dNBR IMAGERY AND XERIC PINE-OAK FOREST STAND CHARACTERISTICS FOR FIRES OF DIFFERENT SEVERITY IN GREAT SMOKY MOUNTAINS NATIONAL PARK

A thesis submitted to the faculty of the Graduate School of
Western Carolina University in partial fulfillment of the
requirements for the degree of Master of Science in Biology

By

Scott A. Abla

Director: Dr. Laura E. DeWald, Biology

Committee Members: Dr. Thomas Martin, Biology
Dr. Peter Bates, Natural Resource Conservation and Management

October 2014

ACKNOWLEDGEMENTS

This study would not have been possible without the guidance and financial help from several organizations and academic institutions. I would like to extend my sincere gratitude and thanks to the Great Smoky Mountains National Park, specifically Robert Klein, Fire Ecologist, who was fundamental in providing logistical support, field training, and necessary equipment to ensure my safety during data collection. I would also like to thank Western Carolina University for providing me with resources and financial support during this project and allowing me to be a part of a great academic community. My committee members were essential in providing me with insight and guidance throughout my master's experience and I would like to thank Dr. Peter Bates and Dr. Thomas Martin for their advice, inspiring questions, and valuable time. I would not have had a remote chance of completing this project without the undying motivation and guidance from my advisor, my school mother, my friend, Dr. Laura E. DeWald.

My family has supported me throughout this journey and a very special thanks goes out to them, especially my mother Jill Abla, my father Steve Abla, and my brother Jonathan Abla. My wife Jennifer was with me every step from trekking through impassable vegetation of the most rugged mountains and then by my side sipping coffee through countless hours of statistical analysis.

I love and thank you all.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Figure Page

PREFACE

This thesis is organized in manuscript format. Chapter 1 is an introduction focusing on the relevance of my project to the southern Appalachian region and Great Smoky Mountains National Park. Chapter 2 is a review of scientific literature related to the topics in Chapters 3 and 4. Chapter 3 is a manuscript focusing on predicting burn severity patterns using a remote sensing tool. Chapter 4 is a manuscript focusing on vegetation responses following multi-severity fires. Chapter 5 is written as a synthesis of Chapters 3 and 4. Literature Cited is for all chapters.

ABSTRACT

dNBR IMAGERY AND XERIC PINE-OAK FOREST STAND CHARACTERISTICS FOR FIRES OF DIFFERENT SEVERITY IN GREAT SMOKY MOUNTAINS NATIONAL PARK

Scott A. Abla

Western Carolina University (October 2014)

Director: Dr. Laura E. DeWald

Fire suppression has changed forest structure and composition on xeric sites in the southern Appalachians from open, pine-oak dominated stands to closed canopy, mixed hardwood stands. Improved understanding of fire-related tools and ecological responses will improve effectiveness of fire management aimed at restoring pre-fire suppression forest communities on these xeric sites. Although occurrence of fire is known to be related to ecosystem functioning, vegetation responses to *multi-severity* fires are not as well understood in the southern Appalachians. Additionally, the relationship between satellite imagery and ground-based methods for designating burn severity (post-fire term describing fire severity) are not established for the Great Smoky Mountains National Park (GSMNP). The purpose of my study was to (1) determine if burn severity designations were consistent between satellite imagery and ground-based methods, and (2) evaluate vegetation responses to different burn severities on xeric sites dominated by pine (*Pinus*) and oak (*Quercus*) species in the GSMNP.

Plots were randomly located using satellite-based (dNBR) burn severity maps. For part (1) of my study these sites were ground-truthed using the FIREMON Composite Burn Index (CBI). Initial scatter plots between CBI and dNBR indicated a saturated growth relationship and square-root transformed dNBR data were overall strongly correlated to ground-based ratings (CBI) for 169 total plots ($p<0.001$, $R^2=0.90$). Strong relationships were found between CBI and dNBR across different xeric forest types and time since burn categories. For part (2) of my study, variables related to stand regeneration were measured at the ground, mid-story, and overstory layers across different burn severities for 48 plots. Differences in post-fire forest structure and composition across burn severity classifications were tested using analyses of variance and relationships between stand variables were evaluated using linear regression. Results showed overstory mortality was significantly higher in moderate and high severity sites versus low severity and no burn sites. Stand density and basal area were lowest in high severity sites and litter layer depth decreased significantly in higher severity fires. Pine regeneration did not vary across burn severities and oak regeneration was highest in moderate severity sites. Mixed mesophytic regeneration was highest in sites absent of fire. Desired pine and oak regeneration was greatest in moderate burn severity sites. Changes in species composition following fire may have been caused by greater amount of exposed mineral soil, increased light penetration to forest floor, and reduced mid-story stem densities. Overall results from both studies show that (1) burn severity can be predicted from satellite imagery and (2) different burn severities are

associated with different forest structure and composition related to pine and oak regeneration on xeric sites in GSMNP.

CHAPTER 1: INTRODUCTION

Open canopy woodlands dominated by fire-adapted trees were historically very common in many areas across North America, and fire likely played an important role in the creation and maintenance of open, shade-intolerant species dominated forests in the southern Appalachians (Harrod et al. 1998; 2000). Active fire suppression policies and alterations in forest management disturbances in the early 20th century led to changes in historical ecosystem structure and reduced wildfire frequency, but increased intensity, increased forest density, and reduced forest understory richness and productivity (Ware et al. 1993, Harrod et al. 2000). Additional changes in disturbance regimes are expected in the 21st century as a consequence of Global Change (IPCC, 2007), particularly changes in climate (Schumacher and Bugmann 2006). During this period of changing climate, developing a better understanding of the temporal and spatial connections between different ecological responses to fire will help resource managers plan, direct, and monitor short- and long-term effects of various fire management activities aimed at restoration (Skinner et al. 2008).

The short- and long-term effects fire has on forest communities can be evaluated at different spatial scales, such as the rates and processes of ecological succession and encroachment which are influenced by different fire severities (Lentile et al. 2006). The term ‘burn severity’ is defined as the degree of environmental change caused by fire (Key and Benson 2006) and is used in my study to describe post-fire effects on vegetation (DeBano et al. 1998). Burn severity expressed as ratings can be related to ecosystem responses to fire in terms of regeneration, moisture levels, and soil erosion (Keeley et al. 2008). Although numerous reports in the literature indicate fire has a significant impact on ecosystem functioning, it is less clear the extent that burn severity controls these responses (Keeley et al. 2008), particularly following relatively small fires (wild or prescribed). Improving our ability to predict ecosystem responses to wildfires and prescribed fires will help improve the effectiveness of post-fire management planning. Creation of accurate burn severity maps based on past fires will allow managers to more efficiently identify and predict environments that are more susceptible to high-severity fires (Wimberly and Reilly 2006). An important prediction tool is mapping fires in a way that can predict ecological change post- fire. Satellite remote sensing is an effective tool for these efforts (Key and Benson 2002).

Over the past 15 years there have been many small fires (<1000 ha) in the Great Smoky Mountain National Park (GSMNP) (Robert Klein, Fire Ecologist, GSMNP, Personal Communication). These wildfires and prescribed fires occurred throughout different ecotypes that typify the different moisture gradients in the Park (Harrod et al. 1998). Moisture gradients and subsequent forest ecotypic and canopy cover variation are related to southwest-northeast trending ridges with steep and highly dissected topography (Harrod et al. 1998, Harmon 1982). In 1940 when the park began suppressing wildfires, the mean fire return interval increased from less than 20 years to over 500 years (Harmon 1982) resulting in changes in canopy structure and composition that otherwise would not have occurred (Harrod et al. 2000). For example, since the 1930's the mean density and basal area of canopy trees in the Park has doubled. During this period and across the eastern US, light-demanding fire-resistant genera such as pine (*Pinus)* and oak (*Quercus)* have declined and more shade-tolerant species such as red maple (*Acer rubrum* L.) have increased (Harrod et al. 1998, Harrod and White 1999, Nowacki and Abrams 2008) making it difficult for managers to predict post-fire regeneration densities, particularly with small fires (less than 1000 ha) of varying severity.

In the pine forests of the semi-arid western U.S., prediction models such as LANDFIRE can predict responses to burn severity (Key and Benson 2006). Wimberly et al. (2009) suggest this type of approach could be adapted to eastern ecosystems, but research to validate these methods has been limited to only a few studies. The purpose of my study was to evaluate the use of one method of determining burn severity using Landsat imagery in GSMNP and to evaluate post-fire differences between sites of varying burn severity. In GSMNP, pine and oak dominated stands are diminishing and restoration of the xeric sites that support these communities are a major concern for Park managers. Park managers have used fire as a tool to restore these sites, but studies on the effects of varying burn severity are lacking. Overall results from my study will help Park managers better understand the relationships between fire, severity, and vegetation responses in GSMNP and allow them to use fire more effectively as a tool to restore xeric sites across the Park.

CHAPTER 2: LITERATURE REVIEW

Forests of the Great Smoky Mountains National Park

The Great Smoky Mountains National Park (GSMNP) is located in the southern Appalachian Mountains, which are characterized by complex topography, elevations ranging from 600 to 2000 meters, and highly variable microclimates resulting in a wide range of biological community types (Waldron et al. 2007, Jenkins 2007, Reilly et al. 2012). The community types in GSMNP defined by Whitaker (1956) are based on species associations occurring on a moisture gradient (Callaway et al. 1987). However, community type classifications differ throughout the literature regarding forested communities in GSMNP (Whitaker 1956, Harmon 1984, White 1998, Madden et al. 2004, Jenkins 2007, Jenkins et al. 2011). These differences are centered on sub-classifications within larger species associations, and studies reporting differences between dominant species on the same types of sites are partially explained by disturbances introduced throughout the 20th century. For example, the first known data set from a vegetation study in GSMNP (Miller 1938) defines eight forested community types including two community types dominated by American chestnut (*Castanea dentata* Marshall [Borkh]), but this species was removed from the forest canopies throughout its range by the mid 1950s due to chestnut blight (*Cryphonectria parasitica* [Murrill] Barr) (Mackenzie and White 1998, Whiteaker 1956). Several other studies in the 1980s (e.g. Golden 1981, Callaway et al. 1987) focused on defining community types in western areas of

GSMNP that were not adequately sampled by Whitaker (MacKenzie and White 1998). In 2004 Madden et al. published a digital vegetation map of GSMNP defining 11 different forested overstory vegetation types that was based on photographs, and Jenkins (2007), later defined eight different forested community types using remotely-sensed data based on multivariate factors. Despite differences in community type classifications, abiotic site characteristics of the different community types are consistently described across all studies (Miller 1938, Whitaker 1956, Callaway et al. 1987, MacKenzie and White 1998, Madden et al. 2004, Jenkins 2007). For the purpose of this thesis, community types will be described using site characteristics, such as moisture or elevation, based on information reported by Whitaker (1956) and Jenkins (2007).

The eight forested community types for GSMNP described by Jenkins (2007) are: montane alluvial, early successional, cove, hemlock, montane oak-hickory, xeric ridge, high-elevation hardwood, and spruce-fir forests. These community types range in moisture class from mesic sites, high moisture sites typified by cove forests, to xeric and, low moisture sites typified by pine (*Pinus*) forests (Madden et. al 2004, Whitaker 1956). The vegetation patterns also follow elevation gradients. Mixed-mesic hardwood forests dominate bottomlands and slopes in low to mid elevations, pine-oak (*Pinus-Quercus*) forests dominate xeric ridges and upper slopes in low to mid elevations, while spruce-fir (*Picea-Abies)* forests dominate the highest elevations (> 1600 m) (Whitaker 1956, Callaway et al. 1987, Jenkins 2007). For this thesis, the terms ‘mesic’ and ‘xeric’ will be used as groups to categorize species associations found on each of the different sites for individual community types described by Whitaker (1956) and Jenkins (2007). For example, sites dominated by shade-intolerant pine and oak species would be considered xeric, while sites dominated by shade-tolerant species, such as maple or birch, would be considered mesic.

Structure and composition of forests depends on how mortality from disturbances and post-disturbance regeneration interact for given site. Trends in disturbance occurrences drive forest succession in eastern deciduous forests including the southern Appalachians (Runkle 1982, Shure and Wilson 1993) and the variation in different disturbance intensities results in high forest diversity in the southern Appalachians (Beckage and Clark 2003).

Forest Communities on Mesic Sites in GSMNP

Jenkins (2007) reported approximately 75% of forests across GSMNP occur on mesic sites ranging from distinctly mesic cove communities (acid cove by Madden et al. 2004) to less mesic montane-oak hickory (*Quercus-Carya*) community types (Whitaker 1956). Mesic sites throughout GSMNP are typically found on north-to-east slopes and bottomlands and are dominated by dense stands of shade-tolerant late successional broadleaf species such as American beech (*Fagus grandifolia* Ehrh.*)* and northern red oak (*Quercus rubra* L.) (Whitaker 1956). The cool, humid microclimates of these sites are maintained by limited airflow, high moisture retention in thick litter layers, and limited solar radiation to the forest floor due to dense canopies (Nowacki and Abrams 2008, Whitaker 1956). Canopy dominants of these sites include American beech (*Fagus grandifolia),* tulip-poplar *(Liriodendron tulipifera* L.*),* mockernut hickory *(Carya tomentosa* Sarg.*),* sugar maple *(Acer saccharum* Marshall*),* yellow birch *(Betula allegheniensis* Britt.*),* northern red oak*,* American basswood *(Tilia Americana var. heterophylla* Vent.*),* yellow buckeye *(Aesculus flava* Sol.*)*, and more recently as a result of anthropogenic disturbances, eastern white pine (*Pinus strobus* L.*)* (Whitaker 1956, Harrod et al. 2000). Mid-stories of mesic sites in GSMNP are often dominated by flowering dogwood (*Cornus florida* L.*),* pignut hickory *(Carya glabra* Miller*),* red maple *(Acer rubrum* L.*),* downy serviceberry *(Amelanchier arborea* [F.Michx.] Fernald*)*, and shrub layers often include rhododendron (*Rhododendron maximum* L.*),* viburnum *(Virburnum acerifolium* L.*),* and spicebush *(Lindera benzoin* [Boerh.] Schaeff.*)* (Whitaker 1956, Jenkins 2007).

Forest communities occurring on mesic sites in GSMNP change following disturbances which range in intensity from single tree deaths to large-scale stand mortality from wind or fire, although the latter is much less frequent for these sites due to high humidity (Runkle 1982, McGrath and Clatterbuck 2013). In the southern Appalachians early forest succession is driven by plant competition for available resources and shade-tolerant species dominate following small-scale disturbances while early successional intolerant species are more competitive following large-scale disturbances (Shure and Wilson 1993, Runkle 1982, McGrath and Clatterbuck 2008). Typically, shade-tolerant tree species regenerate and close small canopy gaps created during small-scale disturbances while regeneration of intolerant species, such as tulip-poplar and black locust *(Robinia pseudoacacia* L.*)*, have a competitive advantage in more open gaps created during larger-scale disturbances that create early successional habitat (Runkle 1982, Phillips and Shure 1990, Shure and Wilson 1993). Shure and Wilson (1993) concluded that high levels of nutrients released during the creation of large forest openings in the southern Appalachians may favor the regeneration of some species over others.

Forest Communities on Xeric Sites in GSMNP

Xeric sites occupy between 16-24% of the forested land in GSMNP and typically occur on south- to west-facing slopes and ridges (Madden et al. 2004, Jenkins 2007). These sites are typified by shallow, acidic soils and relatively drier, warmer microclimates (Whitaker 1956, Jenkins 2007). Pine and oak species dominate the overstory in these sites and vegetation classifications range from xeric pine woodlands to xeric mixed-hardwood forests (Madden et al. 2004, Whitaker 1956). Jenkins (2007) defines only one xeric community type which is the xeric ridge community that includes both dry-site pine and oak dominated sites. Historically, forests occurring on these sites were characterized by a relatively open understory consisting of a high component of grasses and a low density of trees (Elliot and Vose 2005, Delcourt and Delcourt 1997). These forests are now characterized by dense ericaceous (heath family) shrub layers and dense mid-story canopies (Jenkins et al. 2011, Reilly et al. 2012).

Yellow pine species found in the overstory of xeric sites in GSMNP are typically shade-intolerant pioneer species (Whitaker 1956) such as shortleaf pine (*Pinus echinata* P. Mill.), Virginia pine (*Pinus virginiana* Mill.*)*, pitch pine (*Pinus rigida* P. Mill.), and table mountain pine (*Pinus pungens* Lamb.) (Jenkins 2007). Table mountain pine only dominates in forests at relatively high elevations, which in GSMNP is limited to a few small areas (Jenkins et al. 2011, Welch et al. 2000). Compared to historical conditions mid-story components in yellow pine forests include increasingly dense stands of sourwood *(Oxydendrum arboreum* [L.] DC.*)*, black gum (*Nyssa sylvatica* Marsh.*),* and mountain laurel (*Kalmia latifolia* L.) (Whitaker 1956, Jenkins 2007, Welch et al. 2000). Dry oak sites typically occur on lower ridges and less-exposed slopes than yellow pine sites and are typically dominated by chestnut oak (*Quercus montana* L.*)*, white oak (*Quercus alba* L.*)*, black oak (*Quercus velutina* Lamb.), and scarlet oak (*Quercus coccinea* Muenchh.) (Jenkins 2007, Whitaker 1956). In addition to these species eastern white pine, historically absent from these sites, has encroached into these oak sites as a result of land-use change during the early 20th century (Nowacki and Abrams 2008, Jenkins 2007, Whitaker 1956). Previously open mid-story canopies in these sites are now typically dense thickets of rhododendron and mountain laurel (Jenkins et al. 2011, Whitaker 1956, Welch 2000). Mixed pine-oak forests are the most common forest type on xeric sites throughout GSMNP, however, pine species dominate the drier xeric sites while oak species dominate the moister xeric sites (Jenkins et al. 2011, Whitaker 1956, Madden et al. 2004, Waldron et al. 2007).

Successful regeneration of dry-site pine and oak species requires high amounts of solar radiation created by canopy openings and thus, the structure and composition of pine-oak forests are maintained by disturbances (Waldron et al. 2007, Jenkins et al. 2011, Phillips and Shure 1990) such as wind, ice, wildfire, and insect outbreaks of varying intensity, and range in scale from individual trees to thousands of hectares (Nowacki and Abrams 2008, Dumas et al. 2007, Waldron et al. 2007). Species occurring on xeric sites are adapted to the more frequent fire disturbances. Several studies (Waldrop and Brose 1999, Jenkins et al. 2011) reported yellow pine regeneration was absent in unburned and burned sites with less than 80% canopy over. The thick bark of species such as chestnut oak and pitch pine allows them to survive fire and resulted in a dependence on a fire frequency of less than 20 years for successful regeneration (Harmon 1982; 1984, Harrod et al. 1998, Nowacki and Abrams 2008, Waldron et al. 2007). Advanced regeneration (large seedlings and saplings) and stump sprouting by oak species in GSMNP are favored following disturbances such as wind and ice storms (Harrod et al. 1998). Ability to stump sprout also allows oak species to outcompete other species during stem-exclusion stages of stand development due to their established root system (Atwood et al. 2009, Welch et al. 2000). A mixture of small- and large-scale disturbances maintains these dry-site pine and oak early successional forest communities in the stand initiation and stem-exclusion stages of stand development as described by Phillips and Shure (1990).

It is important to note that oak regeneration can be outcompeted both by shade-tolerant species in closed canopy forests and by shade-intolerant species in open canopy forests (Iverson et al. 2008, Nowacki and Abrams 2008). Therefore, the size of the canopy openings is critical to successful oak regeneration. Although Rentch et al. (2003) found that fire disturbance in oak forests favored oak regeneration over other understory competitors, several other studies reported mixed results on the effects of gap openings from fire or other silvicultural treaments on oak regeneration on xeric sites (Iverson et al. 2008, Elliot et al. 1997, Shure and Wilson 1993, Atwood et al. 2009). Some studies have noted that interactions of disturbances occurring on xeric sites results in different communities in GSMNP (Jenkins et al. 2011, Waldron et al. 2007). For example, wind/ice storms and southern pine beetle (*Dendroctonus frontalis* Zimmermann) outbreaks favor oak regeneration by creating relatively smaller gap openings, but when wildfire also occurs, the yellow pine seedlings are favored by the larger canopy openings (Waldron et al. 2007, Jenkins et al. 2011). In summary, the perpetuation of pine-oak forests on xeric sites in GSMNP depends on multiple disturbances acting together to create a mosaic of open and closed canopies that are dominated by pine and oak species.

Fire in the Great Smoky Mountains National Park

Effects of Multi-Severity Fire on Vegetation

Effects of fire on forest communities in GSMNP have been reported in several studies, but studies evaluating specific effects of fire due to different severities (burn severity) are lacking (Lentile et al. 2006, Keeley et al. 2008, Hubbard et al. 2004, Jenkins et al. 2011). Elliot and Vose (2005) reported low severity fires have little effect on plant community composition in the Southern Appalachians. Dumas et al. (2007) and Reilly et al. (2006) both found species composition following fire was a result of site-specific variation in fire severity. Jenkins et al. (2011) reported vegetation responses to wildfire varied as intensity of fire increased due to the various mechanisms through which fire severity acts upon the landscape. Site conditions that favor yellow pine and oak regeneration are consistent with post-fire conditions resulting high light availability to the forest floor, relatively low understory and overstory densities, and litter layer depths less than four centimeters (Jenkins et al. 2011, Welch et al. 2000, Nowacki and Abrams 2008, Brose et al. 2013), although rapid sprouting of shrub and hardwood species following high severity fire often creates a shaded environment where pine and oak species cannot regenerate (Nowacki and Abrams 2008, Harrod et al. 1998).

Specific site changes due to fire vary depending on fire severity (Brown et al. 2014). Wildfire impacts soil properties and nutrient cycles through plant mortality, modifying the availability and uptake of nutrients, and creating differences in moisture-holding capacity (Stephens et al. 2012). Fire can significantly alter carbon and nitrogen cycling by combusting stored carbon and temporarily increasing nitrogen availability through ash deposition, regeneration of nitrogen-fixing vegetation, and low plant-uptake (Hubbard et al. 2004, Vose et al. 1999, Deluca et al. 2006, Brown et al. 2014, Stephens et al. 2012). Many studies reported low and moderate severity fires have minimal effects on nutrient cycling, soil processes, and characteristics such as pH, exchangeable cations, soil bulk density, soil carbon, dead-wood carbon, and soil nitrogen (Brown et al. 2014, Stephens et al. 2012, Hubbard et al. 2004, Vose et al. 1999). Effects of high intensity fires on nutrient cycling and soil processes in the Southern Appalachians are unknown.

Fire History in GSMNP

Nowacki and Abrams (2008) describe Native Americans as managing land with fire for thousands of years and Lafon et al. (2005). Delcourt and Delcourt (1997) examined fossil pollen and charcoal particle records over the past 4000 years from the Horse Cove Bog in Highlands, NC and concluded fire-adapted species such as yellow pine and oak dominated the forests that were maintained by human- and lightening-caused fires (Fesenmyer and Christensen 2010). Low intensity frequent fires were used to manage hunting lands, increase soil fertility, and reduce the risk of stand replacing fires (Delcourt and Delcourt 1997, Dumas et al. 2007). As a result of prescribed burning by Native Americans and wildfire ignited by lightening, forests in the eastern United States were more open and grassy, and dominated by fire-adapted shade-intolerant early successional species such as oak, American chestnut, hickory, and yellow pine species (Delcourt and Delcourt 1997, Elliot and Vose 2005, Lafon et al. 2005, Fesenmyer and Christensen 2010, Hessl et al. 2011, Nowacki and Abrams 2008, Harrod et al. 2000). Fesenmyer and Christensen (2010) found that some xeric slopes and ridge-tops are currently dominated by mesic hardwood ecosystems. This suggests that fire might have suppressed the migration of more mesophytic hardwoods into xeric sites.

In the 17th century, European settlement brought with it changes in land use, forest management, and ignition patterns that led to changes in the southern Appalachians fire regime (Fesenmyer and Christensen 2010, Harmon 1984). European settlers increased access to forests, cleared land, and altered forest fuel conditions resulting in an increase in fire size, frequency, and intensity in some areas (Hessl et al. 2011, Fesenmyer and Christensen 2010). Land-use patterns of burning large areas of forestland for grazing/hunting did not slow down until the early 1900s with the abandonment of farmland, decline of the forest dominant species American chestnut, extensive timber harvesting, landscape fragmentation, and the onset of active fire suppression (Hessl et al. 2011, Fesenmyer and Christensen 2010, Nowacki and Abrams 2008, Dumas et al. 2007, Harmon 1982). This land-use change and fire suppression policies of the early 1900s resulted in a change in forest structure and composition from the open-canopy pine-oak communities to closed-canopy shade-tolerant dominants

(Fesenmyer and Christensen 2010, Nowacki and Abrams 2008). Since the onset of fire suppression in the 1930's in GSMNP, density and basal area of canopy trees on xeric sites within the Park have doubled and light-demanding fire-adapted genera such as *Pinus* and *Quercus* have declined while more shade-tolerant species such as red maple and black gum have increased (Nowacki and Abrams 2008, Harrod and White 1999, Harrod et al. 1998).

Fire Management in GSMNP

Although suppression of wildfire is still practiced across the US, prescribed burning is increasing in an attempt to reduce forest fuels and promote restoration of fire-adapted ecosystems. Many studies have shown that prescribed fire can be used to restore xeric site pine and oak ecosystems in the southern Appalachians (Elliot and Vose 2005, Jenkins et al. 2011, Arthur et al. 1998, Harrod et al. 2000, Chiang et al. 2005). Chiang et al. (2005) found significant oak seedling regeneration following prescribed burning, but this initial response was surpassed by the rapid flush of maple stump sprouts in the following seasons. In fact, pine and oak can be outcompeted by post-fire sprouting of hardwoods and ericaceous shrubs that result in higher sapling densities than pre-fire counts (Waldrop et al. 2008). Some studies indicate that for oak and pine regeneration to be successful, repeated prescribed burns or the creation of larger gaps from high severity fire are needed (Arthur et al. 1998, Brose et al. 2013, Chiang et al 2005, Jenkins et al. 2011). Other studies reported reductions in mid-story tree

(>5 cm) density from a combination of mechanical thinning and prescribed fire was important for successful pine and oak regeneration (Reilly et al. 2012, Hutchinson et al. 2012). Desired results from prescribed fire in the southern Appalachian forests appears to vary both spatially and temporally, and undesired responses often occur including nonnative invasive species encroachment, suppression of pine-oak regeneration in low severity fires, and unwanted canopy mortality in high severity fires (Reilly et al. 2012, Jenkins et al. 2011, Harmon 1984).

In order to use prescribed fire more effectively, managers need to understand the relationship between ecosystem responses and burn severity. Key and Benson (2006) developed the Composite Burn Index (CBI) to rate on-the-ground burn severity based on vegetation responses in pine stands in the western United States. CBI uses visual field estimates of post-fire changes in forest structure and composition using up to 23 different burn severity variables in 30-meter plots. The result is an average burn severity rating for the plot ranging from 0.0 to 3.0, where 0.0 indicates no change due to fire and 3.0 indicates maximum measurable change post-fire (Key and Benson 2006). Many studies have validated CBI for assessing on-the-ground burn severity (Kasischke et al. 2008, Zhu et al. 2006, Lentile et al. 2006).

In addition to on-the-ground methods, satellite imagery has been used to map burn severity (e.g., Wimberly and Reilly 2006, Cocke et al. 2005, Lentile et al. 2006). Satellite imagery can detect fire-related changes in reflectance based on moisture gradients and changes in forest structure caused by fires of different intensity. Patterns of burn severity can be mapped using the different spectral signatures that occur following a fire. Cocke et al. (2005) concluded burn severity maps could be used to predict effects of fire if biological changes seen after that fire can be related to fire severity. Assessing ecological responses to fire, predicting changes in site conditions due to fire, and understanding how fire intensity and burn severity factor into these processes is becoming more attainable as remote sensing satellite imagery technology advances (Lentile et al. 2006)

Currently, the most common satellite imagery tool used to assess burn severity is the Differenced Normalized Burn Ratio (dNBR), which is the ratio of two Landsat satellite-based reflectance values obtained from images pre- and post-fire (Wimberly and Reilly, 2006). For example, light reflectance of an unburned site will differ greatly from the light reflectance of a burned site and this difference varies depending on burn severity. CBI was specifically designed to validate the post-fire dNBR spectral index for western conifer forests (Lentile et al. 2006) and several studies conducted in various other forest types throughout the eastern and western United States found strong relationships between field-based measures using CBI and dNBR (Cansler and McKenzie 2012, Wimberly et al. 2009, Cocke et al. 2005, Rollins et al. 2004). This relationship for small fires and/or highly diverse ecosystems is not as well described.

CHAPTER 3: MANUSCRIPT:

EVALUATING THE USE OF DIFFERENCED NORMALIZED BURN RATIO FOR PREDICTING BURN SEVERITY IN GREAT SMOKY MOUNTAINS NATIONAL PARK

Introduction

Understanding how vegetative patterns differ with changes in fire regimes would be improved by our ability to map, predict, and assess different ecological effects of fire at various scales and intensities (Wimberly and Reilly 2007, Lentile et al. 2006, Kasischke et al. 2008). Evaluating post-fire forest structure and composition and understanding how fire influences differences in vegetative patterns can be partially achieved by quantifying burn severity, which is defined as the total amount of environmental change due to fire (Key and Benson 2006). Lentile et al. (2006) found that both active fire characteristics and post-fire effects could be used to classify burn severity using a combination of field and remote sensing data. Maps depicting burn severity zones can be used to predict and monitor future post-fire effects and identify areas associated with social, ecological, and economical values at risk (Lentile et al. 2006). For example, Wimberly and Reilly (2007) used burn severity maps to identify potential areas for the post-fire restoration of *Pinus* and *Quercus* forests.

The composite burn index (CBI) is a burn severity index calculated from post-fire field measurements. CBI is a commonly used method for assessing and classifying on-the-ground burn severity (Kasischke et al. 2008, Zhu et al. 2006, Lentile et al. 2006). CBI combines direct effects of fire intensity such as char height and canopy mortality with indirect effects such as plant re-growth and colonizing species. These combined effects are measured across multiple layers to calculate an overall burn severity rating ranging from 0.0 to 3.0, where 0.0 indicates no change due to fire and 3.0 indicates significant change (Key and Benson 2006). Field characteristics are rated for five different layers including substrate, low vegetation and shrubs, tall shrubs, sub-canopy trees, and canopy trees (Kasischke et al. 2008, Key and Benson 2006). The CBI ratings correspond to burn severities of low, moderate, and high.

Multi-temporal satellite-based imagery is another tool commonly used to classify burn severity (Picotte and Robertson 2011, Soverel et al. 2010, Escuin et al. 2008, Murphy et al. 2008, Zhu et al. 2006, Epting and Sorbel 2005). The most common type uses the differenced normalized burn ratio (dNBR), which is a pixel value calculated from an algorithm using pre- and post-fire image comparisons of light reflectance to quantify burn severity (Key and Benson 2006). dNBR values are limited by the amount of light reflectance post-fire and range from approximately -1000 to +10000 where higher values indicate higher severity and negative values indicate enhanced growth of vegetation following fire (often associated with very low burn severity in some forest types) (Key and Benson 2006, Murphy et al. 2008). Based on trends in burn severity patterns, dNBR values have been delineated regionally into burn severity categories of no burn, low, moderate, and high to create burn severity maps (Key and Benson 2006). Lentile et al. (2006) and van Wagtendonk et al. (2004) suggest dNBR values need to be validated for a wide range of environments before they can be effectively used to predict local fire effects related to burn severity.

Although CBI was specifically developed to validate dNBR, results of studies from different regions testing the relationship between CBI and dNBR have reached differing conclusions regarding the strength of these tools (Wimberly et al. 2009, Cocke et al. 2005, Wimberly and Reilly 2007, Kasischke et al. 2008). Most of the studies showing a significant relationship between CBI and dNBR have used large fires in either western conifer forests or eastern coastal pine stands where one forest type dominates (Cocke et al. 2005, Zhu et al. 2006, Lentile et al. 2006, van Wagtendonk et al. 2004). In contrast, the relationship between CBI and dNBR is not as well documented for small fires, or for fires occurring in diverse forests such as these found in the southern Appalachian region. Wimberly and Reilly (2007) compared CBI and dNBR for one large fire in a xeric ecosystem in Linville Gorge, North Carolina, but their objective was to create broad descriptions of burn severity versus statistically comparing CBI to dNBR. Although Key and Benson (2006), Waldrop et al. (2013), and Miller et al. (2009a) suggest that CBI and dNBR can be used to assess burn severity up to 4 years post-fire, the CBI-dNBR relationship following different post-fire measurement intervals is also not documented for the southern Appalachian forests. Finally, the CBI-dNBR relationship has also not been well documented for different xeric forest types in the southern Appalachian region.

Using dNBR alone to classify burn severity has many advantages over using CBI due to the difficulty of collecting field data in post-fire forest stands (time, cost, and accessibility). Miller et al. (2009a) reported the accuracy of predicting CBI from dNBR increased as the amount of forested tree cover increased, possibly due to situations where changes due to fire are more prevalent in the overstory forest layer. Soverel et al. (2010) found that dNBR was better at estimating CBI than other Landsat-based (satellite imagery) metrics in the Northern Rocky Mountains, Canada. However, Murphy et al. (2008) found dNBR to be a poor predictor of CBI-based burn severity in Alaskan boreal forests.

One limitation of using dNBR to map burn severity is that changes in surface reflectance related to topography have the potential to alter dNBR independently of burn severity (Miller et al. 2008), which could limit the usefulness of dNBR in the highly variable topography of the southern Appalachians. In addition, the dNBR spectral response saturates in higher severity fires while CBI sensitivity in the higher severity sites decreases (French et al. 2008). Evaluating the relationship between dNBR and CBI across a range of factors including different forest types and length of time since fire of data collection will improve the accuracy of burn severity maps created by dNBR methods. Accurate dNBR burn severity maps could potentially be used to monitor and predict areas for restoration by characterizing different ecological effects related to burn severity (Cocke et al. 2005, Wimberly and Reilly 2007).

Objectives

The purpose of my research was to validate the use of satellite imagery (dNBR) to characterize ground-based burn severity using CBI for small fires (<1000 ha) in the Great Smoky Mountains National Park (GSMNP). This study tested the following prediction:

> there is a relationship between dNBR and CBI among forest types and among different post-fire times of data collection.

Methods

Study Area

The study took place in GSMNP, which encompasses 2072 km^2 (95% forested) in the southern Appalachian Mountains of North Carolina and Tennessee (Figure 3-1). Study sites were located on south- to west-facing slopes and ridges up to 1000 meters in elevation characterized by warm and dry conditions, with rocky and highly weathered shallow, loamy, soils over sandstone, siltstone, and shale bedrock (Jenkins et al. 2011, Whitaker 1956). Historical forest structure on these xeric sites was open, low-density woodlands with conditions suitable for yellow pine and dry-site oak species including pitch pine (*Pinus rigida* Mill.), shortleaf pine (*Pinus echinata* Mill.), Virginia pine (*Pinus*

Fig. 3-1. Location of study in Great Smoky Mountains National Park in North Carolina and Tennessee, USA.

virginiana Mill.), chestnut oak (*Quercus montana* L.), scarlet oak (*Quercus coccinea* Munchh.),and white oak (*Quercus alba L.*) species (Whitaker 1956, Harrod et al. 2000, Delcourt and Delcourt 1997, Jenkins et al. 2011). The four xeric forest types in GSMNP studied for this research include yellow pine, chestnut oak, oak-hickory, and eastern white pine (*Pinus strobus* L.) xeric oak (Jenkins 2007, Madden et al. 2004). There were nine fires (4 prescribed and 5 wildfires (Table 3-1). These fires occurred between 2002 and 2013, were each less than 750 ha, and occurred during late spring and early summer months (March-June). Lightening ignited five of the fires and four were prescribed burns conducted by the GSMNP Fire Effects Monitoring Crew (Robert Klein, GSMNP Fire Ecologist, personal communication). The Jesse Ridge prescribed fire was unique. It burned only at low severity and was the only burn sampled within one year post-fire. Data from this fire were excluded from several of the analyses including forest type and time since burn factor variables. Thus, all but one of these fires resulted in a heterogeneous patchwork of severity ranging from very low to very high, which allowed evaluation of the full range of dNBR values.

Plot Locations, Burn Severity Designations, and Sampling

To evaluate the relationships between dNBR and CBI burn severity designations, ground-based severity data and satellite-based severity data from

Table 3-1. Prescribed and wildfires occurring over the past 10 years and sampled in GSMNP to test the relationships between dNBR and CBI

Fire Name	*Fire Type*	*Size (ha)*	*Date of Fire*	*Latitude (N) X Longitude(W) Within Fire Perimeter*
Green Mountain	Wild	720	03-01-2001	35.624221° X 83.559403°
Tabcat	Prescribed	52	03-12-2003	35.521389° X 83.974167°
Chilly Springs	Wild	364	04-04-2006	35.516667° X 83.911389°
Smokemont	Prescribed	92	03-02-2007	35.564713° X 83.331620°
Wash Ridge	Wild	242	03-24-2007	35.616669° X 83.129021°
Wear Cove Gap	Prescribed	168	04-27-2009	35.682262° X 83.618916°
Laurel Falls	Wild	167	04-27-2009	35.682222° X 83.618889°
Calderwood	Wild	91	05-01-2010	35.728933° X 83.911389°
Jesse Ridge	Prescribed	148	05-01-2013	35.635281° X 83.112968°

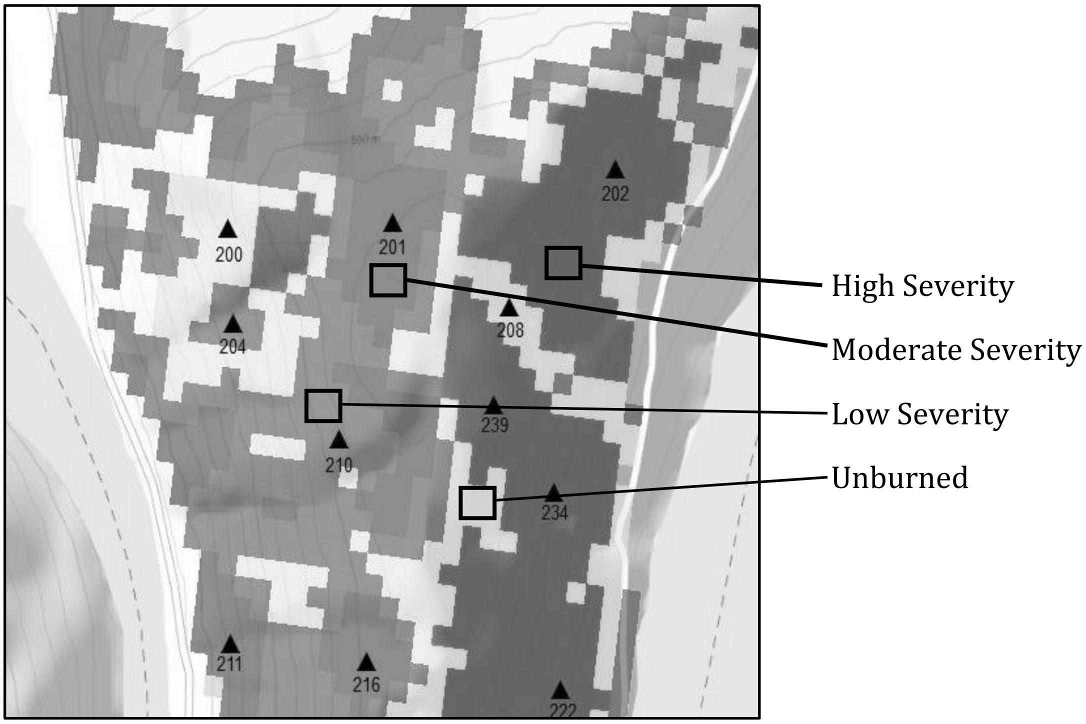
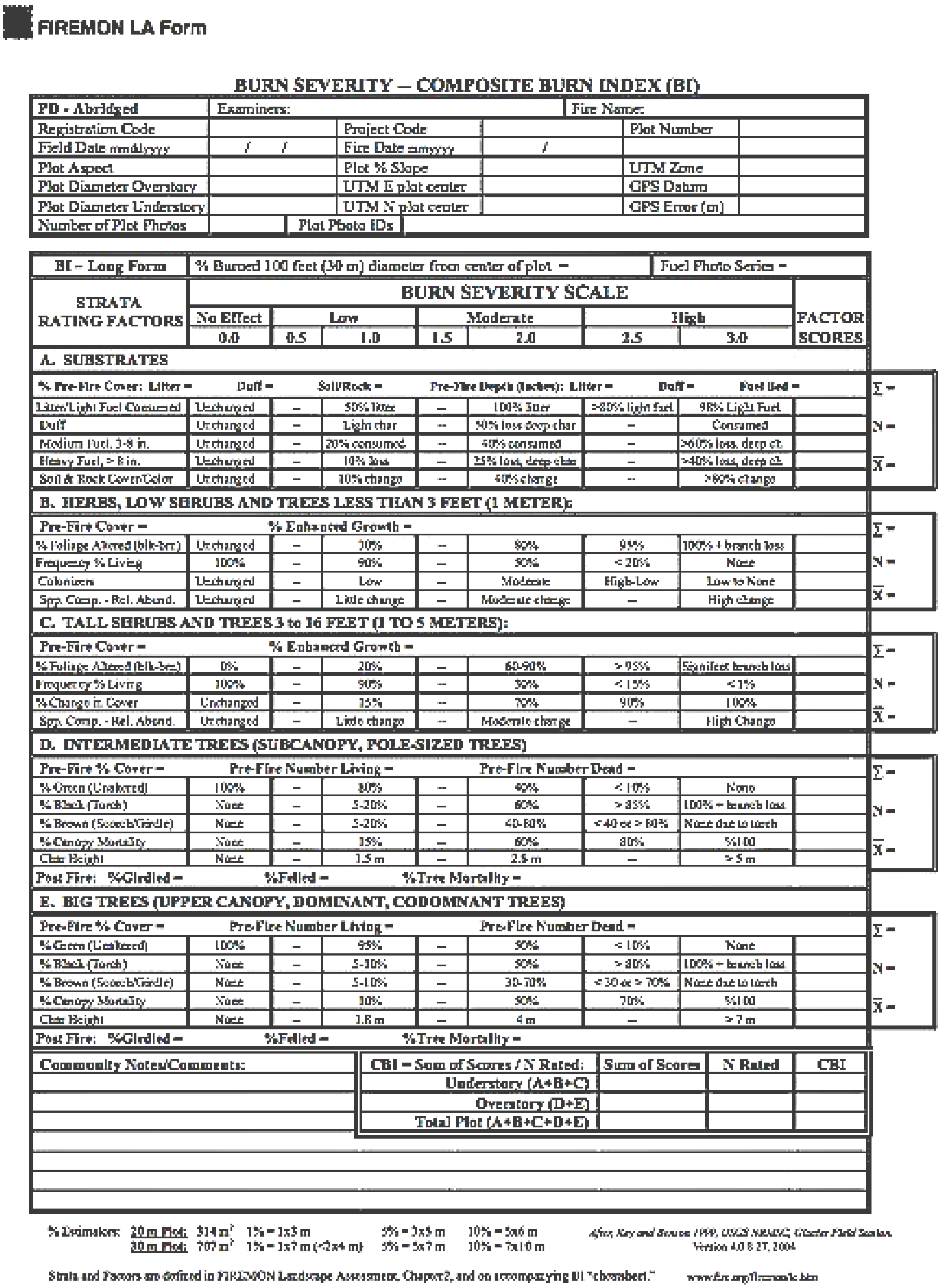


Fig. 3-2. Burn severity map for the Calderwood fire in 2010 with plots indicated by black triangles.

169 plots were collected. The dNBR satellite images were collected between 1 and 5 years before each fire and within 3 months post-fire. GSMNP personnel created burn severity maps, which delineated burn severity classes of unburned, low, moderate, and high severity for each fire (Figure 2). These burn severity maps were used to randomly locate 169 plots for CBI collection. Ground-based severity measures were evaluated and quantified using CBI as described by Key and Benson (2006) in the FIREMON Landscape Assessment (Figure 3-3).

To test the relationship between CBI and dNBR across forest types, plots were randomly stratified in areas with relatively homogenous dNBR values within the four burn severity categories for each fire using GIS layers including topography, dNBR burn severity classes, and forest types for each fire. The four xeric forest types (yellow pine, chestnut oak, oak-hickory, and white-pine xeric oak) were identified on the burn maps and plots were established in each (Jenkins 2007, Madden et al. 2004, Whitaker 1956). To test the relationship between CBI and dNBR across post-fire times of data collection, eight of the fires were categorized into two time treatments (1 to 5 years and 6 to 10 years) based on when they occurred relative to when data were collected. These two categories were chosen to represent the time during extended assessments evaluating immediate and delayed short-term fire effects described by Key and Benson (2006), which occur within 10 years post-fire. The Jesse Ridge prescribed fire was sampled within one year post-fire and was not included in the



FIREMON LA Form

BURN SEVERITY – COMPOSITE BURN INDEX (BI)

PD - Abridged	Examiners:			Fire Name:	
Registration Code		Project Code		Plot Number	
Field Date mmddyyyy	/ /	Fire Date mmyyyy	/		
Plot Aspect		Plot % Slope		UTM Zone	
Plot Diameter Overstory		UTM E plot center		GPS Datum	
Plot Diameter Understory		UTM N plot center		GPS Error (m)	
Number of Plot Photos		Plot Photo IDs			

BI – Long Form	% Burned 100 feet (30 m) diameter from center of plot =					Fuel Photo Series =			
STRATA RATING FACTORS	**BURN SEVERITY SCALE**							**FACTOR SCORES**	
	No Effect	**Low**		**Moderate**		**High**			
	0.0	**0.5**	**1.0**	**1.5**	**2.0**	**2.5**	**3.0**		
A. SUBSTRATES									
% Pre-Fire Cover: Litter = Duff = Soil/Rock = Pre-Fire Depth (inches): Litter = Duff = Fuel Bed =									Σ =
Litter/Light Fuel Consumed	Unchanged	–	50% litter	–	100% litter	>80% light fuel	98% Light Fuel		
Duff	Unchanged	–	Light char	–	50% loss deep char	–	Consumed		N =
Medium Fuel, 3-8 in.	Unchanged	–	20% consumed	–	40% consumed	–	>60% loss, deep ch.		
Heavy Fuel, > 8 in.	Unchanged	–	10% loss	–	25% loss, deep char	–	>40% loss, deep ch.		$\bar{X}$ =
Soil & Rock Cover/Color	Unchanged	–	10% change	–	40% change	–	>80% change		
B. HERBS, LOW SHRUBS AND TREES LESS THAN 3 FEET (1 METER):									
Pre-Fire Cover = % Enhanced Growth =									Σ =
% Foliage Altered (blk-brn)	Unchanged	–	30%	–	80%	95%	100% + branch loss		
Frequency % Living	100%	–	90%	–	50%	< 20%	None		N =
Colonizers	Unchanged	–	Low	–	Moderate	High-Low	Low to None		
Spp. Comp. - Rel. Abund.	Unchanged	–	Little change	–	Moderate change	–	High change		$\bar{X}$ =
C. TALL SHRUBS AND TREES 3 to 16 FEET (1 TO 5 METERS):									
Pre-Fire Cover = % Enhanced Growth =									Σ =
% Foliage Altered (blk-brn)	0%	–	20%	–	60-90%	> 95%	Significant branch loss		
Frequency % Living	100%	–	90%	–	30%	< 15%	< 1%		N =
% Change in Cover	Unchanged	–	15%	–	70%	90%	100%		
Spp. Comp. - Rel. Abund.	Unchanged	–	Little change	–	Moderate change	–	High Change		$\bar{X}$ =
D. INTERMEDIATE TREES (SUBCANOPY, POLE-SIZED TREES)									
Pre-Fire % Cover = Pre-Fire Number Living = Pre-Fire Number Dead =									Σ =
% Green (Unaltered)	100%	–	80%	–	40%	< 10%	None		
% Black (Torch)	None	–	5-20%	–	60%	> 85%	100% + branch loss		N =
% Brown (Scorch/Girdle)	None	–	5-20%	–	40-80%	< 40 or > 80%	None due to torch		
% Canopy Mortality	None	–	15%	–	60%	80%	%100		$\bar{X}$ =
Char Height	None	–	1.5 m	–	2.8 m	–	> 5 m		
Post Fire: %Girdled = %Felled = %Tree Mortality =									
E. BIG TREES (UPPER CANOPY, DOMINANT, CODOMINANT TREES)									
Pre-Fire % Cover = Pre-Fire Number Living = Pre-Fire Number Dead =									Σ =
% Green (Unaltered)	100%	–	95%	–	50%	< 10%	None		
% Black (Torch)	None	–	5-10%	–	50%	> 80%	100% + branch loss		N =
% Brown (Scorch/Girdle)	None	–	5-10%	–	30-70%	< 30 or > 70%	None due to torch		
% Canopy Mortality	None	–	10%	–	50%	70%	%100		$\bar{X}$ =
Char Height	None	–	1.8 m	–	4 m	–	> 7 m		
Post Fire: %Girdled = %Felled = %Tree Mortality =									

Community Notes/Comments:

CBI = Sum of Scores / N Rated:	**Sum of Scores**	**N Rated**	**CBI**
Understory (A+B+C)			
Overstory (D+E)			
Total Plot (A+B+C+D+E)			

% Estimators: 20 m Plot: 314 m² 1% = 1x3 m 5% = 3x5 m 10% = 5x6 m
30 m Plot: 707 m² 1% = 1x7 m (=2x4 m) 5% = 5x7 m 10% = 7x10 m

After, Key and Benson 1999, USGS NRMSC, Glacier Field Station.
Version 4.0 8/27/2004

Strata and Factors are defined in FIREMON Landscape Assessment, Chapter 2, and on accompanying BI "cheatsheet." www.fire.org/firemon/lc.htm

Fig. 3-3. Field data sheet from FIREMON: Landscape Assessment showing variables used to determine CBI (Key and Benson 2006) extended assessment time categories (Key and Benson 2006). Individual CBI forest layer averages were compared to dNBR values to evaluate how well individual components of CBI were related to dNBR (Key and Benson 2006). Once plots were located using burn severity maps, plot centers were used to delineate pixel values for each plot from raw dNBR images. A handheld GPS unit was used to locate the plot centers in the field. CBI data were collected for the 109 newly sampled plots between May and October 2013, and for 60 plots previously sampled by GSMNP between 2001 and 2003 for a total of 169 plots for my study.

Data Analyses

Individual CBI forest layer values from the field (substrate, low-shrub, midstory, sub-canopy, and canopy) were added together and one average overall CBI score was calculated for each of the 169 sampled plots. Forest type data was not known for the 60 plots sampled by GSMNP personnel and thus was excluded from analyses where forest type was a factor. The Jesse Ridge prescribed fire was sampled within the initial assessment time period (Key and Benson 2006) and the 14 plots located within this fire were excluded from analyses where time since burn was a factor. Absolute dNBR values were recorded to evaluate total amount of change, including ‘enhanced plant growth’, post-fire indicated by the difference in Landsat bands 4 and 7 values (Fuller and Fulk 2001, Key and Benson 2006, Miller et al. 2009).

The relationship between dNBR and CBI was examined using a general linear model. Scatter plots from the linear models suggested a saturated growth relationship between dNBR and CBI, thus inferences based solely on linear regression correlation coefficients could be misleading (Hall et al. 2008). Therefore, five different models predicting CBI-based burn severity as a function of dNBR (Table 3-2) were evaluated and tested for best fit using the Akaike Information Criterion (AIC). The AIC is a measure of the relative value of a statistical model for a given set of data where the lowest score indicates the highest value and thus the best fit (Posada and Buckley 2004). A model using square-root transformed absolute dNBR values provided the best fit to the data based on AIC scores (Table 3-2). I used a reduced data set, for which there were no missing values, to test for effects of forest type, time since burn, and the interaction between the two, for each component score of the CBI and the compound CBI score. Analysis of covariance was used to test for differences in the relationship between dNBR and CBI across forest types and time since burn. All data analyses were conducted using software R version 3.0.2 (R development Core Team 2012).

Results

The overall relationship between dNBR and CBI was significant and strongly correlated (Table 3-3, Figure 3-4). Analysis of the relationship between

Table 3-2. Akaike Information Criterion (AIC) for five models describing CBI-based burn severity as a function of remotely-sensed variables derived from Landsat imagery

Model Type	*Model*	*df*	*AIC*
Untransformed Linear Regression	$CBI = 0.003(dNBR) + 0.5961$	3	199.07
Asymptotic Regression	$CBI = Asym*(R0 – Asym)*e^{(\text{-e\^{}lrc*dNBR})}$	4	173.51
Asymptotic Regression through the Origin	$CBI = Asym*(R0 – Asym)*e^{(\text{-e\^{}lrc*dNBR})}$	3	206.98
Square Root Transformed dNBR Linear Regression	$CBI = 0.097917(\surd abs(dNBR))$	3	163.34
Square Root Transformed dNBR Linear Regression through the Origin	$CBI = 0.097917(\surd abs(dNBR))$	2	164.87

dNBR and CBI using the square-root transformed absolute dNBR model showed no evidence of effects of these factors or their interaction (Table 3-4).

Discussion

The strong relationship between square-root transformed absolute dNBR and CBI is consistent with several other studies (van Wagtendonk et al. 2004, Zhu et al. 2006, Wimberly and Reilly 2007). For example, Cocke et al. (2005) reported an R^2 value of 0.84, Murphy et al. (2008) reported an R^2 value of 0.64, and Soverel et al. (2010) reported an average of $R^2 = 0.70$ across four different fires occurring in western conifer forests, while results from my study reflect the dNBR X CBI relationship for fires occurring in GSMNP.

The saturated relationship between dNBR and CBI has been explained as differences in changes in light reflectance between Landsat bands 7 and 4 used to calculate dNBR (Chuvieco et al. 2006, Hall et al. 2008). The increases in light reflectance coincident with Landsat band 7 (2080 – 2350 nm) reach maximum reflectance values when CBI values were above 2.0 to 2.5, but because decreases in reflectance coincident with Landsat band 4 (760 – 900 nm) are consistent with increases in burn severity, the correlation between CBI

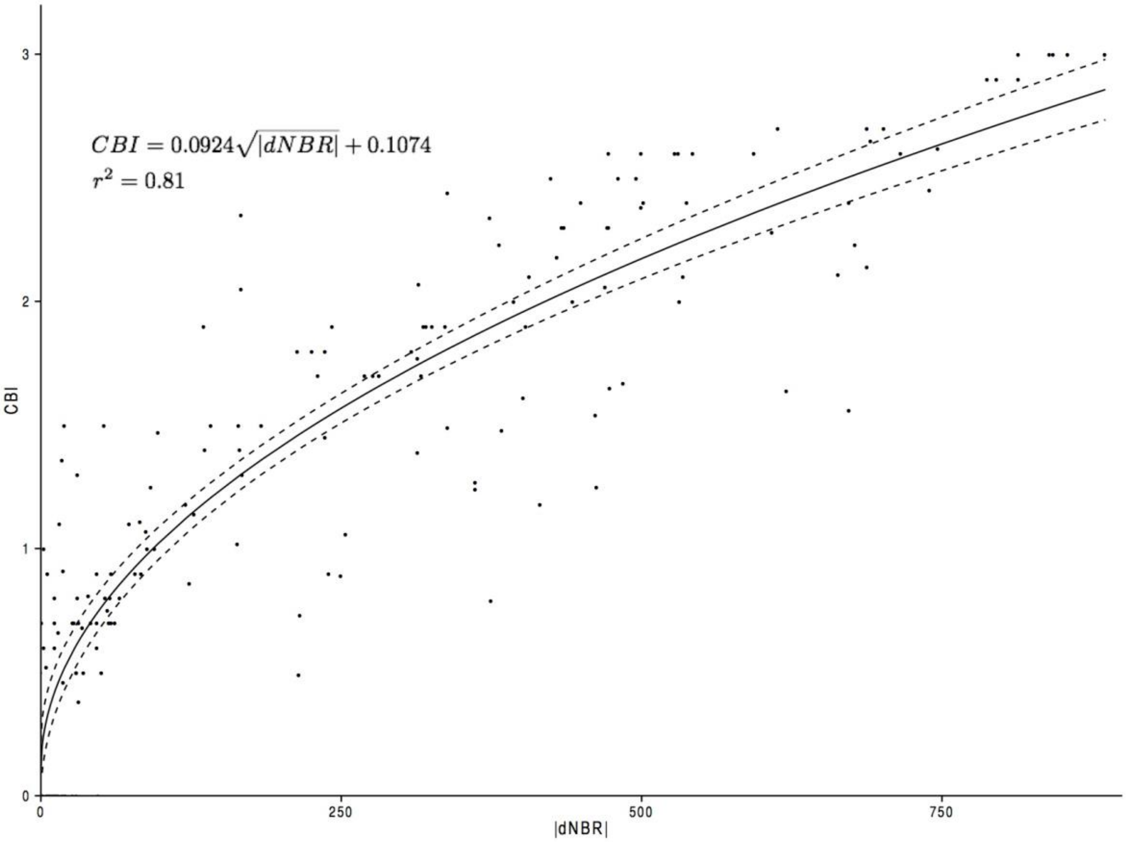


Fig. 3-4. Regression between square-root transformed absolute dNBR and CBI for all data (n=169). Dashed lines represent the 95% confidence interval.

Table 3-3. Linear regression between untransformed dNBR and CBI (LIN) and linear regression between square root transformed dNBR and CBI (SQRT) for all data points (n=169)

dNBR X CBI	*n*	*AveCBI*	*Ave dNBR*	*LIN R^2*	*p-value*	*SQRT R^2*	*p-value*
Overall	169	1.45(0.91)	273(260)	0.76	< 0.001	0.81	< 0.001

Table 3-4. ANOVA comparing differences in the relationship between dNBR and CBI among forest type and time since burn factors. CBI was used as the response variable.

Factor	Df	F-Value	p-value
Forest Type	3	2.39	0.0749
Time Since Burn	1	2.65	0.1077
Forest Type X Time Since Burn	1	2.91	0.0921

and dNBR initially increases rapidly then slows down once CBI values exceed 2.0 to 2.5 (Hall et al. 2008, Chuvieco et al. 2006, Key and Benson 2006).

Conclusion

This study validated the use of dNBR for mapping burn severity as an alternative to using CBI in GSMNP. In addition, this study suggested modeling burn severity can be improved over the other models reported in the literature by using a square root transformation for dNBR values. Finally, dNBR accurately predicted CBI in different xeric forest types up to 10 years ago.

CHAPTER 4: MANUSCRIPT:

COMPARING POST-FIRE STAND CHARACTERISTICS ON XERIC SITES EXPERIENCING DIFFERENT BURN SEVERITIES IN GREAT SMOKY MOUNTAINS NATIONAL PARK

Introduction

Wildfire is one of the ecosystem processes that can create and maintain specific forest structures, composition, and functions across a wide range of spatial and temporal scales (Lentile et al. 2006). A fire regime is the frequency, severity, size, and spatial pattern of wildfire (Morgan et al. 2001). Historical fire regimes in the eastern United States ranged from relatively rare stand-replacing events to frequent low intensity ground fires and supported natural communities adapted to these different fire regimes such as tall-grass prairies, aspen (*Populus*) groves, oak (*Quercus*) dominated hardwoods, and pine (*Pinus)* communities (Nowacki and Abrams 2008). The southern Appalachian xeric sites dominated by pines and oaks occurred on south- to west-facing slopes and ridges and were maintained by a less than 20 year fire interval from natural and human ignitions (Harmon 1982) in the southern Appalachian region throughout the Holocene Epoch (Fesenmyer and Christensen 2010, Delcourt and Delcourt 1997). Plant communities that evolved with this fire frequency are adapted to the low and moderate mixed-severity wildfires, which promote seedbed conditions suitable for regeneration of species such as pine and oak where the mature trees with characteristics such as thick bark, sprouting, and compartmentalization of fire wounds allows them to survive and provide seed (Nowacki and Abrams

2008, Hutchinson et al. 2008, Welch et al. 2000). Several studies reported pine and oak regeneration increase with increased available light, exposed mineral soil, and decreased competition from fire-sensitive species following fire, but the response of different species and species groups to fire varies significantly with fire intensity and pre-fire site conditions (Fesenmeyer and Christensen 2010, Jenkins et al. 2011, Harrod et al. 2000, Lafon et al. 2007).

In the Great Smoky Mountains National Park (GSMNP), xeric sites occupy between 16-25% (30,000-50,000 hectares) of the total forested area (Madden et al. 2004, Jenkins 2007). The most common forest types found on xeric sites in the Park are dominated by yellow pine (*Pinus subgnus Diploxylon)* and dry-site oak (*Quercus*) species (Whitaker 1956). Dominant yellow pine (*Pinus)* species include pitch pine (*Pinus rigida* Mill.), shortleaf pine (*Pinus echinata* Mill.), and Virginia pine (*Pinus virginiana* Mill.), all of which require relatively thin litter layers (≤ 4 cm) and relative high amounts of insolation from canopy gaps for successful germination and establishment (Whitaker 1956, Callaway et al. 1987, Harmon 1982, Waldrop and Brose 1999, Jenkins et al. 2011). Dominant dry-site oak species include chestnut oak (*Quercus montana* Willd.), scarlet oak (*Quercus coccinea* Munchh.), and white oak (*Quercus alba* L.). These species all have relatively thick bark and the ability to sprout following fire (Whitaker 1956, Jenkins 2007, Lafon et al. 2007). These characteristics of the yellow pine and dry-site oak species suggest that perpetuation of these community types relied on relatively frequent fire disturbances (≤ 20 years) (Harmon 1984, Harrod et al. 1998, Brose et al. 2013). When wildfires were suppressed in GSMNP starting in the 1930s ecological succession resulted in many of these sites becoming dominated by high densities of shade-tolerant species such as red maple (*Acer rubrum* L.), black gum (*Nyssa sylvatica* Marsh.), eastern white pine (*Pinus strobus* L.), and eastern hemlock (*Tsuga canadensis* (L.) Carriere), along with a well-developed shrub layer (Nowacki and Abrams 2008, Brose et al. 2013, Harmon 1984, Jenkins et al. 2011). For example, mean canopy density (stems/ha) increased from about 250 stems/ha in 1938 to over 600 stems/ha in 1995 (Harrod et al. 2000, MacKenzie and White 1998).

Competition from fire-sensitive species and an increase in litter layer depths on sites where fire was suppressed since 1940 (Jenkins et al. 2011, Welch et al. 2000, Nowacki and Abrams 2008, Harrod et al. 1998) shifted microclimates in the xeric sites to cooler, moister conditions by creating dense stands and relatively thick (> 10cm) litter layers (Nowacki and Abrams 2008, Jenkins et al. 2011). These conditions hinder pine and oak regeneration and increase decomposition rates and nutrient availability due to accumulations of litter and higher retention of moisture, which favor species such as red maple (Shure and Wilson 1993, Nowacki and Abrams 2008). Managed disturbance such as prescribed fire is needed to decrease competition, open canopies, and reduce litter to restore and maintain pine and oak forest types (Arthur et al. 1998, Jenkins et al. 2011, Harrod et al. 2000, Welch et al. 2000). However, using prescribed fire effectively to achieve restoration objectives depends on our ability to predict the short- and long-term effects of fire and burn severity on forest structure and composition (Jenkins et al. 2011, Kasischke et al. 2008).

Unfortunately, studies characterizing xeric site forest communities following multi-severity fires in GSMNP are lacking. My study was designed to characterize post-fire forest stand structure and composition in pine and oak dominated sites among different burn severity categories in GSMNP.

Objectives

The purpose of this research was to determine if there are differences in post-fire pine and oak stand structure and composition among burn severity categories (unburned, low, moderate, high) and to characterize these differences as they relate to site conditions favoring pine and oak regeneration on xeric sites throughout GSMNP. The following predictions were tested to evaluate forest structure and composition among burn severity categories:

There are differences among burn severity categories in

(1) forest density (stems/ha)
(2) basal area (m^2/ha)
(3) canopy mortality (%)
(4) litter/duff layer depths (cm)
(5) grass cover/bare ground (%)
(6) shrub height (m)
(7) species richness (#/ha)
(8) regeneration densities (stems/ha)
(9) relative regeneration densities (%)
(10) relative overstory importance values (%)

Methods

Study Area

This study took place in the western half of GSMNP in North Carolina and Tennessee (Figure 4-1). Mean aspect for all sites was 189° (+/- 43°) and mean slope was 32% (+/- 10%). Plots were located within xeric site delineations provided by GSMNP (Madden et al. 2004, Jenkins 2007). The fires had occurred on sites characterized by warm and dry conditions with rocky and highly weathered shallow soils (Whitaker 1956). The four prescribed and wildfires used in this study (Table 4-1) occurred between 2006 and 2010 (between 4 and 8 years old) and ranged in size from 91-364 hectares. Lightening ignited three of the fires and one was a prescribed burn conducted by the GSMNP Fire Effects Monitoring Crew (Rob Klein, NPS Fire Ecologist, personal communication). All fires studied occurred during late spring and early summer (March to June). Unburned forested areas adjacent to the burned areas were also sampled in order to further characterize site conditions related to post-fire burn severity.

Plot Locations and Sampling

Burn severity data, including severity maps (Fig. 4-2), fire perimeter maps, and metadata for the four fires, were obtained from the GSMNP. Burn

Fig. 4-1. Fires studied in western Great Smoky Mountains National Park in North Carolina and Tennessee (red star indicates area of fire locations).

Table 4-1. Fires sampled in this study in GSMNP to evaluate forest structure and composition responses to different burn severities.

Fire Name	*Fire Type*	*Size (ha)*	*Date of Fire*	*Latitude (N) X Longitude(W) Within Fire Perimeter*
Chilly Springs	Wild	364	04-04-2006	35.516667° X 83.911389°
Wear Cove Gap	Prescribed	168	04-27-2009	35.682262° X 83.618916°
Laurel Falls	Wild	167	04-27-2009	35.682222° X 83.618889°
Calderwood	Wild	91	05-01-2010	35.728933° X 83.911389°

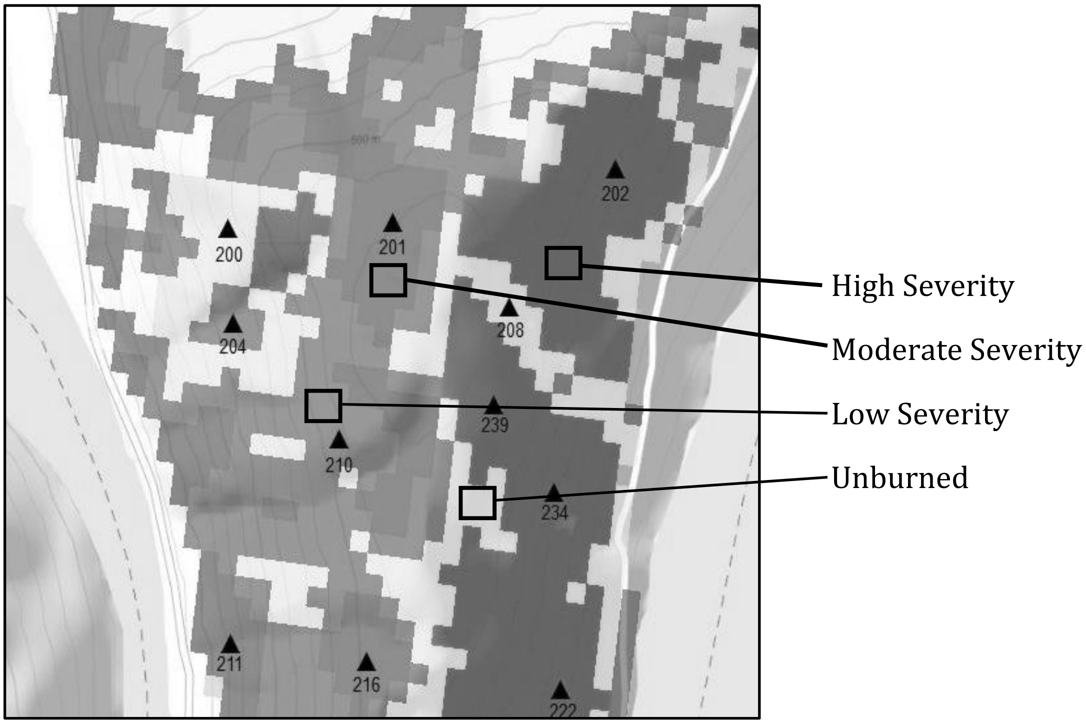


Fig. 3-2. Burn severity map for the Calderwood fire in 2010 with plots indicated by black triangles.

severity maps were created using the differenced normalized burn ratio (dNBR), which is a commonly used remotely sensed measure of burn severity (Wimberly and Reilly 2007, Zhu et al. 2006, Lentile et al. 2006, Key and Benson 2006). These maps were used to randomly locate plots in different burn severities (unburned, low, moderate, and high) for each fire (Figure 4-2). Within each fire and burn severity designation, plots were located in areas with relatively homogenous burn severity and were randomly stratified using topography and forest type GIS layers. A handheld GPS unit was used to locate the plot centers in the field.

Data Collection

Plots were 30 meters in diameter with 10-meter subplots located at plot centers. For 30-meter plots all trees ≥ 5 cm diameter were identified to species, classified as live or standing dead, and diameters were measured at breast height (1.3 meters) to evaluate density, basal area, and mortality among burn severity classes. Canopy mortality (%) was estimated using the ratio of living to standing dead stems (≥5cm dbh). Two organic material layer depths were sampled; litter layer (light dead plant material) and duff layer (dense layer between A-horizon and litter layer [Whitaker 1956]) depths were measured at the cardinal direction points along the outside of the 10-meter subplot to evaluate seedbed differences among burn severity classes. For 10-meter subplots all stems < 5 cm diameter and taller than 8 cm were identified to species and tallied to evaluate relative density and differences in regeneration type and species composition across burn severity classes. Clumped basal sprouts were tallied as single individuals. Bare ground cover (%), grass cover (%), and average shrub height (m) were also estimated for each 10-meter subplot to further characterize the groundstory and understory forest layers among burn severity categories.

Data Analyses

My study consisted of four fires, four severity zones per fire, and 3 plots per severity zone for each variable measured for a total of 48 plots. ANOVA was used to test for differences in overstory/understory density (stems/ha), basal area (m^2/ha), overstory mortality (%) litter layer depth (cm), duff layer depth (cm), bare ground cover (%), grass cover (%), and average shrub height (m) to evaluate differences in forest structure across burn severities. ANOVA was also used to test for differences in species richness (# species), regeneration composition density (stems/ha), and relative density of yellow pine, oak, and mixed mesophytic species to evaluate differences in forest composition across burn severities. For analyzing regeneration composition responses across burn severities, all stems > 8 cm tall and < 5 cm diameter were categorized into three groups: (1) dry-site oak species (chestnut oak, white oak, black oak [*Quercus velutina* Lamb.], and scarlet oak), (2) yellow pine species (shortleaf pine, Virginia pine, pitch pine, and table mountain pine [*Pinus pungens* Lamb.]), and (3) mixed mesophytic species (all remaining species). Because of the ecological importance and similar site condition requirements of both yellow pine and dry-site oak species on xeric sites (Harrod et al. 2000), relative densities of these two species groups were combined and compared to relative density of mixed mesophytic regeneration to evaluate characteristics of burn severity zones related to xeric stand regeneration. Density, dominance, and frequency values were calculated and used to determine relative importance values for overstory tree species to evaluate differences in species dominance among burn severity categories. The most important species for each burn severity category were used to evaluate differences in species composition among burn severity classes. All data analyses were conducted in the analysis software R version 3.1.1 (R development Core Team 2012).

Results

Forest Stand Structure

Overall average overstory density was 354 stems/ha and was significantly lower in high severity compared to the other burn severity categories ($p < 0.001$) (Table 4-2a; 4-2b). Compared to unburned sites, density was 70% less in high severity sites. Overall average understory density was 12,965 stems/ha and did not vary significantly among fire severity categories ($p = 0.124$) (Table 4-2b). Although compared to unburned sites, understory density tended to be lower in low severity fire and was about 30% higher in moderate severity sites,

Table 4-2a. Overall averages and averages of field measures of forest structure and composition for each burn severity category. Values in parentheses are standard deviations of the means.

Field Measure	*Overall*	*Unburned*	*Low Burn*	*Mod. Burn*	*High Burn*
Overstory Density (# live stems/ha)	354 ±(27)	547 ±(28)	440 ±(31)	557 ±(31)	162 ±(11)
Understory Density (# live stems/ha)	12965 ± (1891)	12742 ± (2096)	9326 ± (2210)	17294 ± (3003)	12499 ± (2921)
Overstory Basal Area (m^2/ha)	52.0 ± (6.0)	81.1 ± (7.0)	52.5 ± (5.0)	68.6 ± (5.0)	5.9 ± (0.5)
Overstory Mortality (%)	39.0 ± (37.0)	1.3 ± (1.2)	12.5 ± (6.5)	50.4 ± (10.5)	91.9 ± (6.6)
Litter Layer Depth (cm)	6.9 ± (4.6)	12.3 ± (3.1)	8.7 ± (2.9)	4.0 ± (2.6)	2.8 ± (1.6)
Duff Layer Depth (cm)	2.3 ± (1.8)	3.4 ± (2.6)	1.4 ± (1.5)	2.8 ± (1.3)	1.7 ± (1.6)
Bare Ground Cover (%)	27.6 ± (24.5)	26.4 ± (25.9)	42.9 ± (27.7)	33.5 ± (25.8)	7.7 ± (8.1)
Grass Cover (%)	3.1 ± (2.9)	1.7 ± (3.1)	4.8 ± (3.3)	2.5 ± (1.9)	3.3 ± (5.9)
Average Shrub Height (m)	1.0 ± (0.8)	1.7 ± (0.8)	0.6 ± (0.6)	0.8 ± (0.7)	1.2 ± (0.7)
Overstory Species Richness (#/ha)	17 ± (4)	21 ± (3)	15 ± (4)	21 ± (5)	9 ± (2)
Understory Species Richness (#/ha)	23 ± (4)	22 ± (3)	25 ± (5)	24 ± (5)	21 ± (2)
Oak Regeneration Density (#/ha)	4042 ± (2097)	1909 ± (601)	2674 ± (341)	7894 ± (1426)	3692 ± (1074)
Yellow Pine Regeneration Density (#/ha)	1578 ± (645)	522 ± (105)	1107 ± (408)	2520 ± (808)	2164 ± (1108)
Mesophytic Regeneration Density (#/ha)	7512 ± (2353)	10186 ± (2603)	5538 ± (1112)	7003 ± (2031)	7321 ± (1876)
Relative Pine-Oak Regeneration Density (%)	41 ± (16)	19 ± (8)	41 ± (12)	60 ± (14)	44 ± (11)

Table 4-2b. Tukey's similarities (pairwise comparison of means) for average values of field measures of forest structure and composition across burn severities. Letters within a row represent significant differences in means.

Field Measure	*No Burn*	*Low Burn*	*Mod. Burn*	*High Burn*	*p-value*
Overstory Density (# live stems/ha)	b	b	b	a	< 0.001
Understory Density (# live stems/ha)	a	a	a	a	0.124
Overstory Basal Area (m^2/ha)	b	b	b	a	< 0.010
Overstory Mortality (% snags:total stems)	a	a	b	c	< 0.001
Litter Layer Depth (cm)	c	b	a	a	< 0.001
Duff Layer Depth (cm)	b	a	ab	ab	< 0.050
Bare Ground Cover (%)	ab	b	b	a	< 0.010
Grass Cover (%)	a	a	a	a	0.244
Average Shrub Height (m)	b	a	a	ab	< 0.010
Overstory Species Richness (#/ha)	b	b	b	a	< 0.050
Understory Species Richness (#/ha)	a	a	a	a	0.441
Oak Regeneration Density (#/ha)	a	ab	b	ab	< 0.050
Yellow Pine Regeneration Density (#/ha)	a	a	a	a	0.231
Mesophytic Regeneration Density (#/ha)	b	a	ab	ab	< 0.050
Relative Pine-Oak Regeneration Density (%)	a	ab	b	b	< 0.050

while high severity sites exhibited little difference to sites without fire. Overall average overstory basal area was 52.0 m^2/ha and was significantly lower ($p < 0.01$) (Table 4-2a: 2b) in sites with high burn severity versus unburned sites. Compared to unburned sites, basal area was nearly 95% lower in high severity sites, 40% lower in low severity sites, and only 15% lower in moderate severity sites. Overall average overstory mortality was 39% and was significantly higher in higher burn severity categories ($p < 0.001$) (Table 4-2a; 2b). Unburned and low severity sites had lower overstory mortality and were not significantly different from each other. Average overstory mortality on moderate severity sites was estimated at 50% and significantly different than overstory mortality on high severity sites, which was 92% (Table 4-2a; 2b).

Overall average leaf litter depth was 6.9 cm and was significantly smaller on burned sites ($p < 0.001$) (Table 4-2a; 2b). Average leaf litter depths on high and moderate severity sites were not different from each other, but were significantly smaller than both low severity sites and unburned sites, which were also significantly different from each other. Overall average duff layer depth was 3.4 cm and was significantly higher in unburned sites than in high severity sites, but less different between low and moderate severity ($p < 0.050$) (Table 4-2b). Average bare ground cover was significantly lower in high severity sites than in low and moderate severity sites (Table 4-2a; 2b). Overall average grass cover was 3.1% and did not vary among burn severity categories (Table 4-2a; 2b). Overall average shrub height was 1 meter and was significantly lower in low and moderate severity sites than in unburned sites, while average shrub height was less different in high severity sites (Table 4-2a; 2b).

Forest Stand Composition

Overall average overstory species richness was 17 species/ha. Overstory species richness was significantly lower on high severity sites compared to all other sites ($p < 0.001$) (Table 4-2a; 2b). Overall average understory species richness was 23 species/ha and was not significantly different among burn severity categories ($p = 0.441$) (Table 4-2a; 2b).

Oak regeneration was significantly different among burn severity classes ($p < 0.05$) (Table 4-2a; 2b). Compared to unburned sites, oak regeneration was over 25% higher in low severity sites and three times higher in moderate severity sites. However, oak regeneration was 50% less in high severity sites than in unburned sites. Yellow pine regeneration was not significantly different among burn severity categories ($p = 0.317$) (Table 4-2a; 2b), however, compared to unburned sites, yellow pine density was two times higher in low severity sites and about five times higher in both moderate and high severity sites. Mixed mesophytic regeneration was significantly different among fire severity categories ($p < 0.05$) (Table 4-2a; 2b). Compared to unburned sites, mixed mesophytic regeneration was about 50% lower in low severity sites, while 30% lower in both moderate and high severity fire. Combined pine and oak regeneration was significantly different among fire severity categories ($p < 0.01$) (Table 4-2) and was significantly lower in unburned sites than both high and moderate severity sites, which were not different from each other. Low severity sites were less different to all other fire severity categories. Combined pine and oak regeneration dominated (> 50%) moderate severity sites and occupied less than 50% in all other burn severity categories.

Relative density of the three species groups (yellow pine, oak, and mixed mesophytic) varied among burn severity categories (Table 4-3). Compared to unburned sites, yellow pine relative density was higher in high severity sites. Dry-site oak relative density was twice as high in low and high severity sites, but three times higher in moderate severity sites. Compared to unburned sites, mixed mesophytic relative density was about 25% lower in low and high severity sites, but 50% lower in moderate severity sites.

Chestnut oak was the most important overstory species in unburned sites and second most important species in all other fire severity categories. Black gum (*Nyssa sylvatica* L.) was the most important overstory species in low severity sites and second most important species in high severity sites (Table 4-4). Pitch pine was the most important overstory species in moderate severity sites and third most important species in unburned and low severity sites. Virginia pine was the most important overstory species high severity sites and third most important in moderate severity sites.

Table 4-3. Relative density for regeneration of three species groups (yellow pine, oak, and mixed mesophytic).

Species Group	*Unburned*	*Low Burn*	*Mod. Burn*	*High Burn*
Yellow pine species	0.04	0.12	0.15	0.16
Oak species	0.15	0.29	0.45	0.28
Mixed Mesophytic species	0.81	0.59	0.40	0.56
Totals	1.00	1.00	1.00	1.00

Table 4-4. Relative importance values for overstory species (see APPENDIX Table A-1 for species names that correspond to the four-letter code).

Species	*Unburned*	*Low Burn*	*Mod. Burn*	*High Burn*
ACPE	0.01	0.00	0.00	0.00
ACRU	0.16	0.08	0.13	0.15
BETU	0.01	0.02	0.00	0.00
CARY	0.02	0.05	0.03	0.00
COFL	0.01	0.00	0.00	0.00
FAGR	0.04	0.00	0.00	0.00
HATE	0.01	0.00	0.00	0.00
JUVI	0.00	0.00	0.01	0.00
LITU	0.07	0.00	0.01	0.00
MAFR	0.01	0.00	0.02	0.00
NYSY	0.04	0.24	0.10	0.16
OXAR	0.09	0.07	0.07	0.05
PIRI	0.11	0.14	0.17	0.09
PIST	0.06	0.00	0.04	0.00
PIVI	0.07	0.04	0.13	0.24
PRSE	0.00	0.00	0.00	0.00
QUAL	0.02	0.02	0.00	0.02
QUCO	0.04	0.09	0.10	0.10
QUMO	0.19	0.21	0.14	0.16
QURU	0.01	0.00	0.01	0.00
QUVE	0.00	0.03	0.03	0.00
ROPS	0.01	0.00	0.00	0.03
SAAL	0.00	0.01	0.00	0.00
TSCA	0.02	0.00	0.01	0.00
TOTALS	1.00	1.00	1.00	1.00

Discussion

Forest Stand Structure

Forest structure results from my study suggest closed canopy forests with densities exceeding 500 stems/ha (for trees $\geq$ 5 cm dbh) commonly occur on unburned xeric sites across GSMNP. Fralish et al. (1991) reported similar current stand densities (415 to 506 stems/ha) for fire-suppressed pine-oak dominated forests occurring on south slopes in Illinois, which were measured at only 144 stems/ha in 1807 during an active fire regime of low-to-moderate fires (Nowacki and Abrams 2008, Harmon 1984). Gottschalk (1985) reported that at least 40% reductions in basal area are needed for advanced oak regeneration to survive into canopy status for closed canopy stands. In my study, basal area (m^2/ha) was about 40% lower in low severity sites than in unburned sites suggesting that even low severity fires are associated with conditions more suitable for pine and oak survival than in sites absent of fire. Although moderate severity sites in my study had similar stand basal areas to low severity sites (Table 4-2a), canopy mortality was 40% higher in moderate severity sites, which may have allowed more light to reach the forest floor potentially creating more suitable conditions that favor pine and oak regeneration (Nowacki and Abrams 2008). In my high severity sites canopy mortality averaged 91% (Table 4-2a), which is consistent with mortality needed for pine regeneration (> 85%) in closed canopy stands reported by Waldrop and Brose (1999), suggesting that high severity fire on xeric sites in GSMNP is associated with conditions suitable for pine regeneration. Stand densities on these high severity sites (Table 4-2a) were consistent with other post-fire densities reported by Fralish (1991), however, due to the high mortality of large overstory trees on these sites, they were lacking mature seed trees needed for regeneration as described by Reilly et al. (2007) and Brose et al. (2013).

In my study sites overstory mortality was higher in burned sites compared to unburned sites and was highest in high severity sites (Table 4-2b) suggesting that overstory mortality may be associated with conditions that support pine and oak regeneration due to gaps in forest canopies allowing light to the forest floor (Nowacki and Abrams 2008). Increases in litter layer depth due to an absence of fire has been shown to favor regeneration of mesophytic species throughout the southern Appalachians by hindering germination of xeric regeneration post-fire (Jenkins et al. 2011, Welch et al. 2000). Litter and duff layer depths in my sites were higher in unburned sites than in sites where fire occurred (Table 4-2b) suggesting sites with relatively small litter and duff layer depths (2.8-4.0 cm) are associated with conditions that support pine and oak regeneration. Bare ground cover was also different among burn severities (Table 4-2b) further suggesting that fires occurring at various intensities in GSMNP support ground conditions potentially suitable for pine and oak regeneration. Average shrub height in my sites was lowest in low and moderate severity sites and Jenkins et al. (2011) reported correlations between shrub structure and suitable conditions for pine and oak regeneration suggesting again that low and moderate severity fires are associated with conditions needed for pine and oak regeneration.

Forest Stand Composition

Species richness did not vary among burn severity categories for understory species suggesting that different burn severities are not associated with differences in species richness and overstory species richness was only different in high severity sites possibly due to very low tree densities in these sites (Table 4-2b). Yellow pine and dry-site oak regeneration was relatively low (Table 4-2a) on unburned sites compared to burned sites suggesting that fire occurring on xeric sites in GSMNP may potentially promote higher densities of yellow pine and dry-site oak species. Moderate sites in my study exhibited the highest combined pine and oak regeneration (Table 4-2a) suggesting that conditions suitable for both pine and oak species regeneration are associated with xeric sites experiencing overstory mortality of at least 50% with basal areas around 69 m^2/ha, litter/duff layer depths of 4.0/2.8 cm, and average shrub height less than 1.0 meter.

In contrast to stand composition in moderate severity sites, high severity sites in my study exhibited high relative mesophytic regeneration, such as red maple, versus pine and oak regeneration (Table 4-2a). Although oak species are similar to red maple in their ability to sprout, they allocate their carbohydrate resources to root growth as opposed to shoot growth (Atwood et al. 2009). Red maple allocates resources to above-ground growth which allows it to overtop the oak species, subsequently creating low light conditions not suitable for either oak or pine regeneration (Gilbert et al. 2003). Pine seedling germination on high severity sites may have been limited due to the lack of seed trees on these sites and the effect of maple being favored over oak can be exacerbated by moisture stress when mineral soil is exposed and sprouts from individual trees with established root systems compete for resources (Waldrop et al. 2003, Shure and Wilson 1993).

Shugart (1984) reported that slow-growing shade-tolerant trees such as red maple replace relatively shade-intolerant trees such as yellow pine and dry-site oak species in the absence of disturbance such as fire. Importance value results from my study, such as Virginia pine in high severity sites, reflect stand characteristics consistent with these studies and are evident in the differences between importance values seen among burn severity categories. For example, in sites where fire was absent red maple, a shade-tolerant species, was the second most important overstory species suggesting that conditions in unburned sites are associated with the growth and survival of mesophytic species. Shade-tolerant species have been shown to change the microclimates in xeric sites by decreasing available light and increasing soil moisture levels in the soil through accumulated litter, which hinders pine and oak regeneration (Nowacki and Abrams 2008, Runkle 1982). Similarly, the shade-tolerant black gum species was the most important overstory species in low severity sites where canopy mortality was relatively low (Table 4-2a), suggesting that conditions in low severity sites are also associated with the growth and survival of mesophytic species. However, on xeric sites where moderate severity fire occurred the most important overstory species was pitch pine followed by chestnut oak, suggesting that moderate severity fires are associated with conditions favoring the growth and survival of pine and oak species over mesophytic species, which is necessary for successful pine and oak regeneration (Gottschalk 1985). Virginia pine, a relatively fire-sensitive yellow pine species and a prolific colonizer (Jenkins et al. 2011), was the most important overstory species in sites experiencing high severity fire suggesting that the regeneration in these sites might have been limited to tolerant species establishing and not a result of the perpetuation of an existing stand (Runkle 1982).

Overall Summary

The results of this study agree with other studies and suggest that burn severity is associated with both earlier forest successional stages and regeneration of pine and oak species on xeric sites across the southern Appalachians and within GSMNP (Jenkins et al. 2011, Harrod et al. 2000, Lafon et al. 2007, Waldrop et al. 2013, Nowacki and Abrams 2008). Fires occurring at varying intensities (measured post-fire as burn severity) are associated with differences in forest stand structure, such as canopy mortality (Table 4-2b), and differences in regeneration composition, such as higher density of pine and oak in moderate severity sites. My results are consistent with reports by Brown et al. 2014, Nowacki and Abrams 2008, and Harrod et al. 2000.

Conclusion

Several studies indicate that higher severity fires are needed, as opposed to low severity fires, for successful restoration of yellow pine and oak species in the southern Appalachians (Hutchinson et al. 2012, Brose and VanLear 1998, Brown et al. 2014). Overall it seems that moderate severity fires are associated with the most favorable conditions for the maintenance and perpetuation of xeric stands in GSMNP by opening canopies, reducing competition, and reducing litter/duff layer. However, more research on the frequency of fire events is needed to better understand the differences in forest stand structure and composition between sites experiencing different burn severity. Results from this study suggest that moderate severity fire may be useful for initial restoration management of yellow pine and oak dominated stands on xeric sites in GSMNP.

CHAPTER 5: PREDICTING CHANGES IN VEGETATION PATTERNS USING DNBR BURN SEVERITY MAPS

Variable site conditions can be created by multi-severity fire (Brown et al. 2014), which creates a heterogeneous mosaic of open and closed canopies needed to support the diversity of species in GSMNP. Predicting burn severity and characterizing how burn severity relates to differences in vegetation patterns are two important steps toward understanding the relationship between burn severity, site conditions, and patterns of vegetation associated with those differences.

Among the tools to predict burn severity, dNBR satellite imagery is one of the most cost and time efficient due to the unnecessary field collection required by other tools such as CBI (Cocke et al. 2005, Zhu et al. 2006). Validating the use of dNBR regionally is an important step in understanding the effectiveness of using burn severity indices such as satellite imagery to indicate ground-based methods such as CBI (MIller et al 2009, Key and Benson 2006). My study validated the use of dNBR to remotely determine burn severity on xeric sites in GSMNP. Results from my study suggest dNBR can effectively determine burn severity across xeric forest types, between one and ten years post-fire, and among different layers of forest strata. Being able to identify post-fire burn severity has many advantages for managers such as locating areas for potential restoration efforts, evaluating the effects of prescribed burns, and determining vegetation responses to burn severity pattern.

Understanding differences in vegetation patterns for different burn severities is a key step in the development of fire management goals (Brown et al. 2014). If Park managers can use burn severity maps to determine post-fire burn severity zones and differences in vegetation patterns exist among burn severity categories, then burn severity maps have the potential to be used to predict post-fire vegetation patterns in GSMNP (Key and Benson 2006, Cocke et al. 2005). Several studies indicated that burn severity maps can be used to estimate individual post-fire variables such as overstory mortality, which is directly related to pine and oak regeneration (French et al. 2008, Miller et al. 2009, Jenkins et al. 2011). For example, the linear relationship between transformed burn severity (dNBR) and overstory mortality (dead/total stems) measured in the field was evaluated to better understand the potential for predicting differences in forest structure and composition post-fire based on burn severity maps. The linear relationship between square root transformed dNBR and overstory mortality (%) was significant and strongly correlated ($R^2 = 0.88$, $p < 0.001$) (Figure 5-1) and suggests that dNBR may be used to potentially predict and map canopy mortality post-fire.

Understanding the relationship between burn severity and differences in vegetation patterns is important for restoring particular successional stages such as the xeric yellow pine forests in the southern Appalachians (Jenkins et al. 2011). An important fire management goal in GSMNP focuses on restoration of pine and oak dominated stands occurring on xeric sites

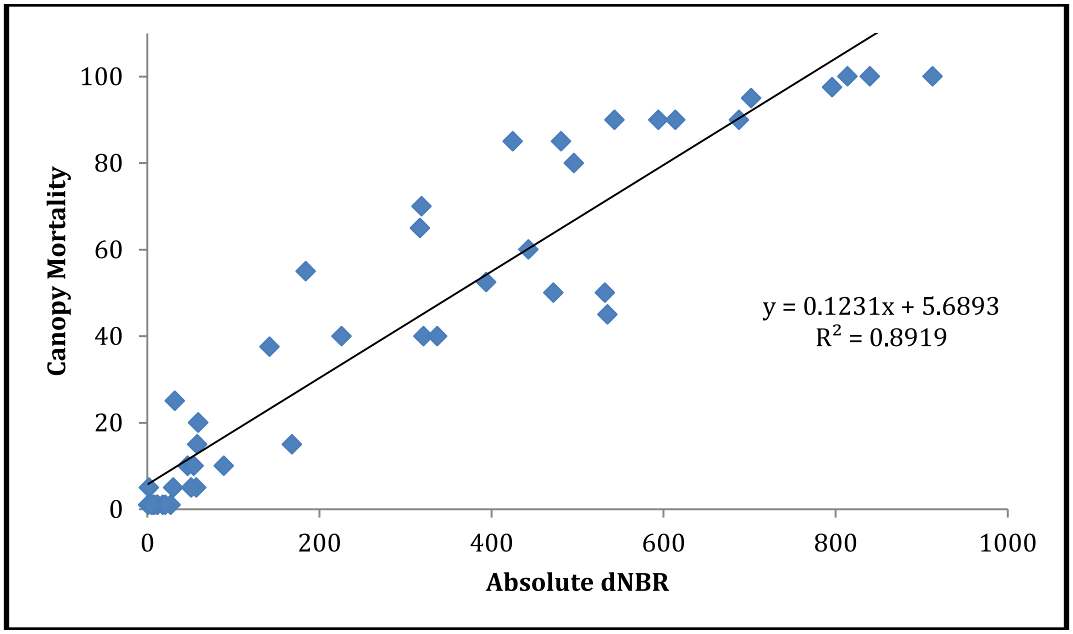


Fig. 5-1. Regression between square-root transformed absolute dNBR values and estimated overstory canopy mortality

(Robert Klein, GSMNP, personal communication), which are being replaced by fire-resistant species resulting in changes to microhabitats and subsequently changes in species composition (Jenkins et al. 2011, Nowacki and Abrams 2008). The decline of fire-dependent communities in the GSMNP, such as yellow pine and dry-site oak forests, is a major concern for Park managers and successful implementation of fire management activities can help maintain and restore these communities (Jenkins et al. 2011, Harmon 1984). Maintenance and perpetuation of yellow pine and dry-site oak species depends on the creation of site conditions suitable for regeneration such as open canopies, well-spaced trees, and relatively thin ($\leq$ 4 cm) litter layers (Callaway et al. 1987, Jenkins et al. 2011). Results from my study suggest that there are differences in vegetation patterns among burn severity and single moderate severity fires occurring on xeric sites in GSMNP are associated with conditions that may favor pine and oak regeneration over other species by opening forest canopies, reducing competition from mesophytic species, and reducing accumulated litter layers from an absence of fire.

Further studies evaluating the use of dNBR as a predictor of ecological fire effects would improve the ability to effectively plan and implement resource conservation and restoration activities by reducing the cost of data collection, more accurately monitoring patterns in burn severity, and by efficiently identifying social, ecological, and economical risks associated with fire.

CHAPTER SIX: LITERATURE CITED

Arthur, M.A., R.D. Paratley, and B.A. Blankenship. 1998. Single and repeated fires affect survival and regeneration of woody and herbaceous species in an oak-pine forest. *The Journal of the Torrey Botanical Society.* 125(3):225-236

Atwood, C.J., T.R. Fox, and D.L. Loftis. 2009. *Forest Ecology and Management* 257:1305-1313

Beckage, B. and J.S. Clark. 2003. Seedling survival and growth of three forest tree species: the role of spatial heterogeneity. *Ecology* 84:1849-1861

Brose, P.H., D.C. Dey, R.J. Phillips, and T.A. Waldrop. 2013. A meta-analysis of the fire-oak hypothesis: does burning promote oak reproduction in eastern North America?. *Forest Science* 59(3)

Brown, D.J., W.H. Nowlan, E. Ozel, I. Mali, D. Episcopo, M.C. Jones, and M.R.J. Forstner. 2014. Comparison of short-term low, moderate, and high severity fire impacts to aquatic and terrestrial ecosystem components of southern USA mixed pine/hardwood forest. *Forest Ecology and Management* 312: 179-192

Cain, M.D. and M.G. Shelton. 1995. Thirty-eight years of autogenic, woody understory dynamics in a mature, temperate pine-oak forest. *Canadian Journal of Forest Research* 25:1997-2009

Callaway, R.M., E.E.C. Clebsch, and P.S. White. 1987. A multivariate analysis of forest communities in the western Great Smoky Mountains National Park. *The American Midland Naturalist* 118:107-120

Cansler, C.A. and D. McKenzie. 2012. How robust are burn severity indices when applied in a new region? Evaluation of alternate field-based and remote-sending methods. *Remote Sensing Environment* 4:456-483

Chiang, J., M.A. Arthur, B.A. Blankenship. 2005. The effect of prescribed fire on gap fraction in an oak forest understory on the Cumberland Plateau. Journal of the Torrey Botanical Society 132(3):432-441

Chuvieco, E., A.I. Riano, and D. Cocero. 2002. Estimation of fuel moisture content from multiemporal analysis of Landsat Thematic Mapper Reflectance data: applications in fire danger assessment. *International Journal of Remote Sensing* 23:2145-2162

Cocke, A.E., P.Z. Fule, and J.E. Crouse. 2005. Comparison of burn severity assessments using Differenced Normalized Burn Ratio and ground data. *International Journal of Wildland Fire* 14:189-198

DeBano LF, Neary DG, Folliot PF (1998). *Fire's effects on ecosystems.* John Wiley and Sons, New York, NY

Delcourt, H.R., and P.A. Delcourt. 1997. Pre-Columbian Native American use of fire on southern Appalachian landscapes. *Conservation Biology* 11:1010-1014

Deluca, T.H., and A. Sala. 2006. Frequent fire alters nitrogen transformations in ponderosa pine stands of the inland northwest. Ecology 87(10):2511-2522

Dumas, S., H.S. Neufeld, and M.C. Fisk. 2007. Fire in a thermic oak-pine forest in Linville Gorge Wilderness Area, North Carolina: importance of the shrub layer to ecosystem response. *Castanea* 72:92-104

Elliot, K.J., L.R. Boring, W.T. Swank, and B.R. Haines. 1997. Successional changes in plant species diversity and composition after clear cutting a Southern Appalachian watershed. *Forest Ecology and Management* 92:67-85

Elliott, K.J. and J.M. Vose. 2005. Effects of understory prescribed burning on shortleaf pine (*Pinus echinata* Mill.)/Mixed Hardwood Forests. *The Journal of the Torrey Botanical Society* 132:236-251

Epting, J., and D. Sorbel. 2005. Evaluation of remotely sensed indices for assessing burn severity in interior Alaska using Landsat TM and ETM+. *Remote Sensing of Environment* 96;328-339

Fesenmyer, K.A. and N.L. Christensen. 2010. Reconstructing Holocene fire history in southern Appalachian forest using soil charcoal. *Ecology* 91:662-670

Fralish, J.S., F.B. Crooks, J.L. Chambers, and F.M. Hardy. 1991. Comparison of pre-settlement, second-growth and old-growth forests on six site types in the Illinois Shawnee Hills. American Midland Naturalist 125(2):284-309

French, N.H.F., E.S. Kaisischke, R.J. Hall, K.A. Murphy, D.L. Verbyla, E.E. Hoy, and J.L. Allen. 2008. Using Landsat data to assess fire and burn severity in the North American boreal forest region: an overview and summary of results. *International Journal of Wildland Fire* 17:443-462

Fuller, D.O., and M. Fulk. 2001. Burned area in Kalimantan, Indonesia, mapped with NOAA-AVHRR and Landsat TM imagery. *International Journal of Remote Sensing* 22:691-697

Gilbert, N.L., S.L. Johnson, S.K. Gleeson, B.A. Blankenship, and M.A. Arthur. 2003. Effects of prescribed fire on physiology and growth of *Acer rubrum* and *Quercus* spp. seedlings in an oak-pine forest on the Cumberland Plateau, KY. *Journal of the Torrey Botanical Society* 130:253-264.

Golden, M.S. 1981. An integrated multivariate analysis of forest communities of the Great Smoky National Park. *The American Midland Naturalist* 106:37-53

Gottschalk, K.W. 1985. Effects of shading on growth and development of northern red oak, black oak, black cherry, and red maple seedlings I: Height, diameter, and root/shoot ratio. P. 189-195 in *Proc. of the 5th Central hardwoods forest conference.* University of Illinois Press, Urbana-Champaigne, IL.

Hall, R.J., J.T. Freeburn, W.J. de Groot, J.M. Pritchard, T.J. Lynham, and R. Landry. 2008. Remote sensing of burn severity: experiences from western Canada boreal fires. *International Journal of Wildland Fire* 17:476-489

Harmon, M.E. 1982. Fire history of the westernmost portion of Great Smoky Mountains National Park. *Bulletin of the Torrey Botanical Club* 109:74-79

Harmon, M.E. 1984. Survival of trees after low-intensity surface fires in Great Smoky Mountains National Park. *Ecology* 65:796-802

Harrod, J.C., M.E. Harmon, and P.S. White. 2000. Post-fire succession and 20th century reduction in fire frequency on xeric southern Appalachian sites. *Journal of Vegetation Science* 11:465-472

Harrod JC and White RD. (1999). Age structure and radial growth in xeric pine-oak forests in western Great Smoky Mountains National Park. *Bulletin of the Torrey Botanical Society* 126:139-146

Harrod, J.C., P.S. White, and M.E. Harmon. 1998. Changes in xeric forests in western Great Smoky Mountains National Park, 1936-1995. *Castanea* 63:340-360

Hessl, A.E., T. Saladyga, T. Schuler, P. Clark, and J. Wixom. 2011. Fire history from three species on a central Appalachian ridgetop. *Canadian Journal of Forest Research* 41:2031-2039

Hubbard, R.M., J.M. Vose, B.D. Clinton, K.J. Eliott, and J.D. Knoepp. 2004. Stand restoration burning in oak-pine forests in the southern Appalachians: effects on aboveground biomass and carbon and nitrogen cycling. *Forest and Ecology Management* 190:311-321

Hutchinson, T.F., D.A. Yaussy, R.P Long, J. Rebbeck, and E.K. Sutherland. 2012. Long-term (13 years) effects of repeated prescribed fires on stand structure and tree regeneration in mixed-oak forests. *Forest Ecology and Management* 286:87-100

Hutchinson, T.F., R.P. Long, R.D. Ford, and E.K. Sutherland. 2008. Fire history and maples in second-growth forests. *Canadian Journal of Forest Research* 38:1184-1198

Intergovernmental Panel on Climate Change. (2007). Climate change 2007: the physical science basis. Summary for Policymakers. IPCC secretariat, World Meteorologist Organization, Geneva, Switzerland.

Iverson, L.R., T.F. Hutchinson, A.M. Prasad, and M.P. Peters. 2008. Thinning, fire, and oak regeneration across a heterogeneous landscape in the eastern US: 7-year results. *Forest Ecology Management* 225:3135-3050

Jenkins, M.A. 2007. Vegetation Communities of Great Smoky Mountains National Park. *Southern Naturalist* 6:35-56

Jenkins, M.A., R.N. Klein, and V.L. McDaniel. 2011. Yellow pine regeneration as a function of fire severity and post-burn stand structure in the southern Appalachians Mountains. *Forest Ecology Management* 262:681-691

Kasischke, E.S., M.R. Turetsky, R.D. Ottmar, N.F. French, E.E. Hoy, and E.S. Kane. 2008. Evaluation of the composite burn index for assessing fire severity in Alaskan black spruce forests. International Journal of Wildland Fire 17(4):551-526

Keeley J.E., Brennan T., and Pfaff H. (2008). Fire severity and ecosystem responses following crown fires in California shrubland. *Ecological Applications* 18:1530-46

Key, C.H., N.C. Benson. 2002. Measuring remote and sensing of burn severity. US Geological Survey Wildland Fire Workshop, 31 October to 3 November 2000, Los Alamos, NM. USGS Open-File Report 02-11.

Key, C.H. and N.C. Benson. 2006. Landscape assessment: ground measure of severity, the Composite Burn Index, and remote sensing of severity, the Normalized Burn Index. In ‘FIREMON: Fire Effects Monitoring and Inventory System’. (Eds D.C. Lutes, R.E. Keane, N.C. Benson, S. Sutherland, and L.J. Gangi) USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-164-CD: LA1-51. Ogden, UT.

Lafon, C.W., J.D. Waldron, D.M. Cairns, M.D. Tchakerian, R.N.Coulson, and K.D. Klepzig. Modeling the effects of fire on the long-term dynamics and restoration of yellow pine and oak forests in the southern Appalachian Mountains. *Restoration Ecology* 15:400-411

Lentile, L.B., Z.A. Holden, A.M.S. Smith, M.J. Falkowski, A.T. Hudak, P. Morgan, S.A. Lewis, P.E. Gessler, and N.C. Benson. 2006. Remote sensing techniques to assess active fire characteristics and post-fire effects. *International Journal of Wildland Fire* 15:319-345

Lutes, D.C., R.E. Keane, J.F. Caratti, C.H. Key, N.C. Benson, S. Sutherland, and L.J. Gangi. 2006. ‘FIREMON: The fire effects monitoring and inventory system.’ USDA Forest Service, Rocky Mountain Research Station General Technical Report RMRS-GTR-164-CD. Fort Collins, CO.

MacKenzie, M.D., and P.S. White. 1998. The vegetation of the Great Smoky Mountains National Park: 1935-1938. *Castanea* 63:346-360

Madden, M., R. Welch, T. Jordan, P. Jackson, R. Seavey, and J. Seavey. 2004. Digital vegetation maps for the Great Smoky Mountains National Park. Center for Remote Sensing and Mapping Science, University of Georgia, Athens, GA.

Meng, Q., R.K. Meentemeyer. 2011. Modeling of multi-strata forest fire severity using Landsat TM Data. *International Journal of Applied Earth Observation and Geoinformation* 13:120-126

McGrath, J.C. and W.K. Clatterbuck. In: Gulden, J.M., ed. 2013. Proceedings of the 15th biennial southern silvicultural research conference. E-Gen. Tech. Rep. SRS-GTR-175. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 81-90.

Millar C.I., Stephenson N.L., Stephens S.L. (2009). Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications, 17(8), 2145-2151

Miller, F.H. 1938. Brief narrative descriptions of the vegetation types in the Great Smoky Mountains National Park. Report on file in the archives of the Great Smoky Mountains National Park, Gatlinburg, Tennessee.

Miller, J.D. and A.E. Thode. 2007. Quantifying burn severity in a heterogeneous landscape with at relative version of the Normalized Burn Ratio (dNBR). *Remote Sensing of Environment* 109:66-80

Miller, J.D., H.D. Safford, M. Crimmins, and A.E. Thode. 2009b. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. *Ecosystems* 12:16-32

Miller, J.D., E.E. Knapp, C.H. Key, C.N. Skinner, C.J. Isbell, R.M. Creasy, and J.W. Sherlock. 2009a. Calibration and validation of the relative differenced Normalized Burn Ratio (RdNBR) to three measures of fire severity in the Sierra Nevada and Kalamath Mountains, California, USA. *Remote Sensing of Environment* 113:645-656

Murphy, K.A., J.H. Reynolds, and J.M. Koltum. 2008. Evaluating the ability of the differenced normalized burn ratio (dNBR) to predict ecologically significant burn severity in Alaskan boreal forests. I*nternational Journal of Wildland Fire* 17:490-499

Nowacki, G.J., M.D. Abrams. 2008. The demise of fire and “mesophication” of forests in the eastern United States. *Bioscience* 58:123-138

Phillips, D.L. and D.J. Shure. 1990. Patch-size effects on early succession in southern Appalachian forests. *Ecology* 71:204-212

Posada, D. and T.R. Buckley. 2004. Model selection and model averaging in phylogenetics: advantages of akaike information criterion and Bayesian approaches over likelihood ratio tests. *Systematic Biology* 53(5):793-808

R Core Team (2012). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria, ISBN 3-900051-07-0, URL http://www.R-project.org/

Reilly, M.J., M.C. Wimberly, and C.L. Newell. 2006. Wildfire effects on plant species richness at multiple spatial scales in communities of the southern Appalachians. *Journal of Ecology* 94:118-130

Reilly, M.J., T.A. Waldrop, and J.J. O'Brien. 2012. Fuels management in the southern Appalachian Mountains. In: Cumulative watershed effects of fuel management in the Eastern United States. USDA Forest Service GTR-SRS-161

Rentch, J.S., M.A. Fajvan, and R.R. Hicks, Jr. 2003. Oak establishment and canopy accession strategies in five old-growth stands in the central hardwood forest region. *Forest Ecology and Management* 184:285-297

Rollins, M.G., R.E. Keane, and R.A. Parsons. 2004. Mapping fuels and fire regimes using remote sensing, ecosystem simulation, and gradient modeling. *Ecological Applications* 14:75-95

Runkle, J.R., 1982. Patterns of disturbance in some old-growth mesic forests of the eastern North America. *Ecology* 63:1533-1546

Schumacher, S. and H. Bugmann. 2006. The relative importance of climatic effects, wildfires and management for future forest landscape dynamics of the Swiss Alps. Global Change Biology 12.8:1435-1450

Shugart, H.H. 1984. *A Theory of Forest Dynamics*. Springer-Verlag, NY

Shure, D.J. and L.A. Wilson. 1993. Patch-size effects on plant phenolics in successional openings of the southern Appalachians. *Ecology* 74:55-67

Skinner CN, Burk JH, Barbour MG, Franco-Vizcaino E, Stephens SL. (2008). Influences of climate on fire regimes in montane forests of northwestern Mexico. Journal of Biogeography 35, 1436-1451

Soverel, N.O., D.D.B. Perrakis, and N.C. Coops. 2010. Estimating burn severity from Landsat dNBR and RdNBR indices across western Canada. *Remote Sensing of Environment* 114:1896-1909

Stephens, S.L., J.D. McIver, R.E. Boerner, C.J. Fettig, J.B. Fontaine, B.R. Hartsough, P.L. Kennedy, and D.W. Schwilk. 2012. The effects of forest fuel-reduction treatments in the United States. *Bioscience* 62:549-559

van Wagtendonk, J.W., R.R. Root, and C.H. Key. 2004. Comparison of AVIRIS and Landsat ETM+ detection capabilities for burn severity. *Remote Sensing of Environment* 92:397-408

Waldron, J.D., C.W. Lafon, R.N. Coulson, D.M. Cairns, M.D. Tchakerian, A. Birt, and K.D. Klepzig. 2007. Simulating the impacts of southern pine beetle and fire on the dynamics of xerophytic pine landscapes in the southern Appalachians. *Applied Vegetation Science* 10:53-64

Waldrop, T.A. and P.H. Brose. 1999. A comparison of fire intensity levels for stand replacement of table mountain pine (*Pinus pungens,* Lamb.). *Forest Ecology and Management* 113:155-166

Waldrop, T.A., D.A. Yaussy, R.J. Phillips, T.F. Hutchinson, L. Brudnak, and R.E.J. Boerner. 2008. Fuel reduction treatments affect stand structure of hardwood forests in western North Carolina and southern Ohio, USA. *Forest Ecology and Management* 255:3117-3129

Ware S, Frost C and Doerr PD. (1993). Southern mixed hardwood forest: The former longleaf pine forest. Pages 447-493 In: Martin, WH, Boyce, SG, and Echternacht, A.C. (eds.) *Biodiversity of the southeastern United States: Lowland terrestrial communities*. John Wiley and Sons, New York, NY

Welch, N.T., T.A. Waldrop, and E.R. Buckner. 2000. Response of southern Appalachian table Mountain pine (*Pinus pungens*) and pitch pine (*P. rigida*) stands to prescribed burning. *Forest Ecology and Management* 136:185-197

Whittaker, R.H. 1956. The vegetation of Great Smoky Mountains. *Ecological Monogr*aphs 26:1-80

Wimberly, M.C., M.A. Cochrane, A.D. Baer, and K. Pabst. 2009. Assessing fuel treatment effectiveness using satellite imagery and spatial statistics. *Ecological Applications* 19:1377-1384

Wimberly, M.C. and M.J. Reilly. 2007. Assessment of fire severity and species diversity in the southern Appalachians using Landsat TM and ETM+ imagery. *Remote Sensing of Environment* 108:1212-1225

Zhu, Z., C. Key, D. Ohlen, and N. Benson. October 12, 2006. Evaluate sensitivities of burn-severity mapping algorithms for different ecosystems and fire histories in the United States. Final Report to the Joint Fire Science Program. JFSP 01-1-4-12

APPENDIX I

Table A-1. Species codes, common name, and Latin name for overstory species.

Species Code	*Common Name*	*Latin Name*
ACPE	striped maple	*Acer pennsylvanicum*
ACRU	red maple	*Acer rubrum*
BETU	birch	*Betula spp.*
CARY	hickory	*Carya spp.*
COFL	flowering dogwood	*Cornus florida*
FAGR	American beech	*Fagus grandifolia*
HATE	silverbells	*Halesia tetraptera*
JUVI	eastern red cedar	*Juniperus virginiana*
LITU	tulip-poplar	*Liriodendron tulipifera*
MAFR	frasier magnolia	*Magolia fraserii*
NYSY	black gum	*Nyssa sylvatica*
OXAR	sourwood	*Oxydendron arboreum*
PIRI	pitch pine	*Pinus rigida*
PIST	eastern white pine	*Pinus strobus*
PIVI	Virginia pine	*Pinus virginiana*
PRSE	black cherry	*Prunus serotina*
QUAL	white oak	*Quercus alba*
QUCO	scarlet oak	*Quercus coccinea*
QUMO	chestnut oak	*Quercus montana*
QURU	northern red oak	*Quercus rubra*
QUVE	black oak	*Quercus velutina*
ROPS	black locust	*Robinia pseudoacacia*
SAAL	sassafras	*Sassafras albidum*
TSCA	eastern hemlock	*Tsuga canadensis*

Table A-2. Understory density for species groups yellow pine, dry site oak, and mixed mesophytic species.

	Understory Density (stems/ha)			
Species Group	*Unburned*	*Low Severity*	*Mod. Severity*	*High Severity*
Yellow pine	522	1107	2520	2164
Dry-site oak	1909	2674	7894	3692
Mixed Mesophytic	10186	5538	7003	7321
Total	12617	9319	17417	13177

Table A-3. Overstory density for individual species (Table A-1) across different burn severities

	Overstory Density (stems/ha)			
Species	Unburned	Low Severity	Mod. Severity	High Severity
ACPE	4.71	0.00	0.00	0.00
ACRU	96.62	38.89	86.13	23.14
BETU	2.36	7.07	0.00	0.00
CARY	16.50	38.89	20.57	0.00
COFL	4.71	0.00	0.00	0.00
FAGR	23.57	0.00	0.00	0.00
HATE	7.07	0.00	0.00	0.00
JUVI	0.00	0.00	3.86	0.00
LITU	35.35	0.00	5.14	0.00
MAFR	4.71	0.00	6.43	0.00
NYSY	35.35	127.26	78.41	24.42
OXAR	58.92	30.05	47.56	9.00
PIRI	44.78	42.42	73.27	32.14
PIST	25.92	0.00	18.00	0.00
PIVI	40.06	21.20	65.56	32.14
PRSE	0.00	0.00	1.29	0.00
QUAL	9.43	10.61	0.00	3.86
QUCO	25.92	33.58	53.99	12.85
QUMO	94.27	77.80	71.99	21.85
QURU	4.71	0.00	7.71	0.00
QUVE	0.00	10.61	10.28	0.00
ROPS	2.36	0.00	0.00	2.57
SAAL	0.00	1.77	0.00	0.00
TSCA	9.43	0.00	6.43	0.00
TOTALS	546.75	440.15	556.62	161.97

Table A-4. Overstory dominance for individual species (Table A-1) across different burn severities

	Overstory Dominance (m^2/hectare)			
Species	Unburned	Low Severity	Mod. Severity	High Severity
ACPE	0.00	0.00	0.00	0.00
ACRU	15.71	2.55	9.76	0.71
BETU	0.00	0.00	0.00	0.00
CARY	0.28	1.98	0.28	0.00
COFL	0.00	0.00	0.00	0.00
FAGR	0.85	0.00	0.00	0.00
HATE	0.14	0.00	0.00	0.00
JUVI	0.00	0.00	0.00	0.00
LITU	3.68	0.00	0.00	0.00
MAFR	0.00	0.00	0.00	0.00
NYSY	1.27	13.87	5.38	0.71
OXAR	6.37	1.56	3.11	0.14
PIRI	14.72	9.91	18.25	0.42
PIST	3.40	0.00	0.71	0.00
PIVI	4.81	1.42	10.61	2.12
PRSE	0.00	0.00	0.00	0.00
QUAL	0.57	0.14	0.00	0.00
QUCO	2.69	4.53	7.92	0.57
QUMO	26.60	16.13	12.03	1.27
QURU	0.00	0.00	0.28	0.00
QUVE	0.00	0.42	0.28	0.00
ROPS	0.00	0.00	0.00	0.00
SAAL	0.00	0.00	0.00	0.00
TSCA	0.00	0.00	0.00	0.00
TOTALS	81.08	52.50	68.63	5.94