed in some of the older closed-canopied forests (Table 3.2). This highlights the necessity of mechanical and/or prescribed fire treatments to control ponderosa pine regeneration density in the Black Hills to reduce the risk of crown fire.

The use of prescribed fire to control ponderosa pine regeneration will require enough fine (<7.62 cm) fuel accumulation to carry a fire and produce intensities high enough to result in mortality. Although fine fuel loadings are often reduced by 50 to 70% (Thomas and Agee 1986, Sackett and Haase 1998, Fulé *et al.* 2002, Fulé *et al.* 2006) immediately after a prescribed fire, inputs from small tree mortality can aid in the recovery of these fuels. Thomas and Agee (1986) noted that within 5 to 10 years, fine fuels accumulated from the mortality of understory trees and recovered to 71% of the pre-burn levels in mixed conifer forests of Oregon. Parsons (1978) reported that fine fuels returned to pre-burn levels within 7 years in sequoia (*Sequoiadendron giganteum* [Lindl.] Buchh.)-mixed conifer forests, likely due to inputs from small tree mortality. In our study, we observed an *increase* in fine fuel loadings within 10 years post-fire, but stand structure influenced the magnitude and timing of the peak and stabilization in fuel loading. Within the first 10 years after fire, predicted total fine fuel loadings increased 64% in open-canopied forests and 55% in closed-canopied forests compared to the baseline value observed in areas not burned for decades (Figure 3.3). However, within 25 years post-fire, fine fuel loadings decreased in the closed-canopied forests to approximately pre-fire levels (Figure 3.3b), but continued to increase in the open-canopied forest with substantially higher fuels loads for several more decades (Figure 3.3a). The temporal pattern in fine fuel load accumulation suggests that mortality and subsequent falling of the regeneration and small diameter trees occurs within a few years in the closed-canopied forests (Table 3.2), but is delayed for several years in open-canopied forests (Table 3.2). In addition, the continued increase in fine fuel loadings in the open-canopied forests is likely due to the initial death of lower branches from the fire and the gradual abscission of these branches over time. In later years, inputs from dead trees and branch breakage helps maintain stable fine fuels loadings for both forest structures (Lundquist 2007). The early differences in fine fuel accumulation rates for open-canopied and closed-canopied forests would result in different potential prescribed fire behavior over time and impact regeneration susceptibility to fire.

The success of using prescribed fire to control regeneration density in fuel treatments will be dependent upon the frequency of burning and overstory stand structure. We examined the temporal pattern of predicted flame lengths that were associated with fine fuel loadings (Figure 3.7) and predicted mortality of seedlings and saplings based on these flame lengths (Figure 3.8). Within the first 10 years, when most of the new regeneration is < 100 cm tall (Figure 3.1), seedlings < 90 cm tall would be highly susceptible to another prescribed fire in both open- and closed-canopied forests (Figure 3.9 a and b). Predicted flame lengths at 10 years since fire were approximately 0.4 m (Figure 3.7) and would have resulted in torching fire behavior within the seedling regeneration layer with high levels of crown consumption and subsequently higher mortality rates (Battaglia 2007). If there were seedlings 120 cm tall present in the open-canopied stand, approximately 50% of them were predicted to die (Figure 3.9 a and b), likely due to less crown consumption (Battaglia 2007). The susceptibility of the advance regeneration that survived the initial treatment entry would depend on the sapling size. For instance, saplings up to 2.5 cm dbh would likely experience 50% mortality, but saplings > 2.5 cm dbh would be between 5 and 15% mortality (Figure 3.10 a and b).

If managers wait 15 years after a fuel treatment is implemented to prescribe burn they will need to consider the fuel dynamics within each overstory stand structure. In open-canopied forests, fine fuel loads are still accumulating (Figure 3.3a) and would result in taller flame lengths (Figure 3.7), but accumulation in the closed-canopied forests is beginning to decrease (Figure 3.3b). Both open- and closed-canopied forests would have seedlings that range in size from <15 cm tall to 137 cm tall and some smaller saplings (0.25 to 2.5 cm dbh) would be present (Figure 3.1). Prescribed fire would still be effective at reducing densities of seedlings in the open-canopied forest with predicted mortality exceeding 90% for seedlings <90 cm tall and 70% for the seedlings 120 cm tall

(Figure 3.9a). Saplings up to 2.5 cm dbh in the open-canopied forests would also be susceptible to fire with approximately 50% predicted to die, but saplings > 2.5 cm would have only 5 and 15% mortality (Figure 3.10a). In contrast, potential for mortality in the closed-canopied forests after 15 years would start to decline. Although predicted mortality would still be >60% for the seedlings <90 cm tall, mortality would be <30% for seedlings >120 cm tall (Figure 3.9b) because the predicted flame lengths would not produce the intensity required to observe mortality (Figure 3.9b). Furthermore, regeneration that had reached dbh would not be susceptible to mortality by a prescribed fire in a closed-canopied forest (Figure 3.10b). The amount of fine fuel in the closed-canopied forest would not be enough to produce the intensities required to kill sapling-sized trees under typical prescribed fire conditions.

Within 20 to 30 years after treatment, the size of ponderosa pine regeneration can range from seedlings <15 cm tall to saplings that are 12.7 cm dbh (Figure 3.1). Fine fuel loads in the closed-canopied forests are stable and predicted flame lengths are about 0.3 m (Figure 3.7). Mortality for the sapling-sized ponderosa pine for these closed-canopied forests under current prescribed fire conditions is unlikely (Figure 3.10b). Seedlings <60 cm tall would still be highly susceptible to prescribed fire, but less than 30% of seedlings 120 cm tall would be susceptible (Figure 3.9b). Open-canopied forests continue to have enough fine fuel to successfully control seedling-sized ponderosa densities (Figure 3.9a), but saplings > 2.5 cm dbh still have predicted mortality less than 20% (Figure 3.10a). As treatments get older (>50 years) and fuels begin to stabilize for the open-canopied stands, seedlings will still be susceptible (Figure 3.10a), but saplings will not (Figure 3.10b).

The flame lengths predicted from the BehavePlus simulations in the above examples does not take into account the total amount of heat produced by smoldering of litter, duff, or coarse woody debris. Smoldering of these fuels can damage tree roots and stem cambium independent of the flaming front (Hartford and Frandsen 1992) and has been shown to increase tree mortality (Ryan and Frandsen 1991, Swezy and Agee 1991, Stephens and Finney 2002). Because CWD is spatially variable in these forests, the effect of the combustion of these fuels would be localized. Furthermore, the total consumption of CWD fuels is unlikely due to the high fuel moisture (Knapp *et al.* 2005, Stephens and Moghaddas 2005) under typical prescribed fire conditions. However, if prescribed burns are implemented when fuel moisture of CWD is not high, the intensity produced by the combustion of these larger fuels could create localized mortality in sapling-sized trees.

The combustion of coarse woody debris during prescribed fires that were sampled by Battaglia (2007) is likely what contributed to the mortality of sapling-sized ponderosa pine trees in that study. Although the flame lengths and associated fuel loads for the areas surveyed were not known before the fires, average bole scorch heights on overstory trees were measured on each plot. These average bole scorch heights were used to estimate flame lengths (Cain 1984, Brown and DeByle 1987). Predicted flame lengths for the plots sampled in Battaglia (2007) ranged from 0.20 to 5.0 m and observed mortality increased with predicted flame length (Figure 3.11). No sapling mortality was observed at flame lengths <0.40 m, which is consistent with the findings of the current study. Observed mortality in saplings >2.5 cm dbh exceeded 50% when flame lengths >

0.80 m tall (Figure 3.11), but for the >5.0 cm dbh saplings, flame lengths of 2.0 m were needed for any appreciable mortality (Figure 3.11).

Achieving flame lengths that will cause mortality in saplings > 2.5 cm may require prescribed burns to be carried out under drier conditions to allow some CWD consumption to contribute to fire intensity. Delayed death of pole-sized (10 to 25 cm dbh) trees from prescribed fire can serve as a source of future inputs of CWD for future prescribed fires. For instance, in the open-canopied forests, there were many snags >10 cm dbh still standing in the open-canopied 20 to 29 year old fires (Table 3.2). These high numbers combined with the substantial increase in CWD loadings around 25 years (Figure 3.4a and 5a) suggests that fire-killed trees were still falling. CWD loadings are also expected to increase in the closed-canopied forests areas burned within the past 10 years due to the high number of snags > 10 cm dbh still standing (Table 3.2). The contribution of fire intensity that CWD can provide should be considered when planning prescribed burns to maintain fuel treatments if mortality of saplings > 2.5 cm dbh is an objective.

An alternative to burning when fuel moistures are low enough to consume CWD is to increase the amount of fine fuels found on the forest floor. For example, current practice in the Black Hill is to whole-tree harvest. This process, which takes the entire tree off site and then piles the residual slash and burns it at the landing, is primarily used to reduce surface fuel loadings, and is very successful in reducing fire intensity. However, if the same area is to be prescribed burned in the future, whole-tree harvesting hinders the ability to effectively reduce ponderosa pine regeneration. Instead, lop and scattering the tops of trees would allow an increase in fine fuel loadings while keeping

CWD fuel loadings at low levels. By leaving the tree tops as activity fuels, burning can still take place when CWD fuel moistures are high, but when the fine fuel moistures are low. Furthermore, having greater fine fuel loadings should increase the probability of mortality in the sapling-size classes.

Conclusions

Based on the relationships between seedling/sapling growth, fuel accumulation, and potential prescribed fire behavior, the use of fire to maintain low regeneration densities in fuel treatments will require some careful planning. Results from this study suggest that current prescribed fire prescriptions and fuel loads are adequate to maintain low densities of ponderosa pine seedlings if burns occur every 10 to 15 years. If managers wait longer than 15 years between burns, regeneration can attain sizes (dbh) that reduce its susceptibility to prescribed fire under typical operational weather conditions. After 20 to 25 years, ponderosa pine will require flame lengths created only by burning under weather conditions that increase the chance for spotting and possibly some overstory tree mortality. These weather conditions should be dry enough to allow CWD to contribute to the fire intensity. Alternatively, managers can augment sites with activity fuels to increase fine fuel loadings to achieve flame lengths that will increase sapling susceptibility to fire, but burn when CWD fuel moisture are high to limit the potential for spotting. While the flames might kill sapling sized trees, mature overstory trees would still maintain some resiliency to fire.

Literature cited:

Agee, JK, Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. For Ecol Manage. 211, 83-96.

Agee, J.K., 2000. Monitoring postfire tree mortality in mixed-conifer forests of Crater Lake, Oregon, USA. Natural Areas Journal 23, 114-120.

Agee, J.K., Huff, M.H., 1987. Fuel succession in a western hemlock/Douglas-fir forest. Can. J. For. Res. 17, 697-704.

Andrews, P.L., Bevins, C.D., Seli, R.C., 2005. BehavePlus fire modeling system, version 3.0: User's Guide. Gen. Tech. Rep. RMRS-GTR-106WWW Revised. Ogden, UT: Department of Agriculture, Forest Service, Rocky Mountain Research Station. 144 p.

Battaglia, M.A., 2007. Maintaining fuel treatments with prescribed fire in ponderosa pine (*Pinus ponderosa*) forests of the Black Hills, South Dakota. Ph.D. Dissertation. Colorado State University, Fort Collins, Colorado.

Bonnet, V.H., Schoettle, A.W., Shepperd, W.D. 2005. Postfire environmental conditions influence spatial pattern of regeneration for *Pinus ponderosa*. Can. J. For. Res. 35: 37-47.

Brown, J.K., Oberhau, R.D., Johnston, C.M., 1982. Handbook for inventorying surface fuels and biomass in the interior West. USDA For. Serv., Gen. Tech. Rep. INT-129, Intermountain Forest and Range Experiment Station, Ogden, UT. 48p.

Brown, J.K., DeByle, N.V., 1987. Fire damage, mortality, and suckering in aspen. Can. J. For. Res. 17, 1100-1109.

Brown, P.M., 2003. Fire, climate, and forest structure in Black Hills ponderosa pine forests. Dissertation. Colorado State University, Ft. Collins, CO. 103 p.

Brown, P. M., Ryan, M. G., Andrews, T. G., 2000. Historical surface fire frequency in ponderosa pine stands in research natural areas, Central Rocky Mountains and Black Hills, USA. Natural Areas Journal. 20 (2), 133-139.

Cain, M.D., 1984. Height of stem-bark char underestimates flame length in prescribed burns. Fire Manage. Notes, 45, 17-21.

Cram, D., Baker, T., Boren, J., 2006. Wildland fire effects in silviculturally treated vs. untreated stands of New Mexico and Arizona. Research Paper RMRS-RP-55. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 28p.

DeBlander. L.T., 2002. Forest resources of the Black Hills National Forest. USDA Forest Service, Rocky Mountain Research Station. 16 p.

Driscoll, D. G., Hamade, G. R., Kenner, S. J., 2000. Summary of precipitation data for the Black Hills area of South Dakota, water years 1931-1998: USGS Open file report 00-329. 163 p

Fulé, P.Z., Covington, W.W., Stoddard, M.T., Bertolette, D., 2006. "Minimal-Impact" restoration treatments have limited effects on forest structure and fuels at Grand Canyon, USA. Rest. Ecol. 14(3): 357-368.

Fulé, P.Z., Covington, W.W., Smith, H.B., Springer, J.D., Heinlein, T.A., Huisinga, K.D., Moore, M.M., 2002. Comparing ecological restoration alternatives: Grand Canyon, Arizona. For. Ecol. Mgmt. 170, 19-41.

Fulé, P.Z., Covington, W.W., 1994. Double sampling increases the efficiency of forest floor inventories for Arizona ponderosa pine forests. Int. J. Wildland Fire. 4 (1), 3-10.

Hall, S.A., Burke, I.C., Hobbs, N.T., 2006. Litter and dead wood dynamics in ponderosa pine forests along a 160-year chronosequence. Ecol. Appl. 16(6), 2344-2355.

Hartford, R.A., Frandsen, W.H., 1992. When it's hot,…., its hot or maybe it's not! (surface flaming may not portend extensive soil heating). Int. J. Wildland Fire 2 (3), 139-144.

Johnson, E.A., Miyanishi, K., 1995. The need for consideration of fire behavior and effects in prescribed burning. Res. Ecology. 3(4), 271-278.

Keyes, C.R., Varner, J.M., 2006. Pitfalls in the silvicultural treatment of canopy fuels. Fire Manage Today, 66, 46-50.

Keyser, T.L., 2007. Changes in forest structure, community composition, and long-term development in ponderosa pine forests following a mixed-severity wildfire in the Black Hills, SD, USA. Ph.D. Dissertation. Colorado State University, Fort Collins, Colorado.

Knapp, E.E., Keeley, J.E., Ballenger, E.A., Brennan, T.J., 2005. Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. For. Ecol. Mgmt, 208, 383-397.

Lundquist, J.E., 2007. The relative influence of diseases and other small-scale disturbances on fuel loading in the Black Hills. Plant Disease. 91(2), 147-152.

Lutes, D., Keane, R., Key, C., Caratti, J., Key, C.H., Benson, N.C., Sutherland, S., Gangi, L., 2006. FIREMON: Fire effects monitoring and inventory protocol. USDA Forest Service General Technical Report, RMRS-GTR-164-CD.

Oliver, W.W., Ryker, R.A., 1990. *Pinus ponderosa* Dougl. ex Laws. Ponderosa pine. *In* Silvics of North America. *Edited by* R.M. Burns and R.H. Honkola. USDA Handb. 654. pp. 173-180.

Parsons, D.J., 1978. Fire and fuel accumulation in a Giant Sequoia forest. J. For. 76, 104-105.

Reinhardt, E., 2003. Using FOFEM 5.0 to estimate tree mortality, fuel consumption, smoke production, and soil heating from wildland fire. In: Proceedings of the Second International Wildland Fire Ecology and Fire Management Congress and Fifth Symposium on Fire and Forest Meteorology, November 16-20, 2003, Orlando, FL. American Meteorological Society. 7p.

Ryan, K.C., Frandsen, W.D., 1991. Basal injury from smoldering fires in mature *Pinus ponderosa* Laws. Int J Wildland Fire 1(2), 107-118.

Ryan, M. G., Joyce, L. A., Andrews, T., Jones, K. 1994. Research natural areas in Colorado, Nebraska, North Dakota, South Dakota, and parts of Wyoming. Gen. Tech. Rep. RM-251. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 57 p.

SAS Institute, 2001. SAS/STAT software, version 9.1 of the SAS system for Windows 2000. SAS Institute Inc., Cary, NC.

Sackett, S.S., Haase, S.M., 1998. Two case histories for using prescribed fire to restore ponderosa pine ecosystems in northern Arizona. In: Pruden, T.L., Brennan, L.A., (eds.). Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings, No. 20. Tall Timbers Research Station, Tallahassee, FL. Pgs. 380-389.

Shepperd, W.D., Edminster, C.B., Mata, S.A., 2006. Long-term seedfall, establishment, survival, and growth of natural and planted ponderosa pine in the Colorado Front Range. West. J. App. For. 21(1): 19-26.

Shepperd, W.D., Battaglia, M.A., 2002. Ecology, silviculture, and management of Black Hills ponderosa pine. USDA Forest Service General Technical Report, RMRS-GTR-97.

Sieg, C.H., McMillin, J.D., Fowler, J.F., Allen, K.K., Negron, J.F., Wadleigh, L.L., Anhold, J.A., Gibson, K.E., 2006. Best predictors for postfire mortality of ponderosa pine trees in the Intermountain West. For. Sci. 52(6), 718-728.

Stephens, S.L., Moghaddas, J.J., 2005. Fuel treatment effects on snags and coarse woody debris in a Sierra Nevada mixed conifer forest. For. Ecol. Manage. 214, 53-64.

Stephens, S.L., Finney, M.A., 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. For. Ecol. Manage. 162, 261-271.

Swezy, M.D., Agee, J.K., 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. Can. J. For. Res. 21, 626-634.

Thomas, T.L., Agee, J.K., 1986. Prescribed fire effects on mixed conifer forest structure at Crater Lake, Oregon. Can. J. For. Res. 16, 1082-1087.

Van Wagner, C.E., 1977, Height of crown scorch in forest fires. Can. J. For. Res. 3, 373-378.

van Wagtendonk, J.W. 1983. Prescribed fire effects on forest understory mortality. In: Seventh Conference on Forest Fire and Meteorology, 25-28 Apr. 1983. Fort Collins, CO. American Meteorological Society, Boston. Pp. 136-138.

Table 3.1: Prescribed fires and wildfires selected for the fuel accumulation time since fire (TSF) study across the Black Hills National Forest. Habitat structural stage (HSS) refers to stand diameter (3 = 12.7 to 22.9 cm dbh; 4 = 22.9 to 35.6 cm dbh; 5 = >35.6 inch dbh) and overstory canopy coverage (A=0 to 40%; B = 40 to 70%; C = >70%) as classified by the Black Hills National Forest (Shepperd and Battaglia 2002). Wildfires are signified by asterisks.

Fire	TSF	HSS	Elevation (m)	Location
Gordon	2	4A	1500 – 1550	43°58'26"N, 103°30'36"W
Gordon	6	4A	1600 – 1650	43°58'54"N, 103°32'00"W
Gordon	6	4A	1700 – 1750	44°00'06"N, 103°33'39"W
Horse Creek	13	4A	1600	44°00'48"N, 103°32'30"W
Boundary*	21	4A	1500 – 1560	44°26'12"N, 104°04'15"W
Ventling Draw*	40	4A	1600 – 1630	43°38'31"N, 103°40'54"W
Deadwood*	47	4A	1450 – 1550	44°24'06"N, 103°43'31"W
McVey*	65	4A	1600 – 1650	44°01'25"N, 103°33'02"W
Spring Creek*	80	4A	1450	44°00'09"N, 103°23'56"W
Medicine	2	4C	1900	43°55'43"N, 103°42'29"W
Medicine	4	4B	1800 – 1950	43°55'56"N, 103°40'40"W
Gordon	7	4C	1700 – 1750	43°59'22"N, 103°33'37"W
Horse Creek	14	3C	1600 – 1650	44°00'58"N, 103°32'29"W
Cicero*	16	4C	1650 – 1700	43°40'34"N, 103°31'33"W
Freeland Basin*	38	4C	1450 – 1500	43°36'32"N, 103°40'54"W
Deadwood*	47	4B	1450 – 1550	44°23'51"N, 103°43'20"W
Big Elk*	57	4C	1450	44°15'08"N, 103°28'48"W
McVey*	67	4C	1600 – 1700	43°59'35"N, 103°32'46"W
Spring Creek*	80	4C	1450 – 1500	43°59'59"N, 103°24'32"W
BHEF*	110	4C	1800	44°10'03"N, 103°38'54"W
Mt. Rushmore NM*	113	5	1520 – 1630	43°52'53"N, 103°27'11"W
Upper Pine Creek NA*	118	5	1660 – 1690	43°52'14"N, 103°30'47"W

Table 3.2: Average (± standard error of the mean) stand structural characteristics of stands measured for the fuel accumulation study after fire study across the Black Hills National Forest. Basal area and density area values are for live trees > 10 cm dbh. Snag densities are for *Pinus ponderosa* only and separated by size class. Snag densities were only measured in summer 2006 stands. N/a are for stands measured in summer 2004 and 2005.

Time Since Fire (years)	Basal Area (m^2/ha)	Trees per hectare	Snags ha^{-1} 0.25 to 10 cm	Snags ha^{-1} > 10 cm	Snags ha^{-1} > 25 cm
		Open-canopied forests			
< 10	15.7 (0.9)	472 (41)	n/a	n/a	n/a
10 to 19	10.7 (2.0)	259 (64)	n/a	n/a	n/a
20 to 29	7.7 (2)[1]	107 (34)[1]	8 (8)	104 (37)	17 (7)
40 to 49	13.2 (4.6)	251 (63)	9 (9)	19 (4)	9 (3)
60 to 69	19.9 (1.6)	532 (98)	n/a	n/a	n/a
80 to 89	15.6 (1.3)	309 (34)	8 (8)	10 (10)	0
		Closed-canopied forests			
< 10	24.8 (2.4)	629 (132)	11 (7)	164 (77)	2 (2)
10 to 19	28.2 (2.0)	1235 (437)	33 (26)[4]	13 (9)[4]	0[4]
30 to 39	21.3 (2.6)	504 (66)	0	49 (17)	12 (5)
40 to 49	22.2 (1.8)	237 (20)	17 (17)	18 (7)	11 (4)
50 to 59	24.2 (2.1)[2]	994 (121)[2]	132 (53)	15 (10)	4(2)
60 to 69	23.7 (1.7)	748 (108)	8 (8)	15 (9)	2 (2)
80 to 89	25.4 (1.4)	907 (123)	91 (82)	16 (9)	0
110 to 119	26.0 (3.1)	588 (134)	n/a	n/a	n/a
		Mature forests			
110 to 119	26.4 (2.8)[3]	885 (335)[3]	254 (137)[5]	29 (15)[5]	12 (5)[5]

[1]*Populus tremuloides*: Basal Area 0.15 m^2/ha and 3 trees ha^{-1}.
[2]*Populus tremuloides*: Basal Area 2.1 m2/ha and 182 trees ha^{-1}.
[3]*Quercus macrocarpa*: Basal Area 0.4 m2/ha and 21 trees ha^{-1}.
[4]Snag densities for Cicero fire only.
[5]Snag densities for Mt. Rushmore National Monument only.

Table 3.3: Average (± standard error of the mean) live seedling (<137 cm tall) and sapling (0.25 to 10 cm dbh) ponderosa pine density (trees per hectare) for the intensively sampled stands measured in summer 2006 for the fuel accumulation study after fire study across the Black Hills National Forest.

Time Since Fire (years)	<15 cm height	15 to 76 cm height	76 to 137 cm height	0.25 to 2.5 cm dbh	2.5 to 5 cm dbh	5 to 7.5 cm dbh	7.5 to 10 cm dbh
Open-canopied forests							
20 to 29	547 (406)	1258 (530)	765 (396)	189 (90)	99 (54)	82 (36)	49 (29)
40 to 49	438 (274)	1449 (246)	328 (55)	231 (231)	103 (103)	33 (25)	50 (50)
80 to 89	0	109 (109)	0	165 (51)	41 (20)	58 (32)	49 (24)
Closed-canopied forests							
< 10	510 (510)	0	0	0	0	0	3 (3)
10 to 19[1]	16347 (5252)	13450 (3491)	328 (275)	99 (49)	41 (33)	58 (29)	41 (23)
30 to 39	2078 (565)	5522 (1678)	383 (177)	8 (8)	25 (18)	17 (11)	25 (18)
40 to 49	984 (544)	4866 (1815)	875 (449)	832 (266)	115 (47)	17 (17)	8 (8)
50 to 59	0	492 (338)	930 (473)	1277 (272)	362 (94)	321 (78)	206 (61)
60 to 69	0	0	0	41 (16)	8 (8)	66 (27)	49 (29)
80 to 89	0	0	55 (55)	99 (54)	198 (89)	173 (69)	74 (54)
Mature forests							
110 to 119[2]	2653 (1239)	3715 (1664)	1013 (395)	1105 (384)	538 (187)	291 (122)	138 (50)

[1]Regeneration densities for Cicero fire only.
[2]Regeneration densities for Mt. Rushmore National Monument only.

Table 3.4: Average (± standard error of the mean) forest floor depth and mass for the fuel accumulation study after fire study across the Black Hills National Forest.

Time Since Fire (years)	Litter depth (cm)	Litter mass (Mg ha^{-1})	Duff depth (cm)	Duff mass (Mg ha^{-1})
Open-canopied forests				
< 10	2.1 (0.1)	12.9 (0.7)	0.68 (0.1)	7.0 (1.2)
10 to 19	2.2 (0.2)	12.9 (1.4)	0.36 (0.1)	3.7 (0.9)
20 to 29	1.4 (0.4)	8.2 (2.6)	1.1 (0.2)	11.1 (1.6)
40 to 49	1.7 (0.2)	10.5 (1.1)	0.41 (0.3)	4.2 (3.1)
60 to 69	2.2 (0.1)	13.3 (0.4)	0.96 (0.1)	9.9 (1.3)
80 to 89	1.7 (0.2)	10.3 (1.2)	0.87 (0.1)	8.9 (1.4)
Closed-canopied forests				
< 10	1.4 (0.3)	8.4 (2.0)	0.80 (0.3)	8.2 (3.0)
10 to 19	2.7 (0.3)	16.3 (1.8)	0.98 (0.5)	10.1 (5.5)
30 to 39	2.5 (0.3)	15.1 (1.9)	0.73 (0.2)	7.5 (2.2)
40 to 49	2.6 (0.4)	15.7 (2.1)	0.29 (0.1)	3.0 (1.7)
50 to 59	2.3 (0.4)	13.9 (2.5)	1.4 (0.3)	14.8 (2.7)
60 to 69	2.2 (0.2)	13.6 (1.3)	1.0 (0.2)	10.4 (1.7)
80 to 89	2.4 (0.2)	14.7 (1.3)	1.0 (0.2)	10.6 (2.0)
110	2.7 (0.1)	16.4 (0.9)	1.6 (0.1)	16.8 (1.3)
Mature forests				
110 to 119	2.6 (0.1)	16.0 (0.7)	1.7 (0.6)	17.1 (6.5)

Table 3.5: Regression coefficients for fuel accumulation of woody debris after prescribed fires and wildfires for each habitat structural stage (HSS).
The model form: Fuel load (Mg ha^{-1}) = $(TSF/\lambda)^{\alpha-1} * (e^{-TSF/\lambda})/\lambda + b*TSF^{c}$.

Fuel type	HSS	λ	α	b	c	r^2	SE
Fine (<7.62 cm) Woody Debris (Mg ha^{-1})	4A	4.068	5.276	3.762	-0.0015	0.93	1.41
	4C	1.798	3.999	3.999	0.0087	0.41	1.09
Sound Coarse (>7.62 cm) Woody Debris (Mg ha^{-1})	4A	n/a	n/a	n/a	n/a	NS	n/a
	4C	n/a	n/a	n/a	n/a	NS	n/a
Rotten Coarse (>7.62 cm) Woody Debris (Mg ha^{-1})	4A	4.680	4.824	-3.65E+22	-32.525	0.91	0.65
	4C	8.414	5.580	-1.171	-518.9	0.45	3.35
Total Woody Debris (Mg ha^{-1})	4A	5.244	4.564	2.340	0.0621	0.88	0.46
	4C	11.923	4.378	2.973	0.0796	0.78	0.16

Figure captions

Figure 3.1: Age distribution of ponderosa pine seedling (a) and sapling (b) sampled in the interior montane region of the Black Hills National Forest. Each shaded box covers the 25th to 75th percentile of the distribution; dotted line is the mean and the solid line represents the median. Whiskers denote the 10th and 90th percentiles. Dots represent the 5th and 95th percentiles.

Figure 3.2: Data and regression relationship between seedling/sapling height and age for Black Hills ponderosa pines less than 50 years old growing in the interior montane region.

Regression: Height (m) =exp(-4.2206 + 1.6727 (ln age)).

Figure 3.3: Temporal dynamics of fine woody (0 to 7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean. Black lines represent the trend predicted by the statistical model (Table 3.5).

Figure 3.4: Temporal dynamics of sound woody (> 7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean.

Figure 3.5: Temporal dynamics of rotten woody (> 7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean. Black lines represent the trend predicted by the statistical model (Table 3.5).

Figure 3.6: Temporal dynamics of total woody (0 to >7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean. Black lines represent the trend predicted by the statistical model (Table 3.5).

Figure 3.7: Temporal dynamics of surface fire flame lengths under typical prescribed fire weather conditions for open-canopied (solid line) forests and closed-canopied (dashed line) forests based on predicted fine fuel loadings and BehavePlus simulations. Dead fuel moistures were 7% for 1-h, 8% for 10-h, and 11% for 100-h. Midflame windspeed was set at 5 km/hr and slope was set at 15%.

Figure 3.8: Probability of mortality of ponderosa pine seedlings as predicted by logistic regressions developed from prescribed burns on the Black Hills National Forest (Battaglia 2007). The seedling model was based on observed flame lengths and associated mortality.

Figure 3.9: Temporal dynamics of the predicted %mortality of ponderosa pine seedlings for (a) open-canopied and (b) closed-canopied forests based on predicted flame lengths associated with fine fuel loading accumulation.

Figure 3.10: Temporal dynamics of the predicted %mortality of ponderosa pine saplings for (a) open-canopied and (b) closed-canopied forests based on predicted flame lengths associated with fine fuel loading accumulation.

Figure 3.11: Observed mortality of ponderosa pine saplings across several prescribed fires with the Black Hills of South Dakota (Battaglia 2007). Flame lengths were predicted from average bole scorch heights on each plot.

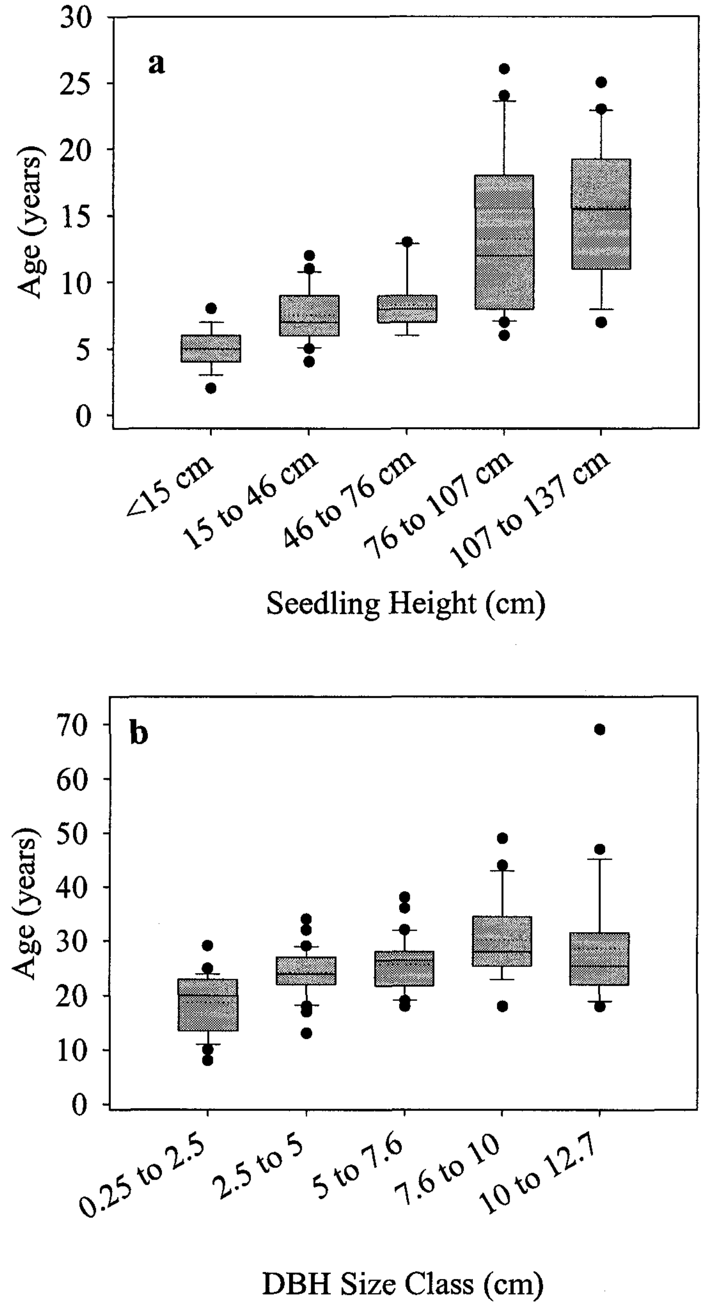

Figure 3.1: Age distribution of ponderosa pine seedling (a) and sapling (b) sampled in the interior montane region of the Black Hills National Forest. Each shaded box covers the 25th to 75th percentile of the distribution; dotted line is the mean and the solid line represents the median. Whiskers denote the 10th and 90th percentiles. Dots represent the 5th and 95th percentiles.

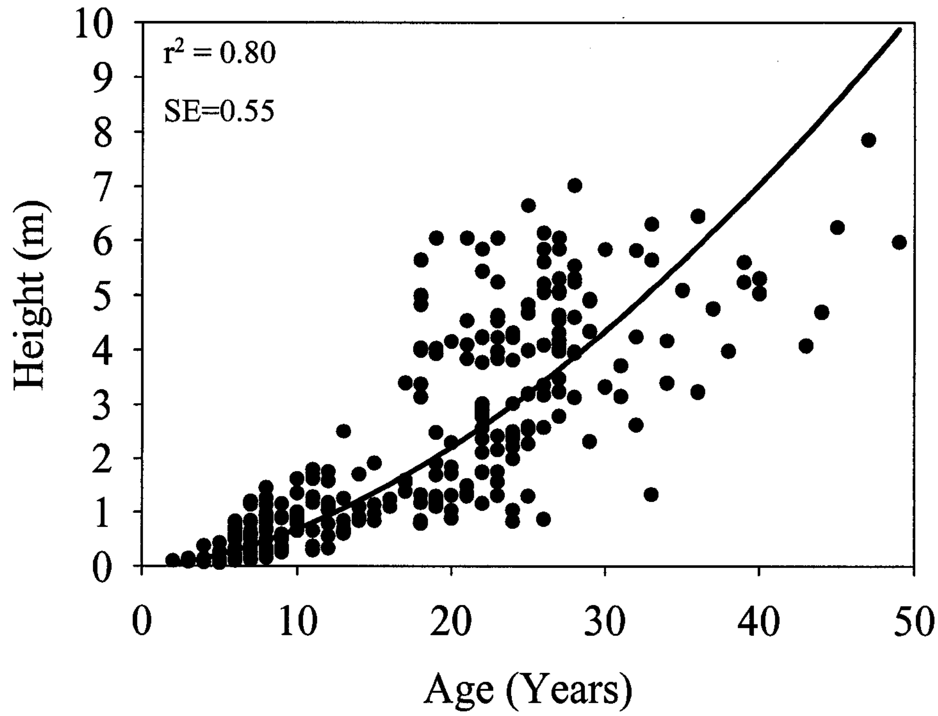

Figure 3.2: Data and regression relationship between seedling/sapling height and age for Black Hills ponderosa pines less than 50 years old growing in the interior montane region.
Regression: Height (m) =exp(-4.2206 + 1.6727 (ln age)).

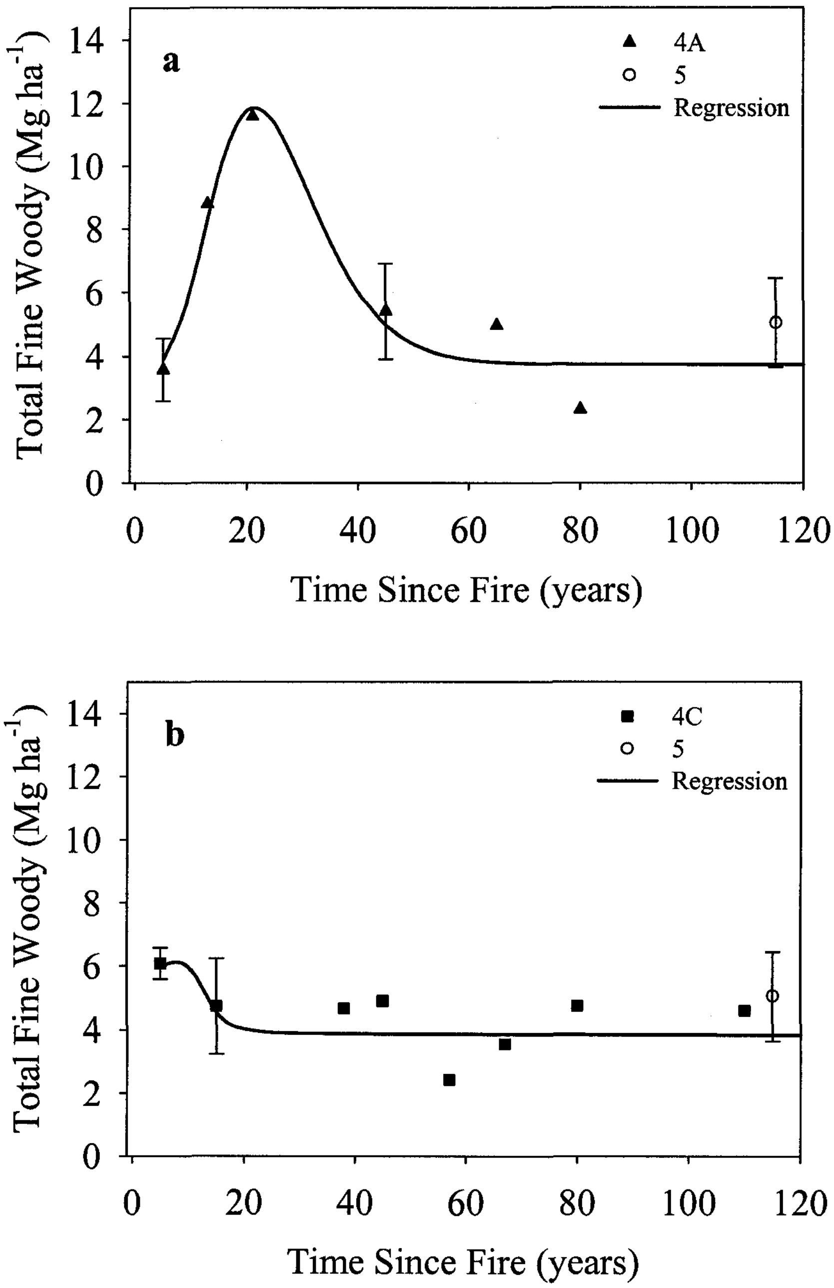

Figure 3.3: Temporal dynamics of fine woody (0 to 7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean. Black lines represent the trend predicted by the statistical model (Table 3.5).

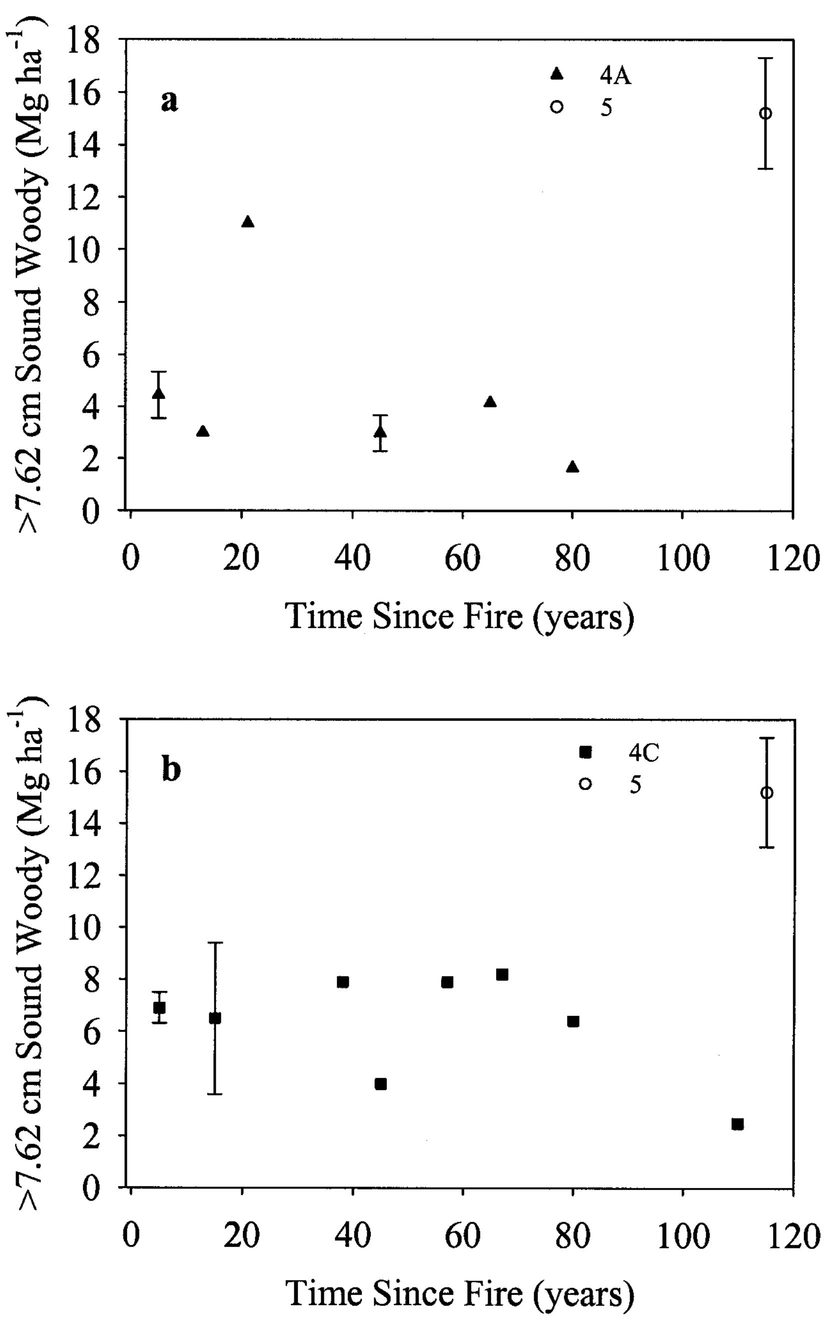

Figure 3.4: Temporal dynamics of sound woody (> 7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean.

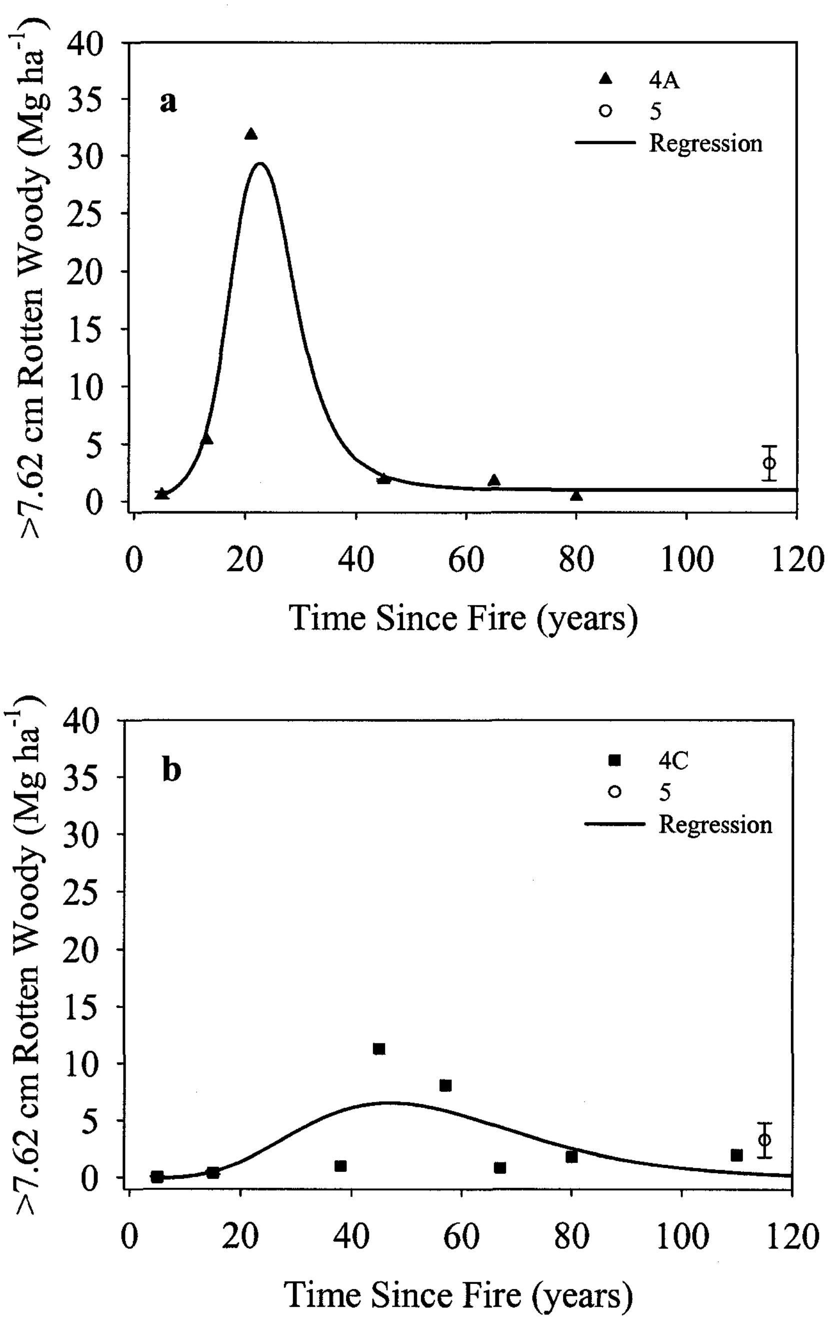

Figure 3.5: Temporal dynamics of rotten woody (> 7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean. Black lines represent the trend predicted by the statistical model (Table 3.5).

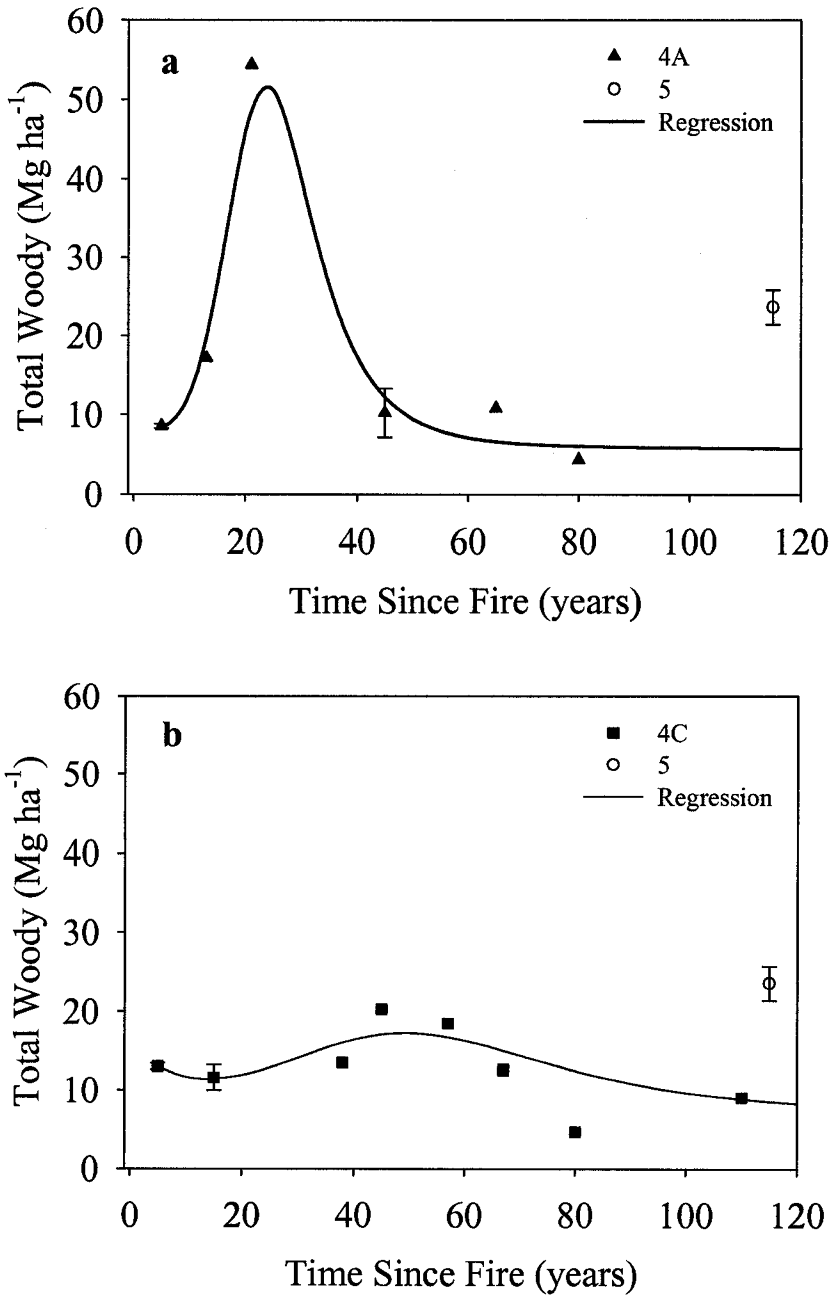

Figure 3.6: Temporal dynamics of total woody (0 to >7.62 cm diameter) fuel loadings for (a) 4A: open-canopied forests and (b) 4C: closed-canopied forests. Mature forests (5) values are presented in both graphs to serve as baselines. Values with error bars are time periods that had more than one stand measured and represent the standard error of the mean. Black lines represent the trend predicted by the statistical model (Table 3.5).

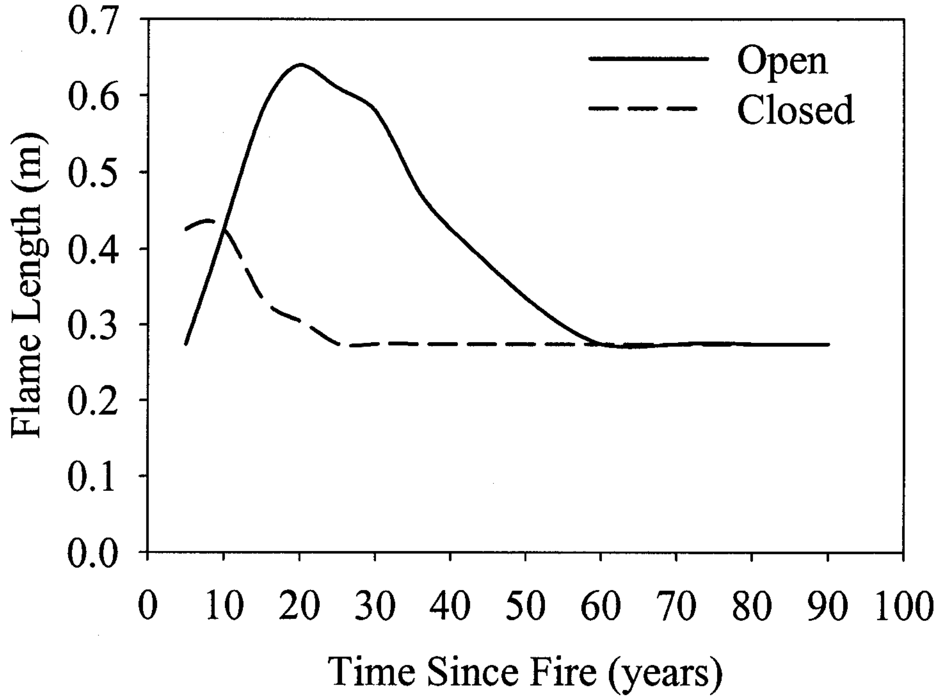

Figure 3.7: Temporal dynamics of surface fire flame lengths under typical prescribed fire weather conditions for open-canopied (solid line) forests and closed-canopied (dashed line) forests based on predicted fine fuel loadings and BehavePlus simulations. Dead fuel moistures were 7% for 1-h, 8% for 10-h, and 11% for 100-h. Midflame windspeed was set at 5 km/hr and slope was set at 15%.

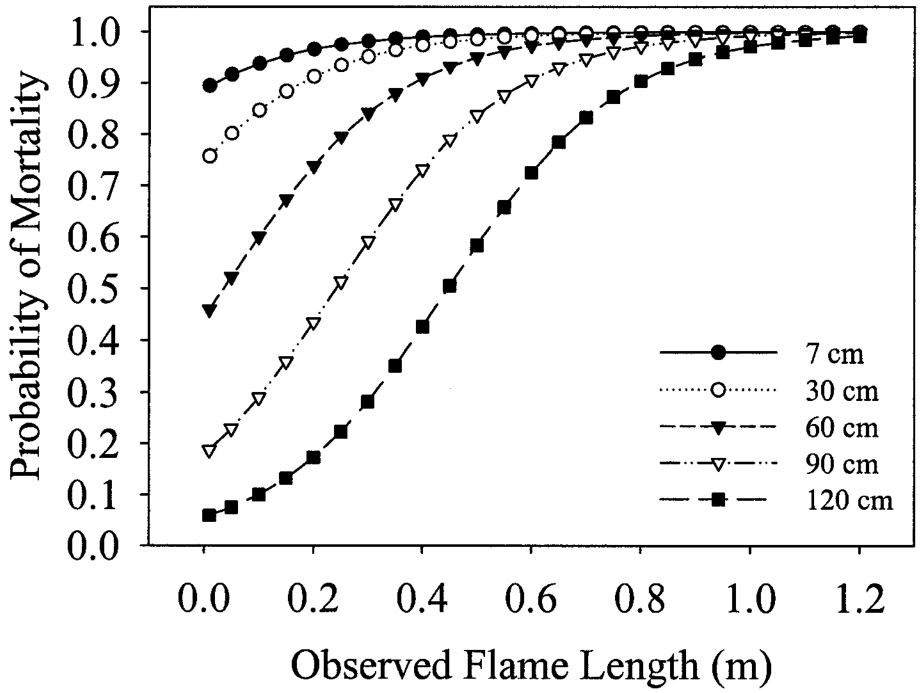

Figure 3.8: Probability of mortality of ponderosa pine seedlings as predicted by logistic regressions developed from prescribed burns on the Black Hills National Forest (Battaglia 2007). The seedling model was based on observed flame lengths and associated observed mortality.

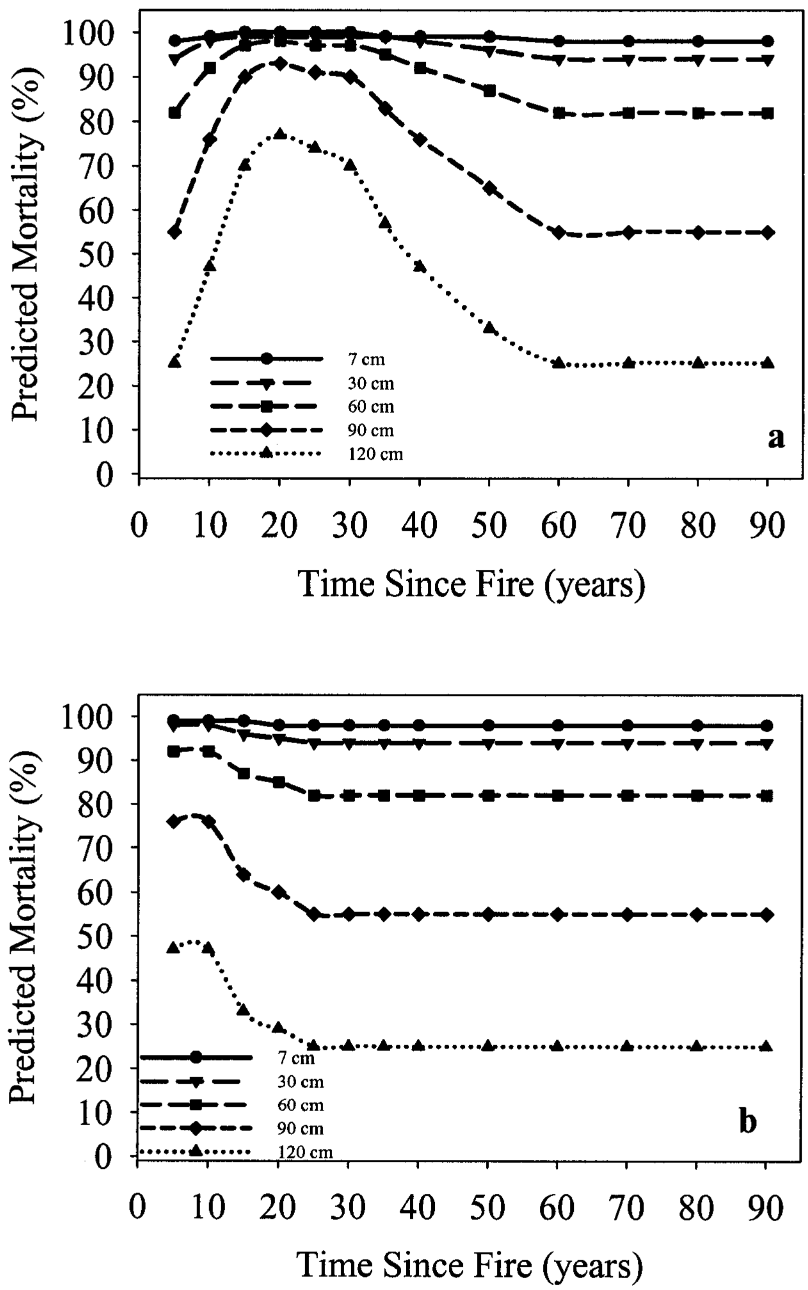

Figure 3.9: Temporal dynamics of the predicted %mortality of ponderosa pine seedlings for (a) open-canopied and (b) closed-canopied forests based on predicted flame lengths associated with fine fuel loading accumulation.

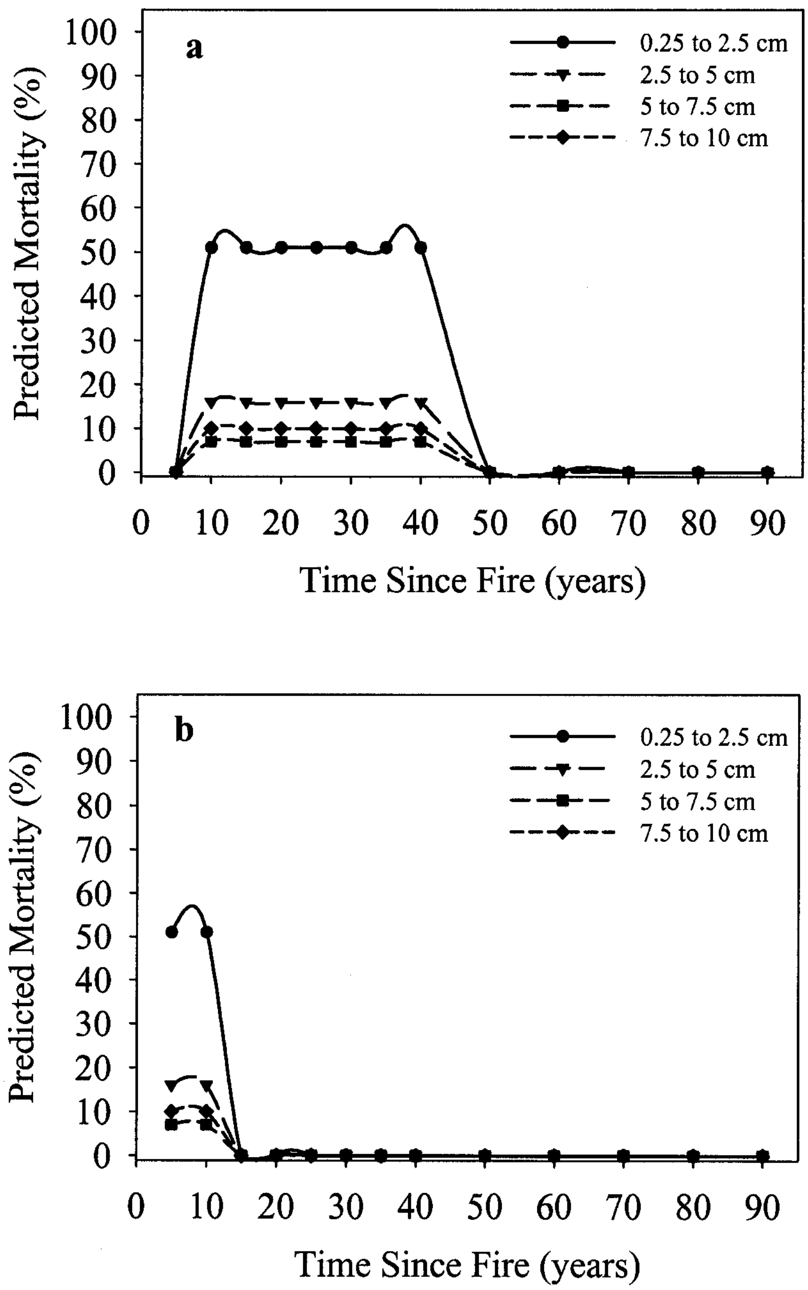

Figure 3.10: Temporal dynamics of the predicted %mortality of ponderosa pine saplings for (a) open-canopied and (b) closed-canopied forests based on predicted flame lengths associated with fine fuel loading accumulation.

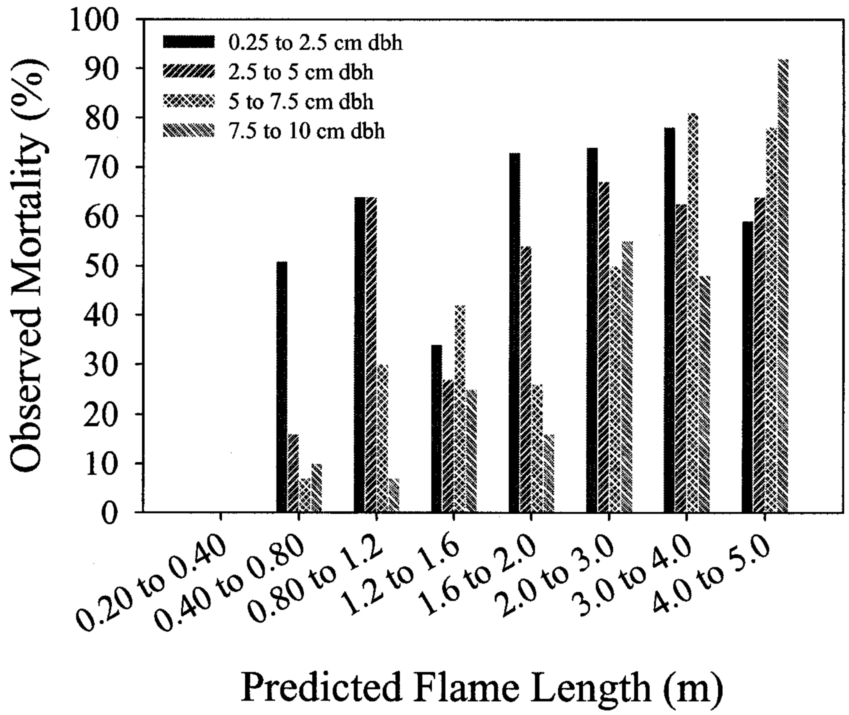

Figure 3.11: Observed mortality of ponderosa pine saplings across several prescribed fires with the Black Hills of South Dakota (Battaglia 2007). Flame lengths were predicted from average bole scorch heights on each plot.

Acknowledgements

This research was supported by Joint Venture Agreement #03-JV-11221616-232 with the USFS Rocky Mountain Research Station, National Fire Plan project 01.RMS.C.4, and by McIntire-Stennis funding to the Colorado State University Department of Forest, Rangeland, and Watershed Stewardship. The authors would like to thank Blaine Cook , Gale Gire, Jay Kurth, Gwen Lipp, and Terry Tompkins of the Black Hills National Forest, Andy Thorstenson and Cody Wienk from Wind Cave National Park, and Peter Brown of the Rocky Mountain Tree Ring Institute for their help in locating prescribed burns and wildfires to sample. The authors also thank Lance Asherin, Jay Aylward, Mike Peterkin, Charity Weaver, Andy Whelan, and Vicky Williams for their assistance in field data collection and entry. We also thank Rudy King for statistical consultations.

CHAPTER 4: POTENTIAL FIRE BEHAVIOR IN EARLY SETTLEMENT BLACK HILLS PONDEROSA PINE FORESTS: IMPLICATIONS FOR FUEL TREATMENT DEVELOPMENT

Abstract

Restoration of pre-EuroAmerican (historical) settlement ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) forest structure is often suggested as a way to reduce the risk of crown fire. Historical ponderosa pine forests of the montane Black Hills were a mosaic of multi-cohort patches of various densities and structure types. Restoration of these forest structures may be in conflict with the fuel treatment objective of lower crown fire risk. In an effort to develop fuel treatments with elements of historical forest structure, we modeled the potential fire behavior of dendrochronological reconstructed historical Black Hills ponderosa pine forest structures under various weather conditions and fire return intervals. Our simulations indicated that historical forest structures of the Black Hills were capable of supporting surface fire, passive crown fire, and active crown fire, but passive crown fire was most likely the dominant fire behavior type. Weather conditions associated with fire behavior determined the size of trees that were likely to be killed and the structural and spatial characteristics of resulting forests. Our results substantiate several lines of evidence which suggest that the historical Black Hills landscape was shaped by a mixed-severity fire regime which created a landscape mosaic of multi-cohort patches of various densities and structure types. Based on our simulations of potential fire behavior in historical Black Hills ponderosa pine forests,

ecological restoration of historical forest structure will not always be compatible with fuel reduction treatment goals. However, there were some historical stand structures that, when burned under extreme weather conditions, had low active crown fire risk, moderate tree mortality, and short flame lengths. These structures could serve as guidelines for fuels reduction treatments if the incorporation of historical forest structures into fuel reduction treatments is a management objective.

Introduction

Recent large-scale wildfires in western ponderosa pine forests has increased efforts to restore pre-EuroAmerican (historical) ponderosa pine *Pinus ponderosa* var. *scopulorum* Dougl. ex Laws.) forest structure with the goal of reducing the risk of crown fire. Such treatments have often been modeled after ponderosa pine forests of the southwest which burned with low-severity surface fires every 3 to 12 years and resulted in open, park-like forests (White 1985, Fulé *et al.* 1997, Covington and Moore 1994, Covington *et al.* 1997, Mast *et al.* 1999, Fulé *et al.* 2004). However, ponderosa pine forests of the Black Hills of South Dakota burned less frequently, every 10 to 35 years (Brown and Sieg 1996, 1999, Brown *et al.* 2000, Brown 2003, Wienk *et al.* 2004). These fires are hypothesized to have consisted of surface, passive crown fire, and active crown fire (Shinnemann and Baker 1997, Ehle and Baker 2003, Brown 2003, Lentile *et al.* 2005, Brown and Cook 2006, Baker *et al.* 2007). The combination of longer fire intervals and mixed fire behavior resulted in a landscape that was a mosaic of multi-cohort patches of various densities and structure types (Brown and Cook 2006). Therefore, restoration of some historical Black Hills forest structures may be in conflict with the fuel treatment objective of lower crown fire risk if these forest structures promoted crown fires. In this paper, we examine the potential fire behavior of historical Black Hills ponderosa pine forests. Specifically, we explore the role of fire frequency intervals, weather, and fire behavior had on the trajectory of forest demography and structure. We then identify historical stand characteristics that could be restored while maintaining the low crown fire risk desired by managers to be incorporated into fuel treatment prescriptions for Black Hills ponderosa pine forests.

Historical descriptions (Ludlow 1875, Newton and Jenney 1880, Graves 1899, Dodge 1965), photographs (Graves 1899, Progulske, 1974, Grafe and Horsted 2002), and dendrochronological reconstructions (McAdams 1995, Brown and Cook 2006) of the Black Hills landscape provide some evidence that the Black Hills was subject to a mixed-severity fire regime. Mixed-severity fire regimes result in landscapes that have a complex matrix of patches on the landscape that include unburned patches, patches of low intensity surface fires, moderate severity patches which burned as surface and passive crown fires killing 1/3 to 2/3 of the vegetation, and high severity patches where most of the vegetation is killed (Agee 2005). Historical descriptions of post-fire areas on the Black Hills landscape ranged from large percentages of old trees injured at the base by fire to large tracts of timber destroyed (Ludlow 1875, Newton and Jenney 1880, Graves 1899, Dodge 1965). Damage by fires included a full range of severity; total mortality, mortality limited to the small second growth, or little injury (Graves 1899). Graves (1899) often commented that forest continuity was broken by fire and the blanks were filled with young growth or that forests were interrupted by open parks and burnt tracks. Photographs in the 1890's USGS survey report show multi-storied forests, forests with heavy surface fuels, dense sapling regeneration, and second growth even-aged forests

(Graves 1899). Dendrochronological reconstructions suggest a diverse landscape mosaic of non-forested patches and open stands of few large trees to quite dense stands with many similar-sized and –aged trees (Brown and Cook 2006). Across the landscape, a fine-scale mosaic of dense and open forests within 100 m of each other (Brown and Cook 2006) was imbedded within a coarse-scale mosaic of forests varying in age, density, and structure. This mosaic of forest structures and openings is consistent with a landscape that developed under a mixed-fire severity fire regime.

The potential for dense, multi-cohort forest structures to have developed in historical Black Hills ponderosa forests is in part due to the prolific ponderosa pine regeneration under a range of climatic conditions and post-fire environments. The favorable average growing season climate in the Black Hills (Shepperd and Battaglia 2002) often allows up to several thousand ponderosa pine seedlings per hectare to establish (Graves 1899, Lentile 2004, Bonnet *et al.* 2005, Battaglia 2007, Keyser 2007). Pulses of copious tree recruitment during pluvial periods also contributed to dense, multi-cohort forest structure (Brown 2006). Even under droughty conditions, regeneration rates within the 2000 Jasper fire perimeter ranged from 376 to 506 seedlings ha^{-1} in unburned areas (Lentile 2004, Bonnet *et al.* 2005) and 660 to 1125 seedlings ha^{-1} in burned areas near unburned edges (Lentile 2004, Bonnet *et al.* 2005). The successful and continuous recruitment of regeneration under various climatic conditions in conjunction with the 10-35 year fire return interval allowed time for ponderosa pine to reach sizes and densities that may have contributed to passive and/or active crown fire behavior.

Fuel reduction treatments are designed to decrease potential flame lengths, reduce potential crown fire initiation (passive crown fire) and spread (active crown fire), reduce landscape level fire severity, and increase the safety of suppression personnel during a wildfire (Finney 2001, Agee and Skinner 2005). Many studies have demonstrated that fuel reduction treatments are effective at reducing fire behavior and severity during a wildfire (Pollet and Omi 2002, Wilmes *et al.* 2002, Finney *et al.* 2005, Raymond and Peterson 2005, Cram *et al.* 2006, Moghaddas 2006), although small acreage (less than 100 ha) or old (>10 to 15 years) fuel treatments may be ineffective under very extreme weather conditions (Finney *et al.* 2003). The goals of fuels reduction treatments are often achieved by reducing surface fuel loadings, increasing canopy base height, and reducing canopy bulk density (Agee and Skinner 2005, Cram *et al.* 2006). In some ponderosa pine ecosystems, the structural outcomes of fire hazard reduction and ecological restoration treatments may correspond, while in others they may be incompatible (Veblen 2003, Schoennagel *et al.* 2004, Platt *et al.* 2006).

Restoration of historical forest structure in southwest ponderosa pine ecosystems has been demonstrated to be compatible with achieving fuel reduction treatment goals. In this low-intensity, surface fire dominated ecosystem, ecological restoration often focuses on recreating historical forest structure by conserving old trees, thinning dense post-settlement trees, and reintroducing surface fire (Covington *et al.* 1997, Moore *et al.* 1999, Fulé *et al.* 2001, Fulé *et al.* 2002, Fulé *et al.* 2006). Several studies in these ecosystems have demonstrated that historical forest structures do result in a reduction in passive and active crown fire potential (Fulé *et al.* 2001, Fulé *et al.* 2002, Fulé *et al.* 2004), flame lengths (Fulé *et al.* 2001, Fulé *et al.* 2002), fire intensity (Fulé *et al.* 2001, Fulé *et al.* 2002), and tree mortality (Fulé *et al.* 2001, Fulé *et al.* 2002).

The appropriateness of applying the southwestern restoration model to ponderosa pine forests of other geographical regions has been questioned (Shinneman and Baker 1997, Baker and Ehle 2001, Shepperd and Battaglia 2002, Brown 2003, Veblen 2003, Baker *et al.* 2006) because of the differences in fire frequency, fire behavior, and historical forest structure between the regions. Restoring a full suite of historical forest structure in Black Hills ponderosa pine may not result in reduced crown fire hazard since crown fires may have been a part of the historic fire regime. Instead, we propose that certain elements of historical forest structure that are associated with low crown fire risk could be implemented into fuel treatments prescriptions to reduce overall fire risk at landscape scales. In this paper we model potential fire behavior of historical Black Hills ponderosa pine forest structures under various weather conditions and fire return intervals. Specifically, we use this information to (i) explore how fire frequency and subsequent fire behavior affect stand development, and (ii) identify historical stand characteristics that were associated with low crown fire risk.

Methods

Study area

The Black Hills is an isolated forested geologic uplift located in southwest South Dakota and northeast Wyoming that extends 200 km from north to south and 100 km from east to west (Shepperd and Battaglia 2002). Elevations in the Black Hills range from 1,000 m to 2,207 m and are approximately 300 to 1,200 m above the surrounding Great Plains. The increased elevation results in an orographically induced microclimate that increases precipitation. Precipitation and temperature patterns differ along elevational and latitudinal gradients. In general, northern locations and higher elevations receive more precipitation and are cooler than the southern and lower elevation areas (Driscoll *et al.* 2000). Annual average precipitation ranges from 41 cm in the south to 74 cm in the north. Most of the precipitation occurs from April to August, with May and June receiving 33% of annual precipitation (Driscoll *et al.* 2000). Annual average temperature ranges from 2.9° to 9° C. The frequent rain showers during the early growing season in combination with the warm temperatures are conducive to prolific natural ponderosa pine regeneration (Shepperd and Battaglia 2002). Ponderosa pine dominates 85% of the forested land base (DeBlander 2002) and is found at all elevations, soil types, and aspects. It can be found mixed with white spruce (*Picea glauca* [Moench] Voss) and aspen (*Populus tremuloides* Michx.) in the moister, higher elevations.

Historical forest structure data: Trees > 10 cm dbh in 1900

The reconstructed pre-EuroAmerican settlement forest structure data was obtained from Brown and Cook (2006). A detailed description of the field collection and laboratory analysis is reported in Brown and Cook (2006). The reconstructed data was collected from 112 randomly chosen plots across the range of precipitation gradients in current Black Hills forest structure. At each plot, the n-tree distance sampling method (Jonsson *et al.* 1992, Lessard *et al.* 2002) was used to collect data on the nearest 30 pre-settlement trees to plot center. Remnant and living trees were classified as pre-settlement trees if they were not "blackjacks" or were at least 30 cm dbh (diameter breast height). Blackjack trees are typically young trees with dark bark. In the Black Hills, it has been observed to take about 75 to 100 years for ponderosa pine bark to change from a dark to buff or orange color (Brown and Cook 2006). Living tree diameters were measured at 10 cm (diameter at sample height: dsh) and at dbh. On remnant trees, diameters were only measured at the 10 cm height and were supplemented with decay status (presence of bark, heartwood, and sapwood). Historical dbh ca. 1900 for individual living and remnant trees was determined using a regression equation based on the relationship of dbh to dsh in living trees (Brown and Cook 2006; dbh = dsh * 0.8645 (n=567; r^2=0.95). Historical dbh ca. 1900 for stumps measured in the plots were calculated based on the state of decay and assumed time of harvest. Based on cross-dated stump data (Brown 2003), it was assumed that stumps with only heartwood present were living around 1900. For these stumps, a conversion was applied based on the ratio of heartwood diameter to total tree dbh in living trees (Brown and Cook 2006). For stumps missing bark, it was assumed that the trees were harvested within the past few decades. Dbh was estimated as the sapwood diameter + (sapwood diameter * 0.1 for estimated bark width). The dbh in 1900 was then calculated using the conversion regression.

Tree density by diameter size class (5 cm intervals) for each plot ca. 1900 was then calculated. At each plot, the radius was calculated as the distance from plot center to center of the farthest tree measured. Plot area was determined as a circular plot of the calculated radius with a bias correction (Moore 1954, Lessard *et al.* 2002, Lynch and Wittwer 2003). Slope corrections were applied to plots with over 10% slope (~60% of all plots) to account for variation in tree basal areas with increasing slopes. The maximum plot size was 0.5 ha, but most plots were about 0.25 ha (Brown and Cook 2006).

Historical forest structure data: Trees < 10 cm in 1900

Precise measurement of sapling-sized trees (0 to 10 cm dbh) ca. 1900 was difficult to reconstruct since any mortality of those trees would likely not be in the remnant record (i.e. decomposition) or could have been misidentified as blackjacks and not measured. Seedling/sapling density ca. 1900 was estimated based on current seedling/sapling density-stand basal area relationships developed from Rocky Mountain FSVEG 2005 data for ponderosa pine forests of the Black Hills. FSVEG provides stand inventory information which includes stand density by diameter class, topographic features, stand productivity, and other useful data for each stand in a region. For each stand basal area, the median seedling density (<2.54 cm dbh) was calculated. Only stands (n=5,570) that were ground surveyed and had more than 5 plots representing a basal area were used to determine the relationship.

Using the stand basal area to median seedling/sapling density relationship seedling/sapling density ca. 1900 was estimated for three fire return interval scenarios (10-yr, 20-yr, and 30-yr). First the basal area of each stand in 1890 (10-yr fire interval), 1880 (20-yr fire interval), and 1870 (30-yr fire interval) was estimated. To do this, forest growth was simulated for each stand reconstruction 30 years into the future (1900 to 1930) using the Forest Vegetation Simulator (FVS), Central Rockies variant (Van Dyck 2000). FVS is an individual-tree, distance-independent, empirically based model of forest growth and yield. For each stand, a regression analysis was performed to determine the relationship between basal area and year. With this relationship the basal area in 1890, 1880, and 1870 was estimated. Based on the predicted basal area for these years, the seedling/sapling density-stand basal area relationship was used to estimate the seedling regeneration for 1890, 1880, and 1870. Once the seedling regeneration density was determined for those years, the amount of seedling/sapling regeneration that would have been present in the historical 1900 forest understory based on the fire return interval scenarios was calculated. For example, in the 10-yr fire return interval scenario, the predicted stand basal area in 1890 was used to predict the seedling density that would have accumulated from 1890 to 1900 and be present in 1900. For a 30-yr fire return interval, regeneration density for 1870, 1880, and 1890 was calculated and summed to predict the seedling density that would have been present in 1900.

Once seedling density for each stand and fire return interval scenario was determined, seedling/sapling dbh and height distribution was estimated. FVS was used to grow four regeneration density scenarios (202.5, 243, 283, and 324 seedlings per hectare) with the average site index (19 m) of the historical reconstructed stands. Each scenario was simulated for 10, 20, and 30 years and the distribution of dbh and height for each regeneration density and time interval was examined. Regeneration density was not a factor in determining the proportion of seedling/saplings within a dbh or height class within a time interval; however, time interval was an important factor. The proportions for each time interval were assigned to the seedling density values within each fire return interval scenario to give an estimate of seedling/sapling dbh and height size ca. 1900.

FVS model runs: Stand development, fire behavior, and fire effects

The FVS Central Rockies variant was used to simulate stand development (Van Dyck 2000) for 100 years for the different fire return interval (10, 20, and 30 years) and weather (80th, 90th, and 97th) scenarios.

The fire and fuels extension (FFE) to FVS was used to calculate crown fuel loads. FFE-FVS calculates canopy bulk density (CBD) and canopy base height (CBH), parameters important for modeling crown fire behavior. CBD of a stand is calculated as the maximum 4.5 m deep running mean of CBD for layers 0.3 m thick. This method is thought to be more realistic than assuming a uniform vertical distribution of canopy fuel

(Scott and Reinhardt 2001). CBH is calculated based on the lowest height above which a minimum of 0.011 kg m^{-3} of available canopy fuel is present (Scott and Reinhardt 2001).

FFE-FVS selects fire behavior fuel models (Albini 1976, Anderson 1982) using a logic-based decision process which is based on the accumulation of woody fuels over time (Reinhardt and Crookston 2003). Instead of allowing FFE-FVS to choose the fuel model in 1900, initial fuel models were selected based on time since fire and stand development. We assumed a fuel model 2 (Anderson 1982) for the 10-year fire return interval scenario because fine herbaceous fuels would be the primary fire carrier due to the length of time it would take in the Black Hills to accumulate enough needle biomass to change the primary fire carrier (Battaglia 2007). For the 20-year fire return interval we assumed a fuel model 5 (Anderson 1982) because the ponderosa pine regeneration density in the understory would be similar to the fire behavior associated with shrub fuel complexes (Battaglia 2007). For the 30-year fire return interval, we assumed a fuel model 9 (Anderson 1982) because enough of the long needles from the ponderosa pine overstory and saplings would have accumulated to serve as the primary fire carrier (Battaglia 2007). Fire behavior fuel models after the initial decade was chosen by FFE-FVS because we could not anticipate the amount of fuel accumulation after fire-induced tree mortality.

For each scenario, seedling establishment was added at the beginning of each decade based on the seedling/sapling density-stand basal area relationship discussed earlier. SDImax, a FVS keyword that specifies the maximum stand density index important for determining stand mortality, was set at 1112 [metric units]. Density-related mortality was set to begin at 55% SDI and maximum density was set at 80% SDI. In addition, a stagnation effect was implemented when SDI > 70% of maximum SDI.

FireFamily Plus, a statistical software program that imports weather data and allows analysis of fire-related variables (Bradshaw and Brittain 1999), was used to construct three fire weather scenarios. Weather data (1966 to 2004) was obtained from the Custer, SD weather station to determine the 80th (moderate), 90th (high), and 97th (extreme) percentile weather conditions for July 1 through September 30 (Table 4.1), the historical fire season (Brown and Sieg 1996, 1999, Brown 2003). Wind speeds provided by the weather databases are based on a single daily windspeed observation and are often a 10-minute average. Since wind speeds much higher can occur at the station during other times of the day, Scott (2003) suggests that a value that represents the near maximum 1-minute average windspeed or a value based on expert knowledge should be used. A wind conversion chart developed by the National Oceanic and Atmospheric Administration (NOAA) was used to convert the 10-minute average windspeed to the probable maximum 1-minute windspeed.

Wildfire and its effects in each stand were simulated for each fire return interval and weather scenario (80th, 90th, and 97th) using the SIMFIRE keyword (Reinhardt and Crookston 2003). Wildfires were assumed to only burn 75% of the stand area because historical fires probably burned patchy due to local environmental and topographic variability. For the 10-yr fire return interval, a wildfire was simulated every decade at the beginning of each simulation cycle. For the 20-yr fire return interval, a wildfire was simulated every 20 years. For the 30-yr fire return interval, a wildfire was simulated every 30 years. FFE-FVS uses equations from FOFEM to predict tree mortality

(Reinhardt *et al.* 1997). FFE-FVS predicted flame lengths and scorch heights based on Rothermel's fire behavior model (Rothermel 1972, Andrews 1986). Fire behavior during the simulated wildfires was predicted using equations developed by Rothermel (1972), Van Wagner (1977), and Scott and Reinhardt (2001). Potential fire behavior (keyword: POTFIRE) was also assessed for each stand within each of the scenarios by decade.

Potential fire behavior, crowning index, and flame lengths were also predicted for each stand by decade under 97^{th} percentile weather conditions. The crowning index is the 6.1 m windspeed required to sustain an active crown fire (Scott and Reinhardt 2001).

Statistical analysis

Potential fire behavior for the 100 year simulation was assessed for each fire return interval and weather scenario. For each decade, FFE-FVS provides an estimate for the potential fire behavior type (surface, passive crown fire, or active crown fire) that occurred for a simulated fire or potential fire. Proc FREQ (SAS Institute 2001) was used to determine the percentage of fire type for each decade within each scenario. The average flame length, average scorch height, and average CBH was calculated for each scenario by fire behavior type for the last 60 years of the simulation.

Mortality associated with fire behavior type was assessed for the last 60 years of the simulation. For each stand in a scenario, the average mortality of each dbh size class by fire behavior type was calculated. Mortality data was arcsin square-root transformed for statistical analysis. We used t-tests for normally distributed data or Wilcoxon rank sum tests for nonnormal data to determine if there were significant differences (α=0.05) in mortality of each dbh size class between fire behavior types for each scenario.

To assess stand structure sustainability, we calculated the relative density and the quadratic mean diameter of trees > 13 cm dbh for each scenario. We chose to limit our analysis to trees > 13 cm dbh because the accuracy of the initial dendrochronological reconstruction data for trees < 13 cm dbh was unknown. The difference in relative density and quadratic mean diameter of trees > 13 cm dbh at the beginning of the simulation and the average relative density and quadratic mean diameter of trees > 13 cm dbh for the last 60 years was calculated for each stand in each scenario. Proc FREQ (SAS Institute 2001) was used to examine the distribution of average relative density and quadratic mean diameter of trees > 13 cm dbh for the last 60 years of each scenario.

Active crown fire risk under 97th percentile weather conditions was assessed for each fire return interval and weather scenario for the last 60 years of each simulation. The crowning index was averaged for each stand over the 60-year period within a scenario. Stands with a crowning index > 80.5 km h^{-1} were classified as low risk, moderate risk from 40.2 to 80.5 km h^{-1}, and high risk = < 40.2 km h^{-1} (Fiedler *et al.* 2004). Proc FREQ (SAS Institute 2001) was used to determine the percentage of each risk category for each scenario. Averages for potential flame length, percentage basal area mortality, and total volume mortality for fires burning under 97th percentile weather conditions was also calculated for each scenario.

We used R (version 2.4.0) to perform a non-parametric classification and regression tree (CART) technique (Jain and Graham 2004, Lentile *et al.* 2006a) to explore relationships between historical stand structure attributes and potential for active crown fire behavior under 97th percentile weather conditions. CART does not require normally distributed data and is useful for non-linear response data (Breiman *et al.* 1984).

CART detects interactions, identifies thresholds, and maximizes homogeneity within a classification. We used CART to produce a decision tree where each branch is split into mutually exclusive subsets based on a decision rule to generate predictions of low, moderate, or high risk for active crown fire based on the dependent variables. Forest structure characteristics at the top of the classification tree indicate a stronger relationship to active crown fire risk type than characteristics at the bottom of the tree.

Results

Potential fire behavior

Potential fire behavior was simulated for a 100 year period, but we limited our estimates of fire behavior and effects after the FVS model equilibrated (the last 60 years of the simulations). The potential fire behavior of historical ponderosa pine forests consisted of surface and passive crown fire (Figure 4.1). The predominance of fire behavior type was dictated by fire return interval and weather conditions.

Fire behavior in stands that developed under a 10-year fire return interval was responsive to weather conditions. Under 80th percentile weather conditions, surface fire occurred in 40% of stands and passive crown fire occurred in 60% of stands (Figure 4.1a). Average flame lengths for surface fires were 1.05 m and 1.14 m for passive crown fires (Table 4.2). Passive crown fire increased to 76% of stands under 90th percentile weather conditions (Figure 4.1b) and to 92% under 97th percentile weather conditions (Figure 4.1c). Under 90th percentile weather conditions, average flame lengths for surface fires were 1.59 m and 1.96 m for passive crown fires (Table 4.2). Average flame lengths increased to 2.33 m for surface fires and 3.20 m for passive crown fires under 97th percentile weather conditions (Table 4.2).

Fire behavior in stands that developed under a 20-year fire return interval was dominated by passive crown fire for all three weather scenarios (Figure 4.1d, e, and f). Under 80th percentile weather conditions, 85% of stands experienced passive crown fire (Figure 4.1d). Average flame lengths for surface fires were 0.77 m and 1.02 m for passive crown fires under 80th percentile weather conditions (Table 4.2). Passive crown fire occurrence increased to 98% under 90th percentile weather conditions (Figure 4.1e) and 100% under 97th percentile weather conditions (Figure 4.1f). Average flame lengths were 2.04 m for 90th percentile weather conditions and increased to 5.06 m under 97th percentile weather conditions (Table 4.2).

Stands that developed under a 30-year fire return interval experienced surface fire and passive crown fire (Figure 4.1g, h, and i). Surface fire occurred 30% and passive crown fire occurred 70% (Figure 4.1g) under 80th percentile weather conditions. Average flame lengths were 0.68 m during a surface fire and 1.03 m during a passive crown fire (Table 4.2). Under more extreme weather conditions, passive crown fire was common (Figure 4.1h and i). Under 90th percentile weather conditions, passive crown fire occurred over 91% of the time (Figure 4.1h). Average flame lengths during a surface fire were 0.69 m and 2.06 m during a passive crown fire (Table 4.2). Passive crown fire occurred 99% of the time under 97th percentile weather conditions (Figure 4.1i) with average flame lengths of 6.3 m (Table 4.2).

The risk for active crown fire in historical Black Hills ponderosa pine forests varied among the three fire return interval scenarios (Figure 4.2). Stands that developed under a 10-year fire return interval were the least likely to sustain an active crown (Figure 4.2a, b, and c). However, the risk for active crown fire in stands that developed under a

20-year or 30-year fire return interval varied and was dependent upon weather conditions (Figure 4.2 d-i). The majority of stands that developed under 80th percentile weather conditions for the 20- or 30-year fire return interval had low active crown fire risk (Figure 4.2d and g), although, 20% and 40% of stands did have moderate active crown fire risk, respectively. Under 90th percentile weather conditions, low and moderate active crown fire risk was similar (Figure 4.2e and h). At 97th percentile weather conditions, the majority of stands were at moderate risk for active crown fire (Figure 4.2f and i), although 14% of stands in the 30-year fire return intervals were rated at high risk for active crown fire (Figure 4.2i).

Mortality associated with fire behavior:

Weather, fire behavior, and tree size all influenced fire-induced tree mortality. Fires caused high mortality of trees < 10 cm dbh, regardless of fire behavior or weather scenario (Figure 4.3). In trees > 10 cm dbh, mortality decreased with tree size, but the magnitude of mortality was influenced by fire behavior and weather conditions. Mortality was generally higher in stands experiencing passive crown fire than stands experiencing surface crown fire across all diameter classes (Figure 4.3).

Mortality in stands experiencing surface fire decreased with tree size (Figure 4.3). Surface fires often killed 72 to 76% of all trees < 10 cm dbh, regardless of fire return interval or weather scenario. In general, surface fires killed 27 to 36% of trees 10 to 20 cm dbh, 13% to 17% of trees 20 to 30 cm dbh, 6 to 10 % of trees 30 to 40 cm dbh, and < 4% of trees > 40 cm dbh (Figure 4.3 a and 4.3b). Surface fire was severe to trees in stands with a 10-year fire return interval under 97th percentile weather conditions. In these stands, 91% of trees 10 to 20 cm dbh, 56% of trees 20 to 30 cm dbh, and 8 to 13% of trees >30 cm dbh died (Figure 4.3c).

Mortality in stands experiencing passive crown fire also decreased with tree size, but the mortality size threshold was influenced by fire return interval and weather conditions. Similar mortality (~75%) in trees < 10 cm dbh was observed for each weather condition and fire return interval (Figure 4.3). Under 80th percentile weather conditions, mortality of trees 10 to 20 cm dbh ranged between 39 to 45% but increased to 70 to 76% under 90th and 97th percentile weather conditions (Figure 4.3). Three patterns emerged in tree sizes $\geq$ 20 cm and < 50 cm dbh: 1) low mortality (6 to 22%) under 80th percentile weather conditions (Figure 4.3a, d, and g), 2) lower mortality in stands with 10-year fire intervals versus 20- or 30-year intervals under 90th percentile weather conditions (Figure 4.3b versus Figure 4.3e and h), and 3) high mortality (68 to 75%) for trees <50 cm dbh under 97th percentile weather conditions, regardless of fire return interval (Figure 4.3c, f, and i). In trees >50 cm dbh, mortality was 5 to 9% for all stands under 80th percentile weather conditions (Figure 4.3a, d, and g) and stands with a 10-year fire return interval under 90th percentile weather (Figure 4.3b). Mortality in trees >50 cm dbh increased to 20 to 23% for stands with 20- or 30-year fire return intervals under 90th percentile weather conditions (Figure 4.3f and i). Under 97th percentile weather conditions, mortality of trees > 50cm dbh increased to 42 to 64% in the 10-year fire return interval stands (Figure 4.3c) and 66 to 71% in the 20- and 30-year fire return interval stands (Figure 4.3f and i).

Impacts on stand structure

The frequency distribution of relative density of trees >13cm ($RD_{>13cm}$) and quadratic mean diameter of trees >13cm ($QMD_{>13cm}$) over 100 years of simulation revealed different patterns for each scenario (Figure 4.4 and 5; Table 4.3). Four patterns emerged (Table 4.3): 1) a decrease in $RD_{>13cm}$ and an increase in $QMD_{>13cm}$; 2) a decrease in $RD_{>13cm}$ and a decrease in $QMD_{>13cm}$; 3) no change in $RD_{>13cm}$ and a decrease in $QMD_{>13cm}$; and 4) an increase in $RD_{>13cm}$ and a decrease in $QMD_{>13cm}$.

In stands with a 10-year fire return interval, the majority of stands developing under 80th and 90th percentile weather conditions had $RD_{>13cm}$ between 0 to 20% (Figure 4.4a and b) and $QMD_{>13cm}$ of 55 to 75 cm (Figure 4.5a and b). Average $RD_{>13cm}$ decreased to 9.7% and average $QMD_{>13cm}$ increased to 59.5 cm for the 80th percentile weather condition (Table 4.3). Average $RD_{>13cm}$ decreased to 7.9% and average $QMD_{>13cm}$ increased to 61.2 cm for the 90th percentile weather condition (Table 4.3). In contrast, the majority of stands under 97th percentile weather conditions had $RD_{>13cm}$ 0 to 10% (Figure 4.4c). $QMD_{>13cm}$ showed greater variation with some stands above 55 cm dbh, but the majority of stands had $QMD_{>13cm}$ between 15 and 45 cm dbh (Figure 4.5c). Average $RD_{>13cm}$ decreased to 2.6% and average $QMD_{>13cm}$ decreased to 35.1 cm (Table 4.3).

For stands with a 20-year fire return interval, the distribution of $RD_{>13cm}$ and $QMD_{>13cm}$ shifted to the lower values as weather became more extreme (Figure 4.4 d, e, and f; Figure 4.5 d, e, and f). Under 80th percentile weather conditions, average $RD_{>13cm}$ was similar (19.1%) to initial values, but average $QMD_{>13cm}$ decreased slightly to 40.0 cm (Table 4.3). $RD_{>13cm}$ ranged between 10 to 30% (Figure 4.4d) and $QMD_{>13cm}$ ranged between 25 and 65 cm dbh (Figure 4.5d). Under 90th percentile weather conditions, the majority of stands had $RD_{>13cm}$ between 10 and 20% (Figure 4.4e) with $QMD_{>13cm}$ between 15 and 65 cm dbh (Figure 4.5e). Average $RD_{>13cm}$ decreased to 13.5% and average $QMD_{>13cm}$ decreased to 37.5 cm (Table 4.2). Under 97th percentile weather conditions, average $RD_{>13cm}$ decreased to 10.9% and average $QMD_{>13cm}$ decreased to 18.2 cm (Table 4.3). The majority of stands had $RD_{>13cm}$ of 10% (Figure 4.5f) and $QMD_{>13cm}$ less than 25 cm dbh (Figure 4.5f).

For stands with a 30-year fire return interval, $RD_{>13cm}$ ranged between 10 and 40% under 80th and 90th percentile weather conditions (Figure 4.4g and h), with an average $RD_{>13cm}$ of 31% and 25%, respectively. $QMD_{>13cm}$ ranged between 15 and 65 cm dbh, but the majority of stands had a $QMD_{>13cm}$ of 25 cm dbh (Figure 4.5 g and h). Under 97th percentile weather conditions, $RD_{>13cm}$ ranged between 10 and 30% (Figure 4.5i) with no change in average $RD_{>13cm}$ (Table 4.2). However, the majority of these stands had $QMD_{>13cm}$ between 15 and 25 cm dbh, with an average $QMD_{>13cm}$ of 23.5 cm (Table 4.2).

Structural characteristics associated with active crown fire risk

Classification tree analysis for active crown fire risk under 97th percentile weather conditions indicated that stand density index (SDI), quadratic mean diameter (QMD), and canopy base height (CBH) were important factors in determining the potential risk of an active crown fire (Figure 4.6).

Stands that were associated with low active crown fire risk were: 1) closed-canopied forests with low densities of seedlings/saplings and moderate canopy base heights; 2) open- to closed-canopied forests with low densities of seedlings/saplings; and

3) very open forests. In stands with SDI between 319.3 and 450, low active crown fire risk was predicted if QMD > 6.8 cm and CBH < 7.2 m (Figure 4.6). In stands with SDI between 139.5 and 319.3, low active crown fire risk was predicted if QMD > 6.5 cm (Figure 4.6). Low active crown fire risk was also predicted for stands with SDI < 139.5 (Figure 4.6).

Stands that were associated with moderate active crown fire risk were: 1) dense forests with low densities of seedlings/saplings; 2) closed-canopied forests with low densities of seedlings/saplings and tall canopy base height; and 3) open- to closed-canopied forests with high densities of seedlings/saplings. In stands with SDI > 450 moderate active crown fire risk was predicted if QMD > 6.8 cm (Figure 4.6). In stands with SDI between 319.3 and 450, moderate active crown fire risk was predicted if QMD > 6.8 cm and CBH > 7.2 m (Figure 4.6). In stands with SDI between 139.5 and 319.3, moderate active crown fire risk was predicted if QMD < 6.5 cm (Figure 4.6).

Stands that were associated with high active crown fire risk were closed-canopied forests with high densities of seedlings/saplings. Stands with SDI > 319.3 and QMD < 6.8 cm were at high risk of active crown fire (Figure 4.6).

Overall accuracy of active crown fire risk classification was 80%. Low and high risk for active crown fire was predicted with 87 – 88% accuracy. Moderate risk for active crown fire was classified correctly 60% of the time.

Discussion

Our simulation results of potential fire behavior of historical ponderosa pine forests of the Black Hills and the resulting structures are consistent with conditions associated with a mixed-severity fire regime (Agee 1998, Baker *et al.* 2007). Historical forest structures of the Black Hills were capable of supporting surface fire, passive crown fire, and active crown fire, but passive crown fire was the dominant fire behavior type. The historic prevalence of passive crown fire is attributed to the regeneration success of ponderosa pine in the Black Hills which resulted in multi-storied stands with low canopy base heights. Passive crown fire initiation is a function of surface fire intensity, forest canopy base height, and foliar moisture of tree crowns (Van Wagner 1977). Stands with a high density of seedlings and saplings require less surface fire intensity to ignite the foliage because of the shorter canopy base heights. In our simulations, stands that burned every 10 years under 80th and 90th percentile weather conditions killed the ponderosa pine regeneration before it could establish, resulting in stands with high canopy base heights and reduced probability of passive crown fire initiation. In contrast, stands that were modeled to burned every 20 or 30 years would allow time for ponderosa pine regeneration to reach sizes and densities that perpetuated the passive crown fire cycle. In some cases, the model projected passive crown fires in stands with tall canopy base heights. These fires occurred under model scenarios with high windspeeds and low foliage moistures, emphasizing the importance of considering the influence extreme weather conditions can have on fire behavior.

Weather conditions during a fire can also influence the severity of that fire. Ponderosa pine fire-induced mortality is size dependent. Greater amounts of crown damage are required to kill a tree as its diameter increases (Wyant *et al.* 1986, Ryan and Reinhardt 1988, Regelbrugge and Conard 1993, Stephens and Finney 2002, McHugh and Kolb 2003, Keyser *et al.* 2006, Sieg *et al.* 2006, Battaglia 2007). The amount of crown damage is, in turn, influenced by the interaction of crown base height and flame lengths.

In our simulations, flame lengths were less than 1 m in height under 80th percentile weather conditions for both surface and passive crown fires (Table 4.2). These resulted in mortality of seedlings, saplings, and some poles, but the larger diameter trees remained intact (Figure 4.3a, d, and g). These flames were not high enough to inflict enough crown damage to kill the overstory trees, as indicated by the simulated scorch heights of less than 6 m (Table 4.2). In contrast, under 97th percentile weather conditions, simulated flame lengths for passive crown fires ranged between 3.2 and 6.3 m with scorch heights ranging from 26.2 to 48.5 m (Table 4.2) resulting in projected high mortality across all diameter size classes and a loss of overstory trees (Figure 4.3c, f, and i).

Surface fires can also result in severe effects (Ryan *et al.* 1988, Raymond and Peterson 2005, Lentile *et al.* 2006b). In our simulations, surface fires that occurred under 97th percentile weather conditions had flame lengths of 2.3 m and scorch heights up to 20 m (Table 4.2), resulting in high mortality of trees <40 cm dbh (Figure 4.3c). The projected tree mortality in surface and passive crown fire under different weather conditions highlights the importance of considering both fire behavior and fire severity when discussing historical fire regimes.

The differences in projected mortality among different tree sizes under various weather and fire frequency scenarios resulted in stands with divergent development trajectories. In our simulations, forests that burned every 10 years under 80th and 90th percentile weather conditions resulted in open-canopied forests with large diameter (60 cm dbh) trees (Table 4.3; Figure 4.4a and b; Figure 4.5a and b). Including extreme weather conditions in the projections resulted in meadows, patches of young dense saplings/poles, or savannas of very few large diameter overstory trees (Table 4.3; Figure

4.4c; Figure 4.5c). These simulated forest structures are consistent with Graves' descriptions of lower elevation ponderosa pine forests along the prairie-woodland ecotone (Graves 1899) which burned every 10 to 12 years (Brown and Sieg 1999, Wienk *et al.* 2004). Graves described these forests as stands with old veterans scattered among the younger crop (Graves 1899:116), large diameter trees mixed with second-growth poles (Graves 1899: 108, 116, 129), even-aged sawtimber-sized forests (Graves 1899: 129), and forests of young, scrubby saplings about 40 to 50 years old (Graves 1899: 108).

Our simulations are also consistent with Graves' description of historical forest structures that he observed in the interior portions of the Black Hills (Graves 1899). Tree ring evidence indicates these forests burned every 20 to 35 years (Brown and Sieg 1996, 1999, Brown *et al.* 2000, Brown 2003). In our simulations, stands that burned every 20 or 30 years resulted in multi-storied and even-aged stands, but the density and the size distribution of the trees were dependent upon the weather conditions and fire frequency. When we modeled a 20 year fire interval under 80^{th} or 90^{th} percentile weather conditions, open- to closed-canopied, multi-storied stands with tree sizes ranging from poles to large sawtimber resulted (Table 4.3; Figure 4.4d and e; Figure 4.5d and e). Modeling under extreme weather conditions produced open, even-aged pole-sized forests (Table 4.3; Figure 4.4f; Figure 4.5f).

Increasing the fire interval to 30 years under 80^{th} and 90^{th} percentile weather conditions resulted in closed-canopied, fully-stocked, multi-storied stands (Table 4.3; Figure 4.4g and h; Figure 4.5g and h). On average, these stands would have high densities of pole-sized trees in the midstory compared to stands that burned every 20 years, but they would still maintain a large sawtimber overstory component. Increasing the fire return interval to 30 years under 97th percentile weather conditions produced open- to closed-canopied stands consisting of mostly pole-sized trees with scattered large sawtimber trees in the overstory (Table 4.3; Figure 4.4i; Figure 4.5i). Each of these simulated forest structures, again, is consistent with Graves' descriptions of interior ponderosa pine forests of the Black Hills (Graves 1899). He described both open-canopied and closed-canopied multi-cohort forests. These multi-cohort forests had a mixture of old veterans, poles, and saplings. Some forests also contained a mixture of large diameter veteran trees with poles (Graves 1899: 114, 123-125, 132, 136, 141, 145) or saplings (105) while others had a mixture of all three size classes (Graves 1899: 105, 109-110, 130, 132, 135-136, 141, 144, 149, 151, 156, 161-162). There were also descriptions of forests dominated by pole-sized trees with scattered veterans (Graves 1899: 108, 110, 131,141, 142, 145-146, 155-156) or pole-sized trees mixed with young trees (Graves 1899: 122). In some cases, forests were even-aged (Graves 1899: 111, 122, 124, 162).

Across the historical landscape, a fine-scale mosaic of dense and open forests within 100 m of each other was imbedded within a coarse-scale mosaic of forests varying in age, density, and structure (Brown and Cook 2006). Fires that occurred under moderate weather conditions contributed to the fine-scale mosaic of forest structure on the historical Black Hills landscape. Although superposed epoch analysis indicates that regional fire years between 1596 and 1900 in the Black Hills were significantly dry years and that non-fire years were significantly wet years (Brown 2006), detailed contemporary fire records from the Custer weather station indicate that lightning-ignited fires have occurred even under relatively wet periods. From 1970 to 2005, between 60 and 270 fires burned annually somewhere in the Black Hills, regardless of climatic conditions. While the majority of these fires were relatively small in size (0.08 to 4.0 ha), the potential of their size to increase was overwhelmed by fire suppression activities. Historically, these fires would likely have burned much larger acreages over the growing season until a weather event, fuel availability, or topographic features limited spread. Our simulations suggest that under moderate weather conditions these surface and passive crown fires would have maintained the same heterogeneous, multi-aged forest structure on a stand and regional level as suggested by Brown and Cook (2006).

While passive crown fire was likely a common event in historical Black Hills ponderosa pine forests, our simulations showed that the potential for active crown fire was low to moderate (Figure 4.2). To sustain an active crown fire the wind speed must be great enough to push the torching flame toward an adjacent tree crown to preheat the foliage to the ignition point. In addition, there must be enough crown biomass (e.g. canopy bulk density) to transport the flame from one crown to another and sustain the crown fire (Van Wagner 1977). In our simulations, stands that burned every 10 years required sustained windspeeds that exceeded 80.5 km h^{-1} in order to sustain an active crown fire. This was due to the lack of ladder fuels and low canopy bulk density. In contrast, stands that burned every 20 or 30 years had a low to moderate probability of an active crown fire due to their greater relative density, presence of ladder fuels, and increased canopy bulk density. The model projected that these longer fire return interval stands would require sustained windspeeds of 40.2 to 80.5 km h^{-1} to maintain an active crown fire event. The rarity of having sustained winds at such high levels in the Black Hills combined with the fine-scale heterogeneity of forest structure would have limited the frequency and extent of active crown fires on the historical Black Hills landscape, but obviously would not have precluded them.

When extreme weather historically did coincide with fire ignition, the resulting fires were large in extent, interacted with the fine-scale structure mosaic, and contributed to the coarse-scale mosaic across the landscape. This may be illustrated by the Jasper fire which burned in the Black Hills under extremely dry and windy conditions (Benson and Murphy 2003) resulted in a complex burn mosaic of low, moderate, and high severity patches on the landscape (Lentile *et al.* 2005, Lentile *et al.* 2006a). These differences in burn severity varied with pre-fire forest structure and slope (Lentile *et al.* 2006a). Stands with low density of large trees (average stand diameter > 24 cm dbh) on gentle slopes burned at low or moderate burn severity, but steeper slopes resulted in high burn severity. Stands with a high density of large trees or many small trees also burned at high severity (Lentile *et al.* 2006a). These differences in burn severity resulted in the coarse-scale mosaic of charred hillsides, open parks, broken forests, even-aged stands, old veterans scattered among younger trees, and multi-aged forests; structures that are consistent with historical descriptions by Graves (1899). It is unlikely that large stand-replacing crown fires were a dominant disturbance that shaped the historical Black Hills landscape. Instead, the Black Hills landscapes was likely governed by a mixture of frequent low to moderate surface and passive crown fires. This created multi-cohort ponderosa pine forests punctuated with infrequent moderate to high severity crown fires as suggested for ponderosa pine forests of the Colorado Front Range (Brown *et al.* 1999, Veblen *et al.* 2000, Huckaby *et al.* 2001, Ehle and Baker 2003) and the Northern Rockies (Barrett 1988, Arno *et al.* 1995, 1997, Baker *et al.* 2007).

Fuel treatments

Based on our simulations of potential fire behavior in historical Black Hills ponderosa pine forests, ecological restoration of historical forest structure will not always be compatible with fuel reduction treatment goals. Fires that would occur under extreme weather in many of the simulated historical forest structures fires would have tall flame lengths (Table 4.4), high mortality (Table 4.4), and moderate to high risk of active crown fires (Figure 4.2). Fortunately, there were some historical stand structures with low active crown fire risk, moderate tree mortality, and short flame lengths under extreme weather conditions that could be used as a guideline for restoration/fuels reduction treatments.

Low active crown fire risk can be achieved in stands with RD ranging from 12.5 to 29% if QMD > 6.5 cm, otherwise risk for active crown fire increases to moderate (Figure 4.6). While these stands are relatively open, this result highlights the importance of maintaining low densities of seedling/sapling regeneration in order to maintain the longevity of fuel reduction treatments in the Black Hills. Low active crown fire risk can also be also achieved in stands with RD that is optimal for timber production (29% to 40.5%) if QMD > 6.8 cm and average canopy base height <7.2 m (Figure 4.6). Active crown fire risk increases to moderate if canopy base height > 7.2 m. This seems counterintuitive since having a taller canopy base height often results in lower crown fire initiation. However, stands with tall canopies and high tree density increases canopy bulk density to levels that puts the stand at risk for an active crown fire. This suggests that both ladder fuels and fuels within the upper canopy stratum should be considered when implementing a fuels reduction treatment. Finally, low active crown fire risk can be achieved in stands containing trees of any size when RD < 12.5% (Figure 4.6) due to low canopy bulk densities.

Some historical stand structures were prone to moderate to active crown fire. Stands currently having these structures should be treated in areas where risk of crown fire needs to be reduced. Our simulations suggest that in stands intended for timber production (RD > 29%), managers should keep QMD > 6.8 cm to avoid high risk of active crown fire (Figure 4.6). Closed-canopied stands with QMD < 6.8 cm have high densities of saplings and consequently high canopy bulk densities and low canopy base heights which make them conducive to active crown fire spread (Figure 4.6) and high severity burning (Lentile *et al.* 2006a). Although dense stands (RD > 40.5%) with QMD > 6.8 cm are at moderate risk for active crown fire (Figure 4.6), they were found to burn with high severity during the Jasper wildfire (Lentile *et al.* 2006a). This suggests that maintaining stands at moderate active crown fire risk does not necessarily equate to low fire severity. As these stands grow to sawtimber size, they are still at moderate risk of active crown fire due to high canopy bulk density.

Another goal of a fuels reduction treatment is to modify fire behavior enough to increase the effectiveness of fire suppression activities (Agee *et al.* 2000). In our extreme weather simulations, the stands structures that resulted from 20- and 30-year fire return intervals and 80^{th} percentile weather conditions scenarios had predicted flame lengths (Table 4.4) conducive to containment with handlines and direct attack by firefighters (Rothermel 1983) and low active crown fire risk (Figure 4.2). Fires with flame lengths between 1.2 m and 2.4 m are too intense for direct attack, but bulldozers, pumpers, and retardant aircraft can be effective in attacking a fire (Rothermel 1983). Our simulations indicated that stands which developed under 20- and 30-year fire return intervals and 90th percentile weather conditions had predicted flame lengths (Table 4.4) within the range for equipment to be effective and were also at low to moderate active crown fire risk (Figure 4.2). These open- to closed-canopied, multi-storied stands (Figure 4.4 and 4.5; Table 4.3) were the most prevalent forest structures described by Graves (1899), suggesting that these forest structures were quite resilient to infrequent, extreme weather events (Holling 1973).

Fires with flame lengths greater than 2.4 m present serious containment problems because of the high potential for torching, spotting, and crowning behavior (Rothermel 1983). Stand structures similar to those that developed under our simulated 20- or 30-year fire return interval and 97th percentile weather conditions would have serious containment problems (Table 4.2) and be at moderate to high active crown fire risk (Figure 4.2) due to a high density of sapling and pole-sized trees (Figure 4.4 and 4.5; Table 4.3). Stands which developed under a 10-year fire return interval were projected to have low active crown fire risk (Figure 4.2) due to low canopy bulk density. These stands would also have serious containment problems (Table 4.4), because windspeeds would not be reduced in the open understory. Such stand structures could also increase the potential for spot fires, increase fire extent, and fire damage depending upon the spatial arrangement of stand structures on the landscape.

Our simulations, dendrochronological reconstructions (Brown and Cook 2006), and historical inventories (Graves 1899) all provide evidence that the Black Hills landscape was influenced by a mixed-severity fire regime and likely consisted of a diverse assortment of stand structures, densities, and tree sizes. Fuel reduction treatments and/or restoration treatments should therefore attempt to create a patchy distribution of diverse forest structures across the Black Hills landscape instead of implementing uniform targets (Stephens and Fulé 2005). To be effective, these treatments should be larger than the stand scale, ranging in sizes of 50 to 500 ha (Agee and Skinner 2005, Stephens and Fulé 2005). Currently, the Black Hills landscape is dominated by large contiguous areas of even-aged forests, with only 2% of the forest land area nonstocked (DeBlander 2002). Current basal areas of these forests are similar to historical basal areas, but average tree size has decreased substantially (Brown and Cook 2006) with the majority of trees < 40 cm dbh (DeBlander 2002). Trees <40 cm dbh are more susceptible to fire damage (Keyser *et al.* 2006), so retention of large diameter trees in fuel reduction treatments and across the landscape is encouraged (Agee and Skinner 2005) to ensure tree survival and to aid in the emulation of historical forest structure. We further suggest that the creation of large openings with mechanical treatments and/or prescribed fire is warranted to serve as natural fuel breaks, similar to those found on the historical landscape. As with any fuels reduction treatment that utilizes mechanical treatments, surface fuels left after harvest will need to be treated with prescribed fire or other measures to ensure the effectiveness of the treatment (Agee and Skinner 2005, Cram *et al.* 2006). Empirical studies are also needed to verify the effectiveness of these treatments in reducing fire behavior and mortality under a range of weather conditions. In addition, a better understanding of the spatial interaction of fuel reduction treatments is needed to reduce the potential for large conflagrations.

Conclusion

Our simulations indicate that historical forest structures of the Black Hills were capable of supporting surface fire, passive crown fire, and active crown fire, but passive crown fire was most likely the dominant fire behavior type. Weather conditions associated with fire behavior determined the size of trees that were likely to be killed and the structural and spatial characteristics of resulting forests. Our projections further show that interactions among weather and fire behavior resulted in diverse forest structures across the historical landscape of the Black Hills. Our results substantiate several lines of evidence which suggest that the historical Black Hills landscape was shaped by a mixed-severity fire regime (Lentile *et al.* 2005, Baker *et al.* 2007) which created a landscape mosaic of multi-cohort patches of various densities and structure types (Graves 1899, Brown and Cook 2006). However, restoration of the full suite of historical Black Hills forest structures will not result in reduced crown fire risk and mortality in all cases. Our results do suggest that it may be possible to create fuel reduction treatments that reduce crown fire risk and mortality while concurrently restoring some elements of historical forest structure.

Criticism of fuel reduction/forest restoration treatments on public lands (Shinnemann and Baker 1997, Schoennagel *et al.* 2004, Platt *et al.* 2006, Baker *et al.* 2007) is often fueled by the knowledge that fire behavior in some forests is not outside the historic range of variability. While these criticisms may have an ecological basis, they ignore many social and political realities including the existence of a wildland-urban interface, human safety, and multiple-use mandates. Although passive and active crown fire was a component of the historical fire regime of the Black Hills, the considerable amount of private land dispersed within the National Forest boundaries and the substantial focus on timber production in the Black Hills limits the reintroduction of crown fire. Instead, management activities within the wildland urban interface and other areas of the forest will more likely focus on fire risk reduction. However, fire risk reduction activities do not necessarily have to preclude ecological processes. Instead, elements of historical forest structure identified in this study that lowered crown fire risk could be incorporated into areas where resources were at highest risk. While this may not fully restore the ecological integrity of the system, we must be cognizant of the social and political issues and work with the human dimension of this problem.

Literature cited

Agee, J.K. 2005. The complex nature of mixed severity fire regimes. In: "Mixed severity fire regimes: ecology and management symposium proceedings', 17-19 November 2004, Spokane, WA, USA. (Eds L. Taylor, J Zelnik, S. Cadwallader, B. Highes).

Agee, J.K. 1998. Fire and pine ecosystems. In: Ecology and biogeography of *Pinus*. Edited by D.M. Richardson. Cambridge University Press, Cambridge, UK. Pp. 193-218.

Agee, JK, Skinner, CN. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211, 83-96.

Agee, J.K., Bahro, B., Finney, M.A., Omi, P.N., Sapsis, D.B., Skinner, C.N., van Wagtendonk, J.W., Weatherspoon, C.P. 2000. The use of shaded fuelbreaks in landscape fire management. Forest Ecology and Management. 127: 55-66.

Albini, F.A. 1976. Estimating wildfire behavior and effects. Gen. Tech. Rep. INT-30. Ogden, UT: USDA, FS, Intermountain Research Station. 92 p.

Anderson, H. E. 1982. Aids to determining fuel models for estimating fire behavior. Ogden, Utah, USDA For. Serv. GTR-INT-122, Intermountain Forest and Range Experiment Station. 122: 22 p.

Andrews, P.L. 1986. BEHAVE: fire behavior prediction and fuel modeling system-BURN subsystem, Part 1. USDA Gen. Tech. Rep. INT-194. 130 p.

Arno, SF, Smith, HY., and Krebs, M.A. 1997. Old growth ponderosa pine and western larch stand structures: influences of pre-1900 fires and fire exclusion. USFS, Research Paper, INT-RP-495, Ogden, UT.

Arno, SF, Scott JH, Hartwell, MG. 1995. Age-class structure of old growth ponderosa pine/Douglas-fir stands and its relationship to fire history. USDA FS, Intermountain Forest and Range Experiment Station Research Paper, INT-RP-481. Ogden, UT.

Baker, W.L., Veblen, T.T., Sherriff, R.L. 2007. Fire, fuels and restoration of ponderosa pine-Douglas fir forests in the Rocky Mountains, USA. Journal of Biogeography 34(2): 251-269.

Baker, W.L., Ehle, D. 2001. Uncertainty in surface-fire history: The case of ponderosa pine forests in the western United States. Can. J. For. Res. 31: 1205-1226.

Barrett, S.W. 1988. Fire suppression's effects on forest succession within a central Idaho wilderness. Western Journal of Applied Forestry. 3: 76-80.

Battaglia, M.A., 2007. Maintaining fuel treatments with prescribed fire in ponderosa pine (*Pinus ponderosa*) forests of the Black Hills, South Dakota. Ph.D. Dissertation. Colorado State University, Fort Collins, Colorado.

Benson, R.P. and M. P. Murphy. 2003. Wildland fires in the Black Hills. In: Proceedings: 2nd International Wildland Fire Ecology and Fire Management Congress. American Meteorological Society. November 2003, Orlando, FL.

Bradshaw, L., S. Brittain. 1999. FireFamily Plus. Software available from USDA Forest Service, Rocky Mountain Research Station, Missoula, MT.

Breiman, L., Friedman, JH, Olshen, RA, Stone, CJ. 1984. Classification and regression trees. Wadsworth International Group: Belmont, CA.

Bonnet, V.H., Schoettle, A.W., Shepperd, W.D. 2005. Postfire environmental conditions influence spatial pattern of regeneration for *Pinus ponderosa*. Can. J. For. Res. 35: 37-47.

Brown, P.M. 2006. Climate effects on fire regimes and tree recruitment in Black Hills ponderosa pine forests. Ecology 87(10): 2500-2510.

Brown, P.M. 2003. Fire, climate, and forest structure in Black Hills ponderosa pine forests. Dissertation. Colorado State University, Ft. Collins, CO. 103 p.

Brown, P. M., Ryan, M. G., Andrews, T. G. 2000. Historical surface fire frequency in ponderosa pine stands in research natural areas, Central Rocky Mountains and Black Hills, USA. Natural Areas Journal. 20 (2): 133-139.

Brown, P.M., Kauffman, M.R., Shepperd, W.D. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology. 14: 513-532.

Brown, P.M. and Cook, B. 2006. Early settlement forest structure in Black Hills ponderosa pine forests. Forest Ecology and Management 223: 284-290.

Brown, P. M., Sieg, C. H. 1999. Historical variability in fire at the ponderosa pine-Northern Great Plains prairie ecotone, southeastern Black Hills, South Dakota. Ecoscience. 6(4): 539-547.

Brown, P. M., Sieg, C. H. 1996. Fire history in interior ponderosa pine forests of the Black Hills, South Dakota, USA. International Journal of Wildland Fire. 6: 97-105.

Covington, W. W., Moore, M. M., 1994. Southwestern ponderosa pine forest structure: changes since Euro-American settlement. Journal of Forestry 92(1): 39-47.
Covington and Moore 1994

Covington, W.W, Fulé,P.Z., Moore, M.M., Hart, S.C., Kolb, T.E, Mast, J.N., Sackett, S.S., Wagner, M.R. 1997. Restoration of ecosystem health in southwestern ponderosa pine forests. J. For. 95(4): 23-29.

Cram, D., Baker, T., Boren, J., 2006. Wildland fire effects in silviculturally treated vs. untreated stands of New Mexico and Arizona. Research Paper RMRS-RP-55. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 28p.

DeBlander. L.T. 2002. Forest resources of the Black Hills National Forest. USDA Forest Service, Rocky Mountain Research Station. 16 p.

Dodge, R. I. 1965. The Black Hills. Minneapolis, MN: Ross and Haines. 151 p.

Driscoll, D. G., Hamade, G. R., Kenner, S. J. 2000. Summary of precipitation data for the Black Hills area of South Dakota, water years 1931-1998: USGS Open file report 00-329. 163 p

Ehle, D.S. and Baker, W.L. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. Ecological Monographs. 73: 543-566.

Fiedler, C.E., Keegan, C.E., III, Woodall, C.W., Morgan, Todd. A. 2004. A strategic assessment of crown fire hazard in Montana: potential effectiveness and costs of hazard reduction treatments. Gen. Tech. Rep. PNW-GTR-622. Portland, OR: USDA, FS, Pacific Northwest Research Station. 48p.

Finney, Mark A. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science 47(2): 219–228.

Finney, M.A., McHugh, C.W., and Grenfell, I.C. 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Can. J. For. Res. 35: 1714-1722.

Finney, M.A. 2003. Calculation of fire spread rates across random landscapes. Int. J. Wildland Fire, 12: 167–174.

Fulé, P.Z., Covington, W.W., Stoddard, M.T., Bertolette, D. 2006. "Minimal-impact" restoration treatments have limited effects on forest structure and fuels at Grand Canyon, USA. Restoration Ecology 14(3): 357-368.

Fulé, P.Z., Crouse, J.E., Cocke, A.E., Moore, M.M., Covington, W.W. 2004. Changes in canopy fuels and potential fire behavior 1880-2040: Grand Canyon, Arizona. Ecol. Mod. 175: 231-248.

Fulé, P.Z., Covington, W.W., Smith, H.B., Springer, J.D., Heinlein, T.A., Huisinga, K.D., Moore, M.M. 2002. Comparing ecological restoration alternatives: Grand Canyon, Arizona. For. Ecol. Mgmt. 170: 19-41.

Fulé, P.Z., Waltz, A.E.M., Covington, W.W., Heinlein, T.A. 2001. Measuring forest restoration effectiveness in hazardous fuels reduction. J. For. 99(11): 24-29.

Fulé, P.Z., Covington, W.W., Moore, M.M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecol. Apps. 7(3): 895-908.

Grafe, E., Horsted, P. 2002. Exploring with Custer: The 1874 Black Hills expedition. Custer, SD: Golden Valley Press.

Graves, H. S. 1899. The Black Hills Forest Reserve. *In:* The nineteenth annual report of the survey, 1897-1898. Part V. Forest Reserves. Washington, DC: U.S. Geological Survey: 67-164.

Holling, C.S. 1973. Resilience and stability of ecological systems. Annu. Rev. Ecol. Syst. 4: 1-23.

Huckaby, L.S., Kaufmann, M.R., Stoker, J.M., Fornwalt, P.J. 2001. Landscape patterns of montane forest age structure relative to fire history at Chessman Lake in the Colorado Front Range. In: Ponderosa pine ecosystems restoration and conservation: Steps toward stewardship, Conference Proceedings. Flagstaff, AZ, April 25-27, 2000. Editors: R.K. Vance, C.B. Edminster, W.W. Covington, and J.A. Blake. Pp. 19-27. USFS Proceedings, RMRS-P-22, Fort Collins, CO.

Jain, TB, and Graham, RT. 2004. Is forest structure related to fire severity? Yes, no, and maybe: methods and insights in quantifying the answer. In: Silviculture in special places: proceedings of the National Silviculture Workshop, 8-11 September 2003, Granby, CO. (Comps. WD Shepperd, LG Eskew). Pp. 217-234. USDA FS, RMRS Proceedings RMRS-P-34, Fort Collins, CO.

Jonsson, B., Holm, S., Kallur, H., 1992. A forest inventory method based on density-adapted circular plot size. Scand. J. For. Res. 7, 405–421.

Keyser, T.L., 2007. Changes in forest structure, community composition, and long-term development in ponderosa pine forests following a mixed-severity wildfire in the Black Hills, SD, USA.

Keyser T.L., Smith, F.W., Lentile L.B., Shepperd, W.D., 2006. Modeling postfire mortality of ponderosa pine following a mixed-severity wildfire in the Black Hills: The role of tree morphology and direct fire effects. For Sci. 52, 530-539.

Lentile, L.B., Smith, F.W., Shepperd, W.D. 2006a. Influence of topography and forest structure on patterns of mixed severity fire in ponderosa pine forests of the South Dakota Black Hills, USA. International Journal of Wildland Fire. 15: 557-566.

Lentile, LB., Holden, ZA., Smith, AMS., Falkowski, MJ., Hudak, AT., Morgan, P., Lewis, SA., Gessler, PE., Benson, NC. 2006b. Remote sensing techniques to assess active fire and post-fire effects. International Journal of Wildland Fire. 15: 319-345.

Lentile, LB, Smith, FW., and Shepperd, W.D. 2005. Patch structure, fire-scar formation and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. Canadian Journal of Forest Research. 35: 2875-2885.

Lentile, L. 2004. Fire Effects and Vegetation Recovery following the Jasper Fire, Black Hills National Forest, SD. Colorado State University.

Lessard V.C., Drummer T.D., Reed DD. 2002. Precision of density estimates from fixed-radius plots compared to n-tree distance sampling. For. Sci. 48(1): 1-6.

Ludlow, W. 1875. Report of a reconnaissance of the Black Hills of Dakota. U.S. Government Printing Office. Washington, D.C.

Lynch, T.B., Wittwer, R.F., 2003. n-Tree distance sampling for per-tree estimates with application to unequal-sized cluster sampling of increment core data. Can. J. For. Res. 33, 1189–1195.

Mast, J.N., Fulé, P.Z., Moore, M.M., Covington, W.W., Waltz, A.E.M. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Eco. Apps. 9(1): 228-239.

McAdams, A.G. 1995. Changes in ponderosa pine forest structure in the Black Hills, South Dakota, 1874-1995. M.S. Thesis. Northern Arizona University, Flagstaff.

McHugh, C.W., Kolb, T.E. 2003. Ponderosa pine mortality following fire in northern Arizona. Int. J. Wild. Fire 12: 7-22.

Moghaddas, J.J. 2006. A fuel treatment reduces potential fire severity and increases suppression efficiency in a Sierran mixed conifer forest. In: Andrews, P.L., Butler, B.W., comps. 2006. Fuels Management-How to measure success: Conference proceedings. 2006 28-30 March, Portland, OR. Proceedings RMRS-P-41. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station.

Moore, P.G. 1954. Spacing in plant populations. Ecology. 35: 222-227.

Newton, H., Jenney, W. P. 1880. Report on the geology and resources of the Black Hills of Dakota. U.S. Department of Interior, U.S. Geographical and Geological Survey of the Rocky Mountain Region. Washington, DC: U.S. Government Printing Office.

Platt, R.V., Veblen, T.T., Sherriff, R.L. 2006. Are wildfire mitigation and restoration of historic forest structure compatible? A spatial modeling assessment. Annals of the association of American Geographers, 96(3): 455-470.

Pollet, J. and Omi, P.N. 2002. Effects of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire 11: 1-10.

Progulske, D. R., 1974. Yellow ore, yellow hair, yellow pine: A photographic survey of a century of forest ecology. Bull. 616. Brookings, SD: Agricultural Experiment Station, South Dakota State University.

Raymond, CL, and Peterson, D.L. 2005. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Canadian Journal of Forest Research. 35: 2981-2995.

Regelbrugge, J.C. and S.G. Conard. 1993. Modeling tree mortality following wildfire in *Pinus ponderosa* forests in the central Sierra Nevada of California. IJWF 3(3): 139-148.

Reinhardt, Elizabeth, Crookston, Nicholas L. (Technical Editors). 2003. The Fire and Fuels Extension to the Forest Vegetation Simulator. Gen. Tech. Rep. RMRS-GTR-116. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 209 p.

Rothermel, R.C. 1983. How to predict the spread and intensity of forest and range fires. Gen. Tech. Rep. INT-143. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station. 161 p.

Rothermel, R.C. 1972. A mathematical model for predicting fire spread in wildland fuels. USDA For. Serv. Res. Pap. INT-115. 40 p.

Ryan, K.C. and Reinhardt, E.D. 1988. Predicting postfire mortality of seven western conifers. Can. J. For. Res. 18(10): 1291-1297.

Ryan, K.C., Peterson, D.L., and Reinhardt, E.D. 1988. Modeling long-term fire-caused mortality of Douglas-fire. For. Sci. 34(1): 190-199.

SAS Institute, 2001. SAS/STAT software, version 9.1 of the SAS system for Windows 2000. SAS Institute Inc., Cary, NC.

Schoennagel, T., Veblen, T.T., Romme, W.H. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54(7): 661-676.

Stephens, S.L. and Fulé, P.Z. 2005. Western pine forests with continuing frequent fire regimes: Possible reference sites for management. J of Forestry. 103(7): 357-362.

Stephens, S.L. and M.A. Finney. 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. For. Ecol. Manage. 162: 261-271.

Scott, Joe H. 2003. Canopy fuel treatment standards for the wildland-urban interface. In: " Fire, fuel treatments, and ecological restoration: conference proceedings; 2002 April 16-18; Fort Collins, CO. Omi, Philip N., Joyce, Linda A., tech. eds. 2003. Proceedings RMRS-P-29. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, p. 29-37.

Scott, J.H., Reinhardt, E.D. 2001. Assessing crown potential by linking models of surface and crown fire behavior. USDA Forest Service Research Paper RMRS-RP-29, Fort Collins.

Shepperd, Wayne D., Battaglia, Michael A. 2002. Ecology, silviculture, and management of Black Hills ponderosa pine. Gen. Tech. Rep. RMRS-GTR-97. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 110 p.

Shinneman, D. J., Baker, W. L. 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in Ponderosa pine landscapes of the Black Hills. Conservation Biology. 11(6): 1276-1288.

Sieg, C.H., McMillin, J.D., Folwer, J.F., Allen, K.K., Negron, J.F., Wadleigh, L.L., Anhold, J.A., Gibson, K.E. 2006. Best predictors for postfire mortality of ponderosa pine trees in the intermountain west. Forest Science 52(6): 718-728.

Veblen, T.T. 2003. Key issues in fire regime research for fuels management and ecological restoration. In: USDA Forest Service Proceedings RMRS-P-29. Pgs. 259-276.

Veblen, T.T., Kitzberger, T., Donnegan, J. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications, 10: 1178-1195.

Van Dyck, M.G. 2000. Keyword reference guide for the forest vegetation simulator. USDA Forest Service. Forest Management Service Center, Fort Collins, CO.

Van Wagner, C.E. 1977. Conditions for the start and spread of crown fires. Can. J. For. Res. 7: 23-34.

White, A.S. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. Ecology. 66: 589-594.

Wienk, C.L., Sieg, C.H., McPherson, G. 2004. Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. For. Ecol. Manage. 192: 375-393.

Wilmes, L., Martinez, D., Wadleigh, L., Denton, C., Geisler, D. 2002. Apache-Sitgreaves National Forests Rodeo-Chediski Fire Effects Summary Report. Springerville (AZ): Apache-Sitgreaves National Forest.

Wyant, J.G., Omi, P.N., Laven, R.D. 1986. Fire induced tree mortality in a Colorado ponderosa pine/Douglas-fir stand. Can. J. For. Res. 32: 49-59.

Table 4.1: Percent weather conditions for the Custer, SD weather station, 1964 to 2004 used for wildfire simulations.

Weather parameter	80th percentile	90th percentile	97th percentile
1-h fuel moisture (%)	6.6	5.9	4.4
10-h fuel moisture (%)	8.3	7.3	5.5
100-h fuel moisture (%)	11.4	10.3	8.2
Herbaceous fuel moisture (%)	72.6	62.3	47.9
Woody fuel moisture (%)	112	101.8	88.5
Temperature (°C)	30.0	31.1	33.3
Probable maximum 1-minute windspeed (km h^{-1})	21	24	30.6

Table 4.2: Predicted average flame length (m), average scorch height (m) and associated average canopy base height (m) for surface and passive crown fires in historic Black Hills ponderosa pine stands developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions. Numbers in parentheses are ± standard error of the mean. Predicted values represent the fire behavior and effects that occurred under the associated weather scenario.

		80th percentile			90th percentile			97th percentile		
Fire Interval (years)	Fire Behavior	CBH (m)	Flame Length (m)	Scorch Height (m)	CBH (m)	Flame Length (m)	Scorch Height (m)	CBH (m)	Flame Length (m)	Scorch Height (m)
10	Surface	8.06 (0.69)	1.05 (0.02)	4.9 (0.08)	14.7 (0.19)	1.59 (0.04)	10.8 (0.26)	16.7 (1.41)	2.33 (0.12)	20.1 (1.1)
	Passive	0.94 (0.04)	1.14 (0.02)	5.4 (0.06)	1.12 (0.05)	1.96 (0.02)	13.2 (0.13)	1.19 (0.06)	3.20 (0.01)	26.2 (0.11)
20	Surface	1.40 (0.35)	0.77 (0.01)	3.8 (0.05)	n/a	n/a	n/a	n/a	n/a	n/a
	Passive	0.76 (0.02)	1.02 (0.02)	5.9 (0.18)	0.74 (0.02)	2.04 (0.04)	15.2 (0.34)	0.68 (0.01)	5.06 (0.01)	40.7 (0.66)
30	Surface	0.94 (0.02)	0.68 (0.01)	3.3 (0.06)	0.93 (0.02)	0.69 (0.01)	3.5 (0.11)	n/a	n/a	n/a
	Passive	0.87 (0.01)	1.03 (0.04)	6.2 (0.33)	0.84 (0.14)	2.06 (0.07)	15.5 (0.51)	0.67 (0.01)	6.3 (0.23)	48.5 (1.6)

Table 4.3: The average relative density and quadratic mean diameter of trees >13 cm and the change in these variables for historic Black Hills ponderosa pine stands burned at different fire return intervals (10-, 20-, and 30-year) and weather percentiles (80^{th}, 90^{th}, and 97^{th}) over a 100 year simulation. Numbers in parentheses are ± standard error of the mean.

			80^{th} percentile		90^{th} percentile		97^{th} percentile	
Fire Interval (years)	Variable	Initial (SEM)	Average (SEM)	Average Change	Average (SEM)	Average Change	Average (SEM)	Average Change
10	RD (%)	19.0 (1.2)	9.7 (0.8)	-9.3	7.9 (0.9)	-11.1	2.6 (0.6)	-16.4
	QMD (cm)	41.8 (1.2)	59.5 (1.2)	+17.7	61.2 (1.3)	+19.4	35.1 (1.7)	-6.7
20	RD (%)	19.0 (1.2)	19.1 (0.7)	+0.1	13.5 (0.7)	-5.5	10.9 (0.3)	-8.1
	QMD (cm)	41.8 (1.2)	40.0 (1.3)	-1.8	37.5 (1.5)	-4.4	18.2 (0.2)	-23.6
30	RD (%)	19.0 (1.2)	31.0 (0.6)	+12.0	25.0 (0.8)	+6.0	18.9 (0.5)	-0.1
	QMD (cm)	41.8 (1.2)	33.3 (1.3)	-8.5	34.0 (1.3)	-7.8	23.5 (0.7)	-18.3

Table 4.4: Predicted average flame lengths, percent average basal area mortality, and average volume (m^3) mortality (± standard error of the mean) for a fire burning under 97th percentile weather conditions for stands that developed under 80th, 90th, and 97th percentile weather conditions and 10-, 20-, and 30-year fire return intervals.

	80th percentile			90th percentile			97th percentile		
Fire Interval (years)	Flame length (m)	Mortality BA (%)	Mortality Volume (m^3)	Flame length (m)	Mortality BA (%)	Mortality Volume (m^3)	Flame length (m)	Mortality BA (%)	Mortality Volume (m^3)
10	2.68 (0.04)	73.6 (2.3)	25.3 (1.5)	2.78 (0.04)	74.7 (2.3)	19.6 (1.4)	3.11 (0.03)	86.3 (2.5)	1.6 (0.2)
20	0.98 (0.02)	34.7 (1.8)	9.5 (0.3)	1.92 (0.03)	61.0 (2.5)	13.2 (0.4)	4.20 (0.06)	98.8 (0.05)	11.8 (0.4)
30	0.94 (0.02)	33.5 (1.2)	14.5 (0.4)	1.76 (0.05)	54.6 (1.8)	22.9 (0.5)	4.90 (0.14)	97.0 (0.35)	31.7 (0.9)

Figure captions

Figure 4.1: Average potential fire behavior (solid=surface fire; hatched=passive crown fire) for historic Black Hills ponderosa pine forest stands (n=112) for stands developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions as predicted by FFE-FVS. Potential fire behavior was calculated for the last 60 years of the 100 year simulation.

Figure 4.2: Frequency distribution for the active crown fire risk under 97th percentile weather conditions for historic Black Hills ponderosa pine forests developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions. Risk was assessed for the last 60 years of the 100-year simulation. Active crown fire risk was based on the Crowning Index, the 6.1 m windspeed required to sustain an active crown fire, for 97th percentile weather conditions. Low risk = >80.5 km/hr, Mod risk = 40.2 to 80.5 km/hr, and High risk = < 40.2 km/hr (Fiedler *et al* 2004).

Figure 4.3: Potential mortality ($\pm$ standard error of the mean) by fire type (surface, passive, or active crown fire) of historic Black Hills ponderosa pine forests for stands developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions as predicted by FFE-FVS. Significant differences ($p<0.05$) of mortality among dbh size class for each fire behavior type within a fire return interval/weather scenario are designated by an asterisk.

Figure 4.4: Frequency distribution for the average relative density of trees>13 cm for the last 60 years of the 100-year simulation of historic Black Hills ponderosa pine forest stands (n=112) developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions.

Figure 4.5: Frequency distribution for the average quadratic mean diameter of trees>13 cm for the last 60 years of the100-year simulation of historic Black Hills ponderosa pine forest stands (n=112) developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions.

Figure 4.6: Classification tree for estimating risk of active crown fire based on stand attributes of historic Black Hills ponderosa pine forests under 97th percentile weather conditions. SDI is the Stand density index and is defined as the number of trees per hectare stands would have if the trees had an average size of 25.4 cm DBH. CBH = canopy base height (m), QMD = quadratic mean diameter (cm), Low = low active crown fire risk (>80.5 km/hr), Moderate = moderate active crown fire risk (40.2 to 80.5 km/hr), and High = high active crown fire risk (<40.2 km/hr). Values greater than the presented value classify to the right, lesser values classify to the left.

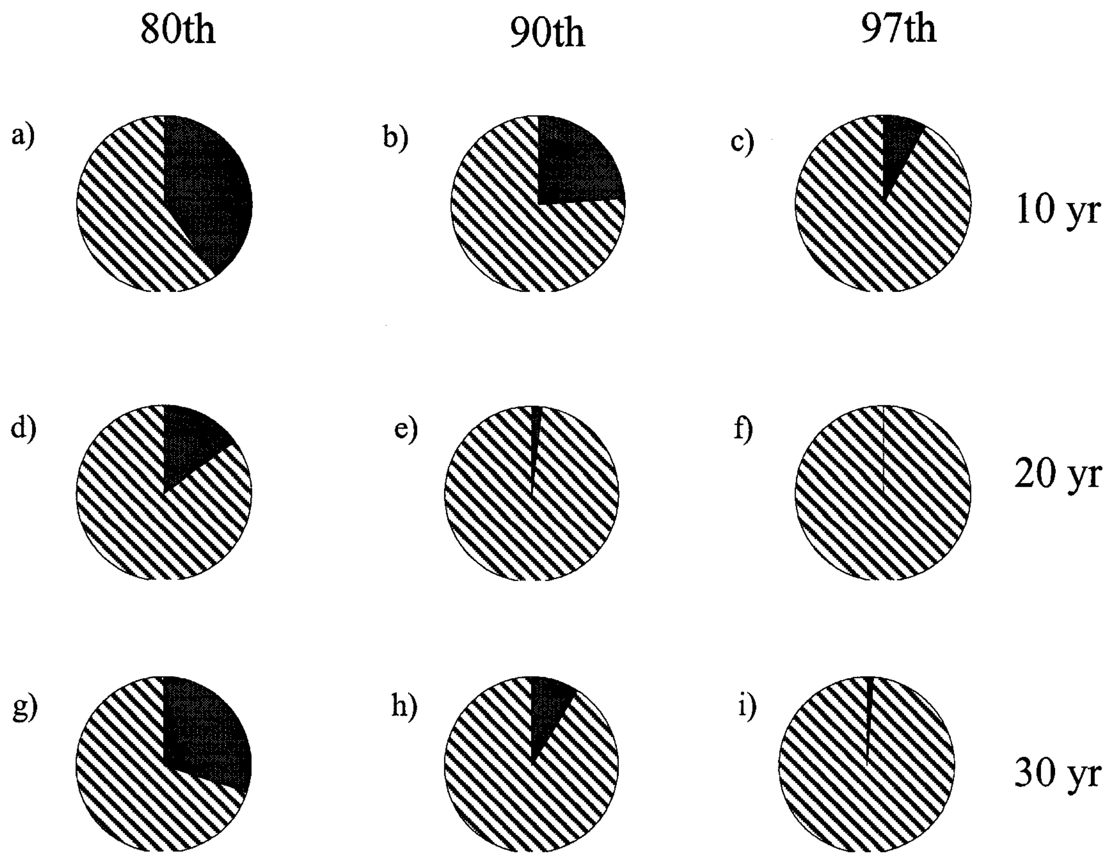

Figure 4.1: Average potential fire behavior (solid=surface fire; hatched=passive crown fire) for historic Black Hills ponderosa pine forest stands (n=112) for stands developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions as predicted by FFE-FVS. Potential fire behavior was calculated for the last 60 years of the 100 year simulation.

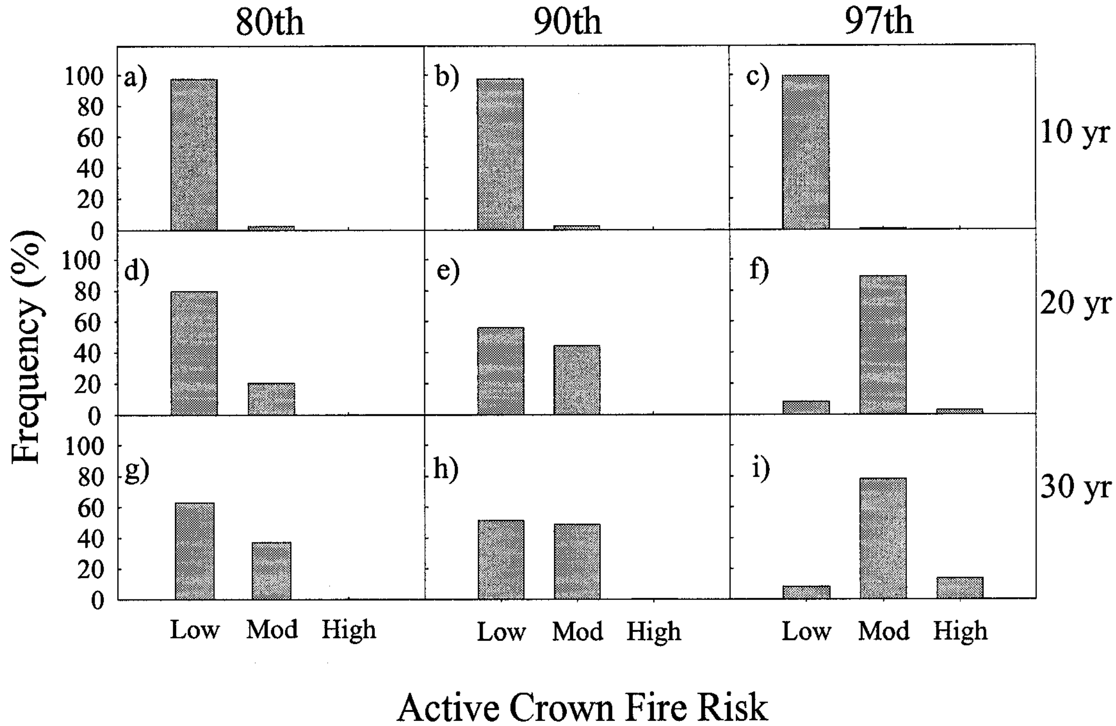

Figure 4.2: Frequency distribution for the active crown fire risk under 97th percentile weather conditions for historic Black Hills ponderosa pine forests developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions. Risk was assessed for the last 60 years of the 100-year simulation. Active crown fire risk was based on the Crowning Index, the 6.1 m windspeed required to sustain an active crown fire, for 97th percentile weather conditions. Low risk = >80.5 km/hr, Mod risk = 40.2 to 80.5 km/hr, and High risk = < 40.2 km/hr (Fiedler *et al* 2004).

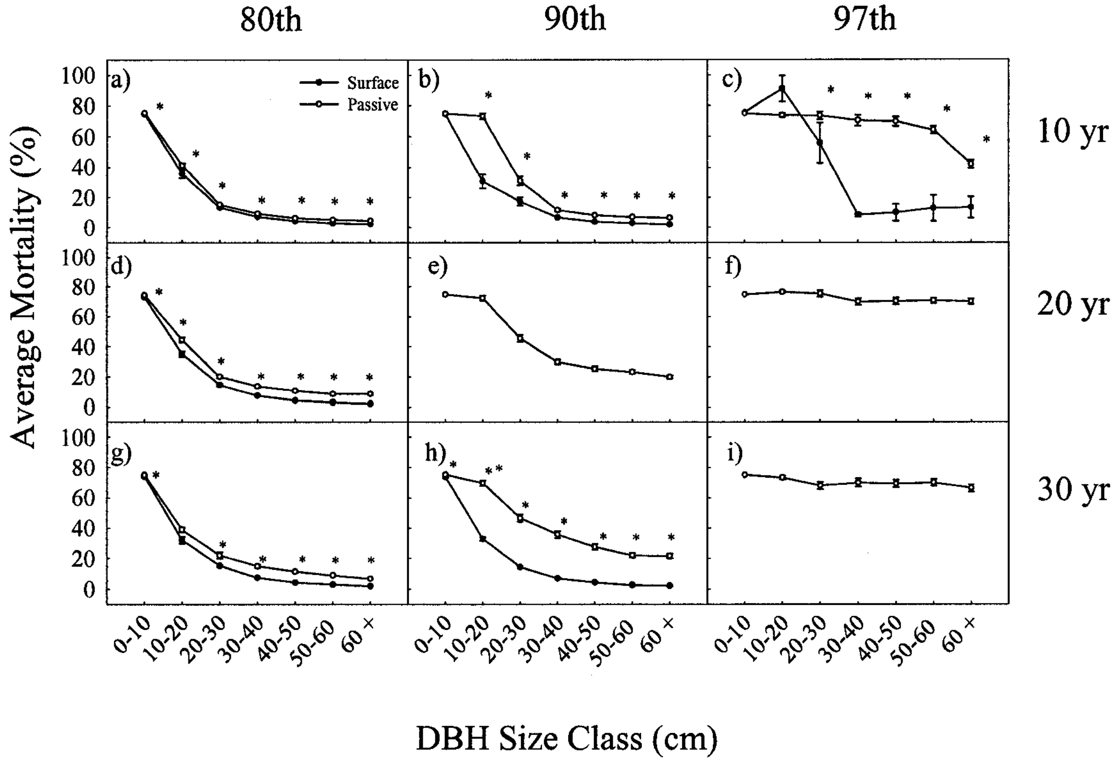

Figure 4.3: Potential mortality (± standard error of the mean) by fire type (surface, passive, or active crown fire) of historic Black Hills ponderosa pine forests for stands developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions as predicted by FFE-FVS. Significant differences ($p<0.05$) of mortality among dbh size class for each fire behavior type within a fire return interval/weather scenario are designated by an asterisk.

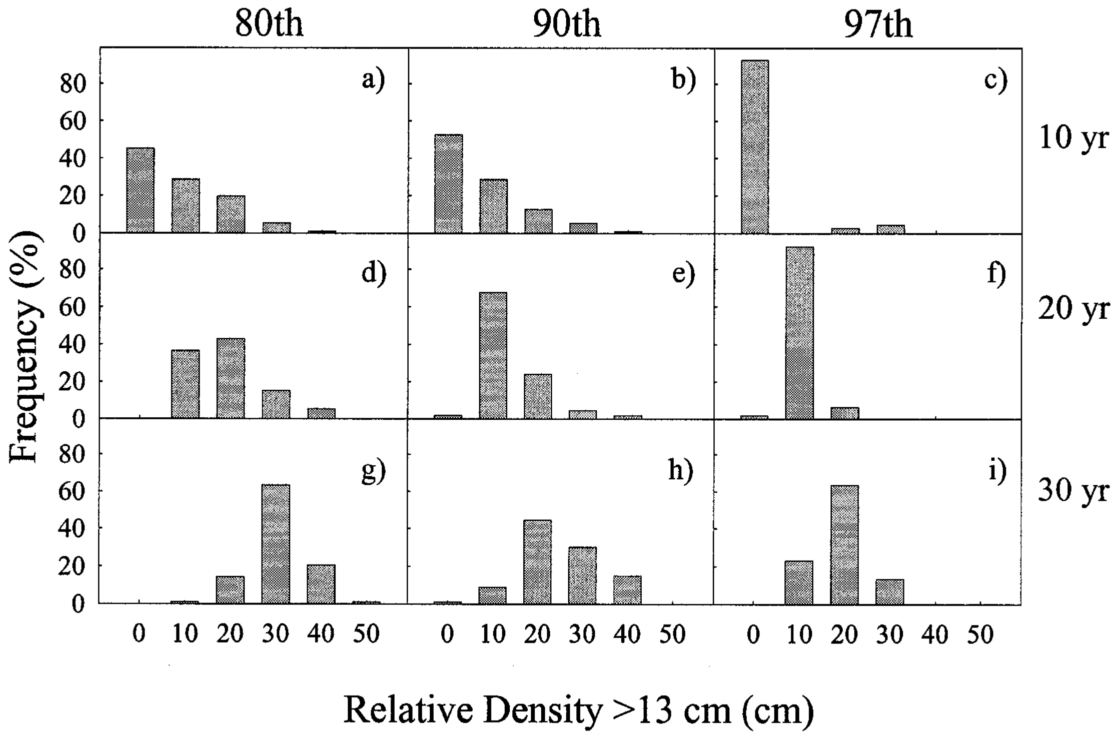

Figure 4.4: Frequency distribution for the average relative density of trees>13 cm for the last 60 years of the 100-year simulation of historic Black Hills ponderosa pine forest stands (n=112) developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions.

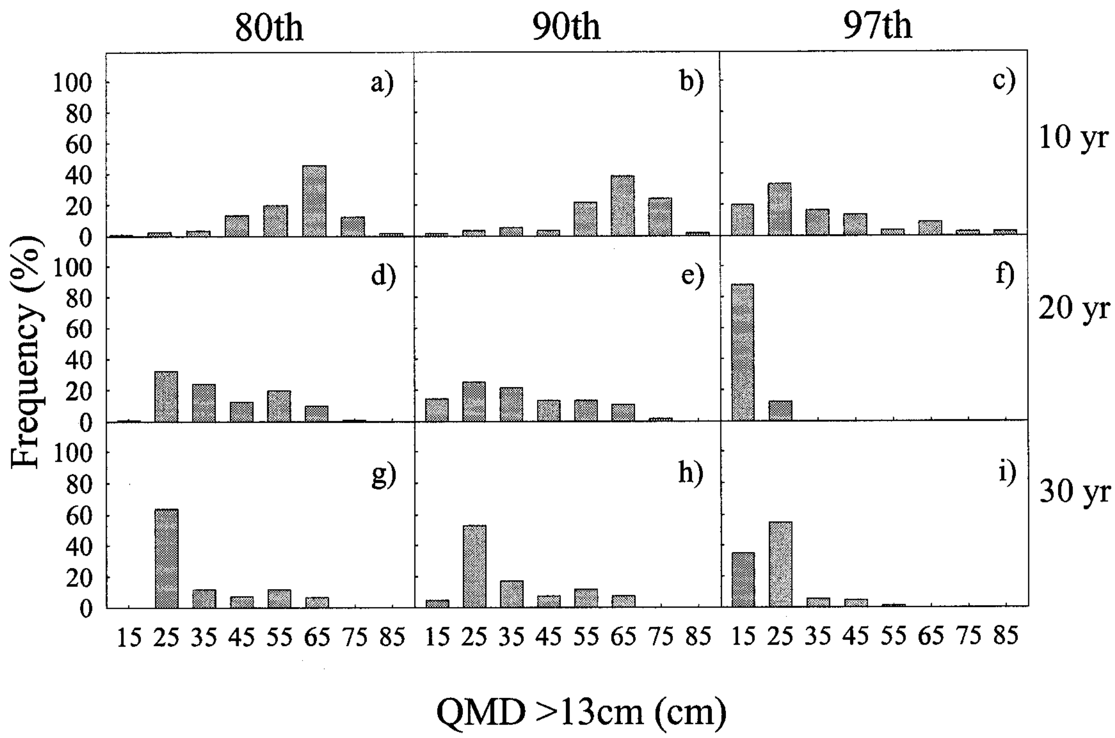

Figure 4.5: Frequency distribution for the average quadratic mean diameter of trees>13 cm for the last 60 years of the100-year simulation of historic Black Hills ponderosa pine forest stands (n=112) developing with a 10-yr, 20-yr, and 30-yr fire return interval under 80th, 90th, and 97th percentile weather conditions.

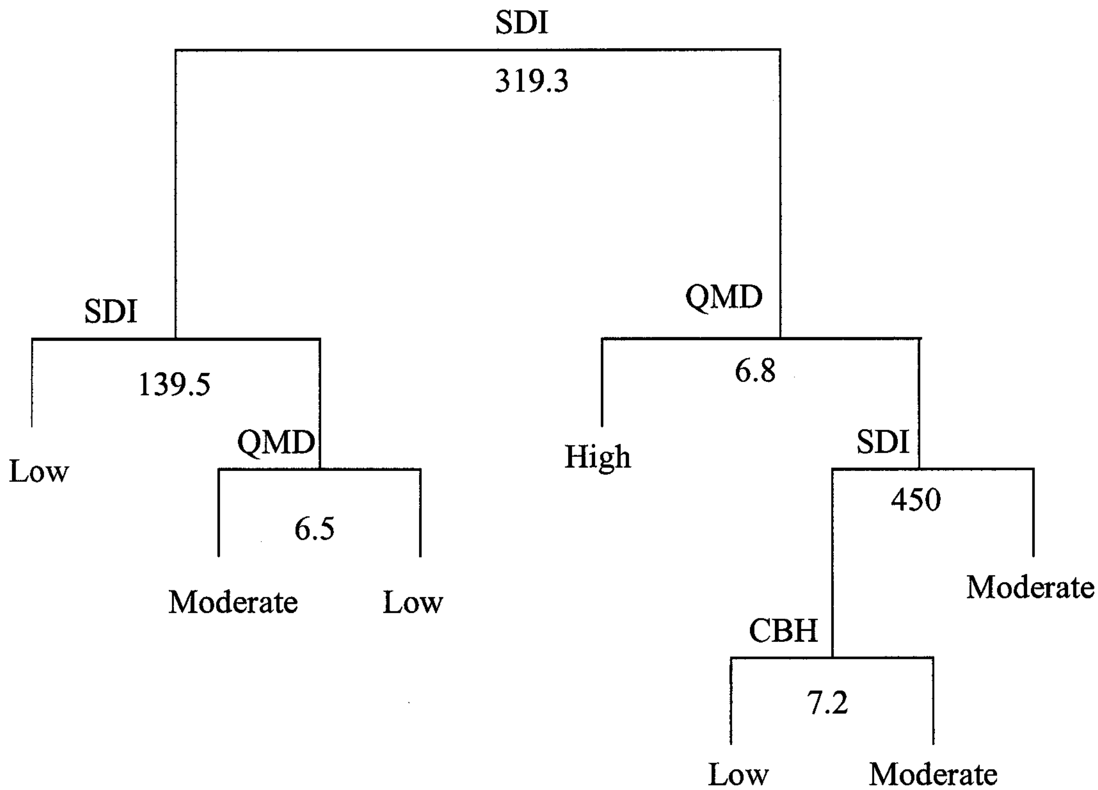

Figure 4.6: Classification tree for estimating risk of active crown fire based on stand attributes of historic Black Hills ponderosa pine forests under 97th percentile weather conditions. SDI is the Stand density index and is defined as the number of trees per hectare stands would have if the trees had an average size of 25.4 cm DBH. CBH = canopy base height (m), QMD = quadratic mean diameter (cm), Low = low active crown fire risk (>80.5 km/hr), Moderate = moderate active crown fire risk (40.2 to 80.5 km/hr), and High = high active crown fire risk (<40.2 km/hr). Values greater than the presented value classify to the right, lesser values classify to the left.

Acknowledgements

This research was supported by Joint Venture Agreement #03-JV-11221616-232 with the USFS Rocky Mountain Research Station, National Fire Plan project 01.RMS.C.4, and by McIntire-Stennis funding to the Colorado State University Department of Forest, Rangeland, and Watershed Stewardship. The authors would like to thank Chad Keyser and Stephanie Rebain of the USDA Forest Service, Washington Office, Forest Management group for their advice and help with the Forest Vegetation Simulator. We also thank Rudy King for statistical consultations.

CHAPTER 5: CONCLUSIONS

The results of my dissertation provide a scientific basis for the development and maintenance of fuel-reduction treatments in ponderosa pine forests of the Black Hills.

In **Chapter 2**, I successfully developed models that predicted seedling and sapling mortality based on fire related damage variables. I identified fire-related damage thresholds that were associated with mortality for each regeneration size class. These models also provide evidence that mortality of ponderosa pine seedlings and saplings is a result of cumulative injuries to the crown, roots, and cambium. The use of these models will aid managers in determining if treatment objectives were met.

Traditionally, prescribed burns are written with the premise of limiting mortality. However, the use of prescribed burn to maintain low densities will require managers to plan burns that limit survival of small trees while limiting mortality of larger trees. The model developed in **Chapter 2** that predicts seedling mortality based on associated flame lengths should aid managers in the planning stages of prescribed burns. This model provides benchmark flame lengths that are required to increase the probability of mortality based on seedling height. A similar model for saplings is currently in development, but plots were recently burned in Fall 2006 and a growing season is needed to elapse before the data is ready.

In **Chapter 3**, I used the models developed in **Chapter 2** to examine the temporal susceptibility of ponderosa pine seedling and saplings to prescribed fire in an effort to maintain fuel treatments over time. Based on the relationship between seedling/sapling growth, fuel accumulation, and potential prescribed fire behavior, the use of fire to maintain low regeneration densities in fuel treatments will require some careful planning. This study indicated that if ponderosa pine regeneration is not controlled, the effectiveness of a fuel treatment is diminished within a couple of decades. As time elapses, the new regeneration cohort grows to sizes that require greater fire-related damage to cause mortality. However, the amount of fuel available to burn under typical prescribed fire conditions does not produce the required flame lengths to cause mortality in the sapling-sized ponderosa pine trees. Current prescribed fire prescriptions and fuel loads are adequate to maintain low densities of ponderosa pine regeneration if burns occur every 10 to 15 years. However, if managers wait longer than 15 years between burns, regeneration can attain sizes that require weather conditions that will allow coarse woody debris to contribute to the fire intensity. Alternatively, managers can augment sites with activity fuels to increase fine fuel loadings to achieve flame lengths that will increase sapling susceptibility to fire, but burn when coarse woody debris (CWD) fuel moisture are high to limit the potential for spotting. While the flames might kill sapling-sized trees, mature overstory trees would still maintain some resiliency to fire.

Data for a study that examines the relationship between fuel loadings, fire behavior, and sapling mortality is pending. Unfortunately, as with most prescribed fire research studies, the researcher is at the mercy of the burn window. Although that study was installed in the summer of 2005, only one of the two blocks has been burned by the Fall of 2006. Data collected this summer should yield some useful information for the site burned in Fall of 2006. The other study site will hopefully burn in the Fall of 2007.

In **Chapter 4**, the simulation results of the potential fire behavior of historical ponderosa pine forests of the Black Hills confirmed the importance of maintaining low densities of ponderosa pine regeneration. The simulations indicated that stands which burned every 10 years killed the ponderosa pine regeneration before it could establish, resulting in stands with high canopy base heights and reduced probability of passive crown fire initiation. In contrast, stands that were modeled to burned every 20 or 30 years would allow time for ponderosa pine regeneration to reach sizes and densities that perpetuated the passive crown fire cycle. These results are consistent with the findings in **Chapter 3**, which shows that fuel treatment effectiveness would diminish within 20 to 30 years if regeneration is not controlled.

Chapter 4 also addressed the issue of historical fire behavior in Black Hills ponderosa pine forests. The simulations substantiated several lines of evidence which suggest that the historical Black Hills landscape was shaped by a mixed-severity fire regime which created a landscape mosaic of multi-cohort patches of various densities and structure types. This indicates that the restoration of the full suite of historical Black Hills forest structures would not always result in reduced crown fire risk and mortality. With a classification tree analysis, I was able to identify some historical stand structure attributes that would be compatible with fuel-reduction treatment objectives. Again, these fuel treatments would need to maintain low densities of ponderosa pine regeneration to prevent crown fire initiation.

The overall goal of my dissertation was to provide managers with scientifically based information that would promote prescribed fire as a tool to maintain fuel treatments in Black Hills ponderosa pine forests. My conclusions suggest that ponderosa pine regeneration densities must be controlled to maintain fuel treatment effectiveness over time. Prescribed fires should occur frequently, every 10 to 15 years, in order to effectively reduce seedling densities. If longer intervals are used, fires will need to be more intense in order to reduce the densities of the sapling-sized trees. Restoration of some structural elements of historical forest structure is compatible with fuel treatment objectives, but it will still be important to control ponderosa pine regeneration.