FIRE ECOLOGY AND FIRE MANAGEMENT IN THE SOUTHERN APPALACHIANS: RATIONALE FOR AND EFFECTS OF PRESCRIBED FIRE

Naomi Beth Schwartz

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Approved by:

Aaron Moody

Peter White

Dean Urban

Abstract

NAOMI BETH SCHWARTZ: FIRE ECOLOGY AND FIRE MANAGEMENT IN THE SOUTHERN APPALACHIANS: RATIONALE FOR AND EFFECTS OF PRESCRIBED FIRE
(Under the direction of Aaron Moody)

Fire suppression in the Southern Appalachians has led to changes in forests dominated by yellow pine (*Pinus* subgenus *pinus*) and oak (*Quercus*) species. Recently, management agencies have begun to prescribe fire with the aim of restoring pre-suppression conditions. Here, I examine the use of prescribed fire in the Southern Appalachians from two perspectives. First, I review the values and goals that underlie fire management, how they apply in the Southern Appalachians, and what the implications of these are for fire management planning. Second, I use long-term monitoring data to examine how prescribed fire affects forest structure and composition in the Great Smoky Mountains National Park and how these effects vary with environment and fire severity. I find that prescribed fire creates conditions conducive for pine reproduction and is particularly effective at high severity and at lower elevation sites where fire sensitive species are still confined to smaller size classes.

Dedicated to my grandmother, Lillian Schwartz.

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Table of Contents

List of Tables

List of Figures

List of Abbreviations

DBH: diameter at breast height

GSMNP: Great Smoky Mountains National Park

CHAPTER 1:

INTRODUCTION

Prior to European settlement, Native American use of fire was widespread throughout North America, and left a lasting imprint on the North American landscape (Pyne 1982). Changes to these fire regimes due to land-use change and active fire suppression, among other drivers, have resulted in changes to ecosystem structure, composition, and diversity, and today, many managers aim to reverse or mitigate these changes through the use of prescribed fire and other fire management techniques. This is true in the Southern Appalachians, where Native Americans used fire for at least the past 4000 years (Delcourt and Delcourt 1997, 1998, Brose et al. 2001, Fesenmyer and Christensen 2010). These fires created open, park-like conditions favorable for establishment and regeneration of yellow pine (*Pinus* subgenus *pinus*) and oak (*Quercus*) species (Harrod et al. 1998). Today, forests dominated by these fire-adapted species remain on the landscape as a legacy of past Native American burning.

A period of fire suppression in the Southern Appalachians began in the twentieth century. As a result, yellow pines and oaks have stopped regenerating on many sites where they were once dominant, and red maple (*Acer rubrum*), white pine (*Pinus strobus*), and other thin-barked, fire-sensitive species have begun to invade (Harmon 1984, Harrod et al. 1998, Harrod and White 1999). In some parts of the Southern Appalachians, as many as 98% of stands previously dominated by pine have little or no remaining pine (Vose et al. 1995).

Recently, management agencies in the Southern Appalachians such as the National Park Service and the US Forest Service have begun to prescribe fire with the aim of restoring pre-suppression conditions. These efforts are relatively recent, with the first prescribed fire in the Great Smoky Mountains National Park occurring in 1997, and their long-term effects have yet to be established. Resources for prescribed fire are limited, and there is likely not the capacity to restore fire to the Southern Appalachians at the frequency and extent that it would have been present under Native American fire regimes. Therefore, managers must prioritize areas in which to focus prescribed fire efforts. These decisions are motivated by the values and goals underlying conservation and management efforts and should also be based in the scientific understanding of the role of fire in the Southern Appalachians and of the effects of prescribed fire.

In this thesis, I address the use of prescribed fire in the Southern Appalachians from two perspectives. In the first chapter, I examine how the various values and goals that can underlie fire management apply to the Southern Appalachians, and what the implications of these goals are for fire management policies and planning. Different sets of goals can have different implications for management, and in designing and implementing fire management plans, managers should be explicit about what their goals are. In the second chapter, I analyze the effects of prescribed fire in the Great Smoky Mountains National Park on forest structure and composition. I use long-term data collected from 1997 until 2011 across 39 plots and 21 fires to assess how forest communities change after fire, and examine how environmental variables and fire severity affect the magnitude of this change. Most previous research on the effects of prescribed fire in the Southern Appalachians has focused on the short-term effects of just one prescribed fire; my thesis goes beyond these studies by comparing fire effects across multiple prescribed fires and over a longer time period. Together, these chapters aim to provide more insight into why and how managers should use prescribed fire in the Southern Appalachians, and the findings may help managers decide how and where to focus often-limited resources for fire management in the region.

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CHAPTER 2:

RATIONALE FOR FIRE IN CONSERVATION MANAGEMENT: A CASE STUDY OF THE SOUTHERN APPALACHIANS

Introduction

Fire is an important driver of ecosystem structure, composition, and function in many ecosystems worldwide (Bond and Keeley 2005, Bowman et al. 2009, Pausas and Keeley 2009). However, fire regimes and the functional role of fire vary from place to place (Krawchuk et al. 2009), and even within a single ecosystem, the role of fire is often complex and variable. Fire management planning is likewise not straightforward: plans can vary significantly depending on the goals articulated by managers. Articulating these goals and discussing their basis in science and policy is critical. In this paper, I briefly review six alternative, though not mutually exclusive, bases for fire management and then turn my attention to an interesting case: fire in the Southern Appalachians.

Fire is used to accomplish six general management goals (table 1). First, in systems where altered fire regimes have led to major changes in fuel abundance, vertical structure, and spatial pattern, managers may use prescribed fire to reduce the risk of catastrophic fire and associated ecological damage, including abrupt and permanent shifts in ecosystem structure (Kloor 2000, Allen et al. 2002). Second, in some ecosystems, fire is used to maintain, restore, or otherwise manipulate ecosystem processes such as carbon and nitrogen cycling, soil formation, and hydrology (Neary et al. 2005). Third, fire management also uses fire to conserve biodiversity by improving habitat for endangered species (e.g. the red cockaded woodpecker; James et al. 1997), promoting regeneration for species adapted to fire (Angelstam 2009), or promoting landscape heterogeneity (Brockett et al. 2001). Fourth, fire management is used to restore “natural” processes and ecosystems. Fire has burned vegetation since shortly after the appearance of the first terrestrial plants (Scott and Glasspool 2006) and has played an important role in shaping the characteristics and distributions of global vegetation (Bond and Keeley 2005). Therefore fire is necessary to maintain some ecosystems in their natural states, and as fire regimes change with human influence, deliberate fire management is necessary. Fifth, humans have used fire for a variety of purposes for millennia with widespread ecological impacts (Bowman et al. 2011), and fire may be used to maintain historic anthropogenic conditions. Finally, despite the paradigm shift towards using fire to maintain natural ecosystems, the major goal of fire management is still, in many places, to minimize damage to human life and property (Haines et al. 1998, Fernandes and Botelho 2003).

Fire is important for the establishment and maintenance of forests dominated by yellow pines (*Pinus* subgenus *diploxylon)* in the Southern Appalachians. These forests have changed as a result of fire suppression over approximately the last 70-80 years, and yellow pines are in decline (Vose et al. 1999). In some parts of the Southern Appalachians, as many as 98% of stands previously dominated by pine have little or no remaining pine (Vose et al. 1995). Recently, fire management efforts have taken root around the Southern Appalachians, with agencies such as the National Park Service and US Forest Service, and other land management organizations using prescribed fire in an attempt to restore pine dominance and promote pine regeneration. Here, I examine to how the five goals described above apply to fire management in pine-dominated forests (including sites both currently and historically dominated by pine) or on sites presumed to be appropriate for pine in the Southern Appalachians, and what the implications of each is for management planning.

Fire and pines in the Southern Appalachians

Pine and mixed pine-oak forests in the Southern Appalachians are found on xeric exposed slopes and ridgetops at mid-elevations, and are dominated by the yellow pine species pitch pine (*Pinus rigida*), shortleaf pine (*Pinus echinata*), Virginia pine (*Pinus virginiana*), and the Appalachian endemic Table Mountain pine (*Pinus pungens*), as well as oaks such as scarlet oak (*Quercus coccinea*) and chestnut oak (*Quercus montana*) (Whittaker 1956, Jenkins et al. 2011). Fire promotes pine and oak regeneration in these forests by killing shade tolerant, fire sensitive species such as red maple (*Acer rubrum*), black gum (*Nyssa sylvatica*), Eastern hemlock (*Tsuga canadensis*), and white pine (*Pinus strobus*) and by creating high light and low litter conditions conducive to germination and establishment of yellow pines (Barden and Woods 1976, Harrod et al. 2000). Yellow pines display several adaptations to fire: they have relatively thick bark, are shade intolerant, and regenerate only when litter and duff are reduced (Harmon 1984, Jenkins et al. 2011). In addition, *Pinus pungens* is partially serotinous, and will release seeds after fire (Zobel 1969).

Historically, fires in Southern Appalachian pine-oak forests were frequent, with dendrochronological studies finding average fire return intervals between 3 and 13 years (White 1987, Harmon 1982, Aldrich et al. 2010, Feathers 2010). This fire regime was characterized by frequent, low-intensity surface fires, which maintained an open, well spaced canopy and shallow litter and duff layers, and occasional severe fires, often following another disturbance such as a southern pine beetle outbreak or ice storm, that promoted episodes of pine recruitment (figure 1; White 1987, Lafon and Kutac 2003, Brose and Waldrop 2006, Aldrich et al. 2010). Lightning fire in the region is too infrequent to account for the observed pre-suppression fire frequency, unless fire sizes were much larger in the past (Barden and Woods 1976, Harmon 1982, Lafon and Grissino-Mayer 2007). Fire appears in the Southern Appalachian charcoal record as far back as 10,000 years ago, and it has regularly occurred since about 4000 years ago coinciding with the arrival of the Woodland tradition Native Americans in the region (Delcourt and Delcourt 1997, Fesenmyer and Christensen 2010). Therefore, anthropogenic fire likely played an important role in creating and maintaining pine forests

For about the past 75 years successful fire suppression in the Southern Appalachians has resulted in changes to vegetation structure and composition. Pine and pine-oak forests have become more dense, with higher basal area and canopy species richness than pre-fire suppression forests (Harrod et al. 1998). Fire suppression has allowed species that are thin-barked and slow growing which would have been excluded under past regimes of frequent fire to get large enough that they are resistant to fire and present in the canopy (Harmon 1984). Yellow pines and oaks are still present in the canopy of these forests; however, there are few individuals in smaller size classes suggesting that these species are failing to regenerate and that they could eventually disappear from areas where they once dominated (Harrod et al. 1998). In addition, Harrod et al. (1998) show that pine and oak canopy basal area, after peaking about 1980, is now declining. Today, southern pine beetle outbreaks, ice storms, or other non-fire disturbances are not followed by fire and episodes of pine recruitment as they would have been in the past (Brose and Waldrop 2006). As a result these disturbances now accelerate succession to hardwood forest types instead of playing a role in maintaining them (Figure 1).

Recently, managers have begun to use prescribed fire in an attempt to promote pine regeneration and restore an open structure to pine-oak forests in the Southern Appalachians. These fires have had mixed results, with effectiveness dependent largely on the severity of the prescribed fire. Elliott et al. (1999) found that while *Pinus rigida* seedlings were abundant after a prescribed fire in Nantahala National Forest, North Carolina, most died within two years after fire and *Pinus rigida* seedling density was lower two years following prescribed fire than before. This may have been caused by competition for light from shrubs like *Kalmia latifolia*, which resprouted vigorously after fire. Similarly, Welch et al. (2000) found that while *Pinus rigida* and *Pinus pungens* seedling were present after low intensity prescribed fire, shrubs and hardwoods were not exposed to lethal temperatures, and resprouted to such an extent that understory density after fire was twice that before fire. Jenkins et al. (2011) found a threshold of fire severity, measured as changes to fuels loads, overstory density, and understory density, below which no yellow pine seedlings were observed following prescribed fire in the Great Smoky Mountains National Park. These studies suggest that to successfully promote pine regeneration, prescribed fire must be hot enough to open the canopy, reduce litter and duff, and expose competitors to lethal temperatures. Simply reintroducing low-intensity surface fires will not work for ecosystem restoration. If fire suppression has continued long enough, techniques outside the "historical range of variability" (Landres et al. 1999), such as very severe fires, stand thinning, or herbicide treatment may be required to restore the ecosystem to its pre-suppression state.

Because there is ambiguity about the natural or historical role of fire in the Southern Appalachians and restoration pathways are not clear-cut, fire management must be clear about its goals. Additionally, it is important to acknowledge what is known, what is unknown, and what will unlikely ever be known for certain about fire in the Southern Appalachians; and what the implications of this knowledge, or lack there of, are for management decisions. Below, I review the previously described goals of fire management as they apply to the Southern Appalachians, in an attempt to highlight the most important considerations for fire management in the region and how our existing knowledge and unanswered questions inform decisions about where, why, and how to manage.

Goals of fire management in the Southern Appalachians

Goals 1 and 6: Prevent catastrophic fire and reduce risk to human life and property

Here, I consider management the first and sixth goals (see table 1) together. In some ecosystems, prescribed fire and other fire management activities decrease the intensity of subsequent wildfire, mitigate fire hazard, protect human life and property, and prevent catastrophic ecological changes by reducing fuel loads and disrupting vertical and horizontal fuel continuity (Haines et al. 1998, Fernandes and Botelho 2003). In these ecosystems, fire management generally aims to mitigate or reverse the effects of fire suppression in terms of changes to an ecosystem's structure and composition that increase fire hazard. For example, in *Pinus ponderosa* forests in the Southwestern US, fire suppression has allowed the development of a dense midstory which provides abundant and vertically continuous fuels and raises the likelihood of large, stand-replacing crown fires (Parsons and DeBenedetti 1979). These fires create risks for human life and property, and can result in major changes to ecosystem structure and function. In the short term, stand-replacing fires in *Pinus ponderosa* ecosystems may amplify erosion and flooding (Robichaud 2005), and in the long term they can leave treeless scars on the landscape which can persist as grasslands or shrublands for decades to centuries (Allen et al. 2002). Thus, a major goal of forest management in the Southwestern US has been to use prescribed fire and mechanical treatments to restore conditions that are within the "natural range of variability" (Landres et al. 1999, Allen et al. 2002).

In the Southern Appalachians, managing fire to reduce fuels and prevent catastrophic fire is not generally considered a major goal of fire management for a number of reasons. First, while fuels are abundant in the Southern Appalachians, humidity and fuel moisture are rarely low enough to allow for fire and when fire weather does occur, it is rarely accompanied by lightning (Barden and Woods 1976). Second, topography and vegetation in the Southern Appalachians are heterogeneous and exert strong controls on spread of fire, especially in moister regions like the Great Smoky Mountains (Flatley et al. 2011). Fire-prone vegetation is topographically restricted to south and southwest facing slopes and ridgetops (Whittaker 1956, Jenkins 2007), meaning that there is little continuity of flammable fuels across the landscape and therefore low probability of intense fire burning across large areas and producing major hazards to people or catastrophic changes to the landscape. All this means that fire suppression is relatively easy in the Southern Appalachians: even in the driest conditions, fire suppression is generally successful at limiting the extent of wildfire (Vose et al. 1995).

Last, although fire suppression in Southern Appalachian yellow pine forests does result in changes to forest structure and species composition, there is evidence that these changes in forest structure actually decrease the risk of wildfire. With fire suppression, open, fire maintained areas are converted to shady, closed canopy forests, and conditions improve for shade-tolerant, fire-sensitive, mesophytic plants. As these species increase in abundance and forest stands get more dense, microenvironmental conditions become more shady and mesic, preventing establishment of shade-intolerant, fire-adapted species, increasing average fuel moisture conditions, and decreasing the likelihood of fire in a process dubbed “mesophication” (Figure 1; Nowacki and Abrams 2008). So, in the Southern Appalachians, prescribed fire or fuel treatments might actually create conditions more favorable to wildfire than would occur if fire suppression and accompanying mesophication continued.

In the Southern Appalachians, forests stay forests. Discussions around slowing or reversing the effects of fire suppression in the Southern Appalachians lack the sense of urgency present in discourse about southwestern forests, which are more prone catastrophic fire and accompanying sudden and drastic changes to vegetation structure. As dominant yellow pines and oaks die off, other trees replace them. With the exception of some logging-related fires in the early 20th century there have been no recent fires in the Southern Appalachians that have led to conversion from forest to non-forest as can occur after fire in *Pinus ponderosa* ecosystems. Succession from pine to hardwoods in the Southern Appalachians is gradual. Still, mesophication and the shift away from pine-oak dominance may represent a shift to an alternate stable state. Restoring fire dependent vegetation once a community has become mesophytic may be prohibitively difficult and expensive (Figure 1; Abrams 2005, Nowacki and Abrams 2008). So, even though prevention of catastrophic fire is not a major goal in the Southern Appalachians, the changes that are occurring may prove to be catastrophic in the sense of being difficult or impossible to reverse, if not ecologically dramatic.

Finally, while there is not currently a major risk of dangerous of catastrophic wildfire in the Southern Appalachians, in future years this could change. Anthropogenic climate change is expected to increase fire risk in many parts of the world, including in the Southeastern US (Scholze et al. 2006). To my knowledge there have not been any detailed predictions about the effects of climate change on fire regimes in the Southern Appalachians, but changes in the amount or timing of precipitation and increases in temperature could affect the potential for wildfire. Further research into the potential effects of climate change on wildfire risk in the Southern Appalachians will be necessary to better inform long-term fire management planning.

Goal 2: Maintain ecosystem function

Fire affects physical, chemical, and biological properties of soils, hydrology, and water quality along with carbon and nutrient cycling and storage (Boerner 1982, Richter et al. 1982, Neary et al. 2005, Bowman et al. 2009). Fire management may thus be motivated by considerations related to soil conservation, nutrient cycling, carbon storage, productivity, or hydrology. To my knowledge, there have not been any studies documenting the effects of fire suppression on ecosystem processes in the Southern Appalachians. Several studies have examined the impacts of prescribed fire on biomass, carbon, and nitrogen cycling, and have found they are limited. In a study of the biogeochemical effects of a prescribed fire in the Nantahala National Forest, Vose et al. (1999) found no significant changes in carbon pools or soil and stream chemistry, and found significant nitrogen loss only in small wood and litter pools in areas that experienced the most intense fires. Similarly, Hubbard et al. (2004) found minimal effects of prescribed burning on ecosystem processes in the Consauga River watershed. Groeschl et al. (1993) found that wildfire significantly reduced nutrient contents in the forest floor and surface soil, although low severity fire had modest impacts relative to high severity fire. These studies, along with the fact that fire-prone vegetation covers less than 5 percent of the Southern Appalachian landscape (Vose et al. 1995), suggest that the effects of fire on ecosystem processes in the Southern Appalachians are likely small. Therefore, managing fire to maintain, restore, or manipulate ecosystem function in the Southern Appalachians is not currently a major concern. Still, more research as to the stand and landscape level effects of prescribed fire and fire suppression on ecosystem processes in the Southern Appalachians is needed.

Goal 3: Conserve biodiversity

Fire management decisions have implications for biodiversity at local, landscape, and regional scales, and managers may devise fire management plans to conserve biodiversity at some of all of these scales. Some species are adapted to particular fire regimes and have higher fitness under those conditions (Keeley et al. 2011); fire management may be necessary to maintain viable populations of these species. In fire-adapted ecosystems, local scale species richness can increase after fire, often due to post-fire increases in light or nutrient availability (Gilliam and Christensen 1986, Reilly et al.

2006). Fire also affects landscape diversity and heterogeneity and may help maintain a patch mosaic of successional stages or community types across a landscape (White 1979, Baker 1992). Fire management is therefore important for biodiversity conservation from several angles: to promote regeneration and maintenance of species that are more fit under a given fire regime, to maintain species rich communities, and/or to promote heterogeneity in community types or successional stages across a landscape.

Within fire dependent stands in the Southern Appalachians, fire suppression can lead to increases in tree species richness in the short term, as new hardwood species are able to establish themselves on sites where they once would have been excluded (Harrod et al. 1998). Over time, however, richness is expected to decline as fire dependent species are eventually excluded (Nowacki and Abrams 2008). Herbaceous cover and understory species richness have been found to increase immediately after fire, especially hotter, more severe fires (Groeschl et al. 1992, Harrod et al. 2000, Reilly et al. 2006, Wimberly and Reilly 2007). Some understory herbaceous species, such as *Schizachyrium scoparium* and *Solidago odora,* flower and fruit more in recently burned plots than in unburned plots (Harrod 1999).

Fire management is probably not essential for the survival of species that are adapted to fire in the Southern Appalachians. Yellow pines and co-dominant oaks will maintain themselves without fire at sites that are too dry and infertile to allow succession to hardwood forest (Whittaker 1956, Barden 1977). Even *Pinus pungens*, which is partially serotinous and requires fire to create conditions favorable for reproduction and establishment throughout most of its range (Zobel 1969), is not dependent on fire for reproduction everywhere. About 40 percent of Table Mountain cones open after two years without fire (Barden 1979), and populations of Table Mountain pine on shallow, rocky soils maintain themselves in the absence of fire (Barden 1977, 1988, 2000, Newell and Peet 1998).

Here, I refer to these xeric, infertile sites where pine will persist in the absence of fire as their "edaphic core". Yellow pine species will remain at their edaphic core in the Southern Appalachians regardless of management decisions about fire. If the goal of fire management were just to prevent regional extinctions, prescribed fire would not be absolutely necessary. There has been little research on the extent or distribution of this edaphic core, but these sites are rare relative to the entire range of conditions occupied by yellow pines in the Southern Appalachians (Waldrop and Brose 1999). Therefore, continued fire suppression and succession from pine to hardwoods on less xeric (non-core) pine-dominated sites would represent a loss of landscape diversity. Fire *is* necessary to maintain yellow pines at their current extent, and the distributions of these species would contract to only the driest and most infertile sites in the absence of continued fire.

Fire suppression in the Southern Appalachians has been going on for a relatively short period of time, and there is still a great deal that is unknown about the effect of changes in fire regimes on biodiversity in the Southern Appalachians. We may not have yet seen the full long-term effects of fire suppression in terms of changes to species composition, and the absence of fire may result in unforeseen local extinctions or more drastic changes to species composition than have been observed until now. There has been little research on the effects of fire suppression on animal diversity or habitat quality for animal species of conservation concern. Additionally, we do not know to what extent continuing fire suppression could drive loss of genetic diversity for yellow pines, oaks, and other species commonly found in pine-oak forests or loss of the capacity to adapt to climate or other environmental change.

Goal 4: Conserve 'natural' ecosystems

Fire occurs without anthropogenic ignitions in the Southern Appalachians: it has been present in the region for at least 10,000 years, and lightning fires, although currently rare, do occur (Cohen et al. 2007, Lafon and Grissino-Mayer 2007, Fesenmyer and Christensen 2010). Therefore, the use of fire to conserve "natural" ecosystems is applicable in the Southern Appalachians. However, this is complicated by two factors. First, in the Southern Appalachians, as elsewhere, 'natural' can mean different things to different people and its meaning in a particular context is often not specified. Second, even within a particular definition of natural, we lack the necessary information to know what "natural" should look like in the Southern Appalachians. Here, I consider natural to mean "free from human influence." This definition can manifest itself in a number of forms, summarized by Aplet and Cole (2010) as: 1) free from intentional human manipulation and intervention; that is, devoid of management actions meant to manipulate ecosystems, and 2) pristine and unaffected by humans, in conditions resembling those found prior to human settlement. Aplet and Cole also present a third definition of natural as true to historic conditions, which I consider as a separate case below.

Adherence to the first definition of natural would require a hands-off approach, in which managers take no actions to manipulate or control ecosystem structure, composition, or processes (Landres 2010). Under such an approach, managers would allow fires started by natural ignitions (i.e. lightning) to burn unsuppressed, but would not prescribe fire. While this would serve the purpose of preserving autonomous nature, our current understanding is that a lightning fire regime would not provide the conditions required for the maintenance of fire-dependent vegetation. The frequency of lightning fires is not enough for the maintenance of pine and oak dominance: Harmon (1982) calculated that lightning fires would have to burn 258 to 1070 times more frequently in his study area, or else be far larger than under the fire suppression regime to account for the observed pre-suppression fire rotation of 10 to 40 years. Similarly, Lafon et al. (2005) calculated a fire rotation period of 6138 years for natural fires based on fire occurrence data from 1970-2003 in the Central Appalachians. Although it seems unlikely that lightning fire could have maintained pine-oak forests, it could have been the case that despite infrequent ignitions, naturally ignited fires in the past were much larger than they are now, or that anthropogenic fire interacted with lightning fire to make it less frequent. Additionally, a truly hands-off approach is, in most contemporary landscapes, impossible. Even if we do not actively suppress fires, we manipulate the size and frequency of contemporary natural fires. Fragmentation, roads, and other built structures serve as barriers to the spread of fire (Sharitz et al. 1992, Duncan and Schmalzer 2004), decreasing the so-called "fire compartment size," the area over which fire can spread unimpeded (Frost 2006). A truly hands-off management approach seems impossible, and would result in a loss of landscape diversity, local scale species richness declines, or declines in regional species richness.

Managing to recreate ecosystems or landscapes that resemble pre-human conditions may also be impossible in the Southern Appalachians. Humans have been in the Southern Appalachians for at least 10,000 years, and have made frequent use of fire for at least 4000 (Delcourt and Delcourt 1997, 1998, Fesenmyer and Christensen 2010). There is also a long history of fire-use by European settlers (Jurgelski 2008). As far as we know, none of the fire dependent ecosystems in the Southern Appalachians could be considered pristine and unaffected by humans. Additionally, we lack a clear idea of what pre-human fire regimes and vegetation looked like in the region. There are few natural lakes in the Southern Appalachians, making charcoal and pollen studies difficult. Trees like *Pinus rigida* and *Pinus pungens* rarely live to be older than 350 and 250 years, respectively (Burns et al. 1990), precluding dendroecological analysis of pre-human conditions. Due to this dearth of information, perhaps the best approach to restore pre-human conditions would be to let naturally ignited fires burn unsuppressed, refrain from prescribed fire, and suppress anthropogenic fires.

While humans were undeniably important in shaping the fire regimes of the Southern Appalachians, there are places more prone to lightning fire and that may have burned frequently in the absence of anthropogenic fire. Recent fire history data suggests that natural fires are more common during the growing season, at lower elevations, and on southwest facing aspects (Lafon et al. 2005, Lafon and Grissino-Mayer 2007, Flatley et al. 2011). A study of unsuppressed lightning fires in the Great Smoky Mountains found that such fires may burn for relatively long durations (up to 38 days) and exhibited a range of fire behaviors (Cohen et al. 2007). If the pine-oak ecosystem is largely intact and the effects of fire suppression have not been severe, management actions focused on prescribing fire or ending fire suppression during the fire season in these areas would be most likely to maintain fire dependent vegetation in places where it was found before human settlement. These places could be considered a “fire core,” analogous to the edaphic core that exists for pine forests: places where fire, and thus fire-dependent vegetation would have historically persisted without human intervention, and could persist in the future without the aid of prescribed fire or with minimal prescribed fire. Further research should attempt to identify if such a "fire core" exists, and if so, how extensive it is. Existing information tends to suggest such a fire core, even if present, would not account for the degree of pine forest dominance in the pre-suppression era.

Goal 5: Historical fidelity

Fire management may aim to maintain or restore historic conditions in ecosystems, generally taken to mean "pre-European" or "pre-industrial." This strategy has historically been important for management of National Parks and other protected areas. The report by Leopold et al. (1963) said: "As a primary goal, we would recommend that the biotic associations within each park be maintained, or where necessary recreated, as nearly as possible in the condition that prevailed when the area was first visited by the white man. A national park should represent a vignette of primitive America." This was the thinking behind the first National Park Service policy that formally recognized the ecological importance of fire, allowed managers to let natural fires burn, and permitted the use of prescribed burning.

As mentioned above, virtually no fire dependent ecosystem in the Southern Appalachians can be considered free from human influence. Native American fires were common throughout the Eastern United States, including in the Southern Appalachians (Abrams 1992), and fire has been especially frequent in the last 1000 years, since the arrival of the Woodland tradition Native Americans (Fesenmyer and Christensen 2010). Native Americans probably used fire to maintain hunting and gathering grounds, and most fires set by Native Americans were probably low-intensity surface fires (Delcourt and Delcourt 1997, 1998, Brose et al. 2001). However, the overall extent of Native American burning relative to the pre-human, "natural" distribution of fire and the degree to which Native American burning is responsible for pine-oak forests' current extent are not well understood. While most agree that at least some of these forests were created and maintained by Native American fire, some have originated relatively recently and could be artifacts of fires caused by early European settlers or industrial logging (Williams and Johnson 1990, Williams 1998). This confusion may undermine attempts to recreate a "vignette of primitive America" at the landscape scale, and future research into how much pine forest is a result of Native American burning vs. other sources could help better inform management in National Parks and elsewhere.

Historical fidelity may also refer to pre-industrial (but not necessarily pre-European) conditions. European settlers in the pre-industrial Southern Appalachians used fire for a variety of purposes, including: to improve common grazing lands, to improve conditions for hunting and gathering, for pest and disease prevention, or for entertainment and revenge (Jurgelski 2008). These fires created open, park-like forests with grassy or herbaceous understories, or in some cases, areas completely devoid of trees (Jurgelski 2008). Despite changes in the use of fire, fire regimes under early European settlement may not have been all that different from Native American fire regimes. Aldrich et al. (2010) use dendrochronological data over a 350 year period to demonstrate that the changes in land- and fire-use that came with European settlement did not cause significant changes to fire frequency, meaning that pre-industrial use of fire in the

Southern Appalachians would have resulted in vegetation with similar characteristics to that created by Native American fire.

Practical implications

Of the six fire management goals, the ones that are most relevant in the Southern Appalachians are managing for biodiversity, in particular landscape diversity, for naturalness, and for historical fidelity (Table 1). Each has different implications for management, but none of them are mutually exclusive and often, application of one requires careful examination and analysis of the others.

Managing for biodiversity first requires the definition of a metric of interest (local or regional species richness, landscape diversity, number of populations of fire-associated species, etc.) and a target level or state for the chosen metric. Here, there is overlap with the goals of naturalness and historical fidelity: defining a target for landscape diversity requires consideration of what pre-human, Native American, or early European settlement levels of landscape diversity were, and then a decision as to whether management should aim to recreate those conditions or novel ones. Similarly, in defining targets for numbers of populations for fire-associated species, managers may consider establishing a target in line with past conditions, or may define a target number based on a metapopulations model or a model looking at population viability in the face of climate change.

Adhering to a “hands-off” definition of naturalness would be logistically simple as it would not require any active management—just a passive commitment to “let things burn.” But, as described above, this would result in further loss of fire-dependent vegetation and landscape diversity and could thus run counter to biodiversity goals. In some places, managers may decide this is acceptable, in particular given the limited resources available for fine management in the Southern Appalachians. Under such a scheme there will still be at least some fire, and pine forests would remain on at least the edaphic core and perhaps the "fire core," under current climate and environmental conditions.

Managing for historical fidelity to pre-European or pre-industrial conditions would require extensive restoration and continued prescribed fire, as these conditions were maintained by widespread anthropogenic fire. This may just involve reintroducing fire to fire suppressed areas according to the hypothesized Native American fire regime of frequent, low severity fires with occasional high severity fires, but could also require some other restoration treatments such as fuel treatments or very high severity fires. There has been some success at regenerating pines and oaks with prescribed fire, but most studies suggest that higher severity prescribed fires are necessary to kill off encroaching hardwoods and shrubs to allow pines to establish (Elliott et al. 1999, Welch et al. 2000, Jenkins et al. 2011). To my knowledge, there have been no studies that have tracked long-term effects of prescribed fire on pine and oak regeneration or the effects of multiple prescribed fires; this is an area in need of further research to guide restoration and management activities.

Recreating historical conditions on the level of a few stands may be relatively straightforward, involving prescribed fire and other restoration treatments, but doing so on a landscape scale would be difficult for a number of reasons. First of all, the degree to which the current extent of pine in the Southern Appalachians reflects the historical extent is unknown. Second, given the existing financial and personnel capacity for prescribed fire in the Southern Appalachians it is unlikely that managers would be able to replicate the frequency and extent of Native American fire across the whole Southern Appalachian landscape, or even at the scale of a single national park or wilderness area. Therefore, managers must prioritize areas within the landscape in which to concentrate prescribed fire and other restoration efforts. This prioritization should not be done arbitrarily and should be based on other fire management goals, such as promoting biodiversity.

Here, again, the goals of historical fidelity and biodiversity complement each other and can be used in concert to help prioritize areas for management focus. Areas that currently maintain the largest populations of fire dependent species, contain the most diverse arrays of fire dependent tree species, or those with a relatively diverse herb-dominated understory could be targets for prescribed fire. Alternatively, management efforts could be focused on places that are prone to wildfire and would have likely supported fire dependent communities prior to the proliferation of anthropogenic fire across the landscape, in adherence to the "pristine" definition of natural. Further research to identify the environmental conditions most favorable to wildfire in the Southern Appalachians would be useful in applying this management goal. Additionally, managers should consider the concept of mesophication when making priorities. At wetter sites, mesophication may have progressed past the point where reversing its effects are extremely difficult, if not impossible (Nowacki and Abrams 2008). At other sites, prescribed fire may be urgently needed to slow or reverse the process of mesophication. More research to identify the threshold at which restoration will require more intensive treatments beyond prescribed fire due to mesophication will help managers identify sites at which to focus their prescribed fire efforts.

Anthropogenic climate change will likely alter fire regimes and vegetation in the Southern Appalachians. Climate change could affect fire risk in the Southern Appalachians, though we do not yet know how, and could affect the prevalence and frequency of southern pine beetle outbreaks and other non-fire disturbances or rates of succession. Warming climates and shifts in the location, timing, and amount of precipitation could also result in shifts in species distributions and no-analog communities (Williams and Jackson 2007). No-analog communities may also have no-analog fire regimes as fuel structure, moisture, and connectivity could be different from conditions that exist today. These changes may require consideration of goals that are not currently applicable to fire management in the Southern Appalachians, such as managing for ecosystem processes or to prevent catastrophe, and will complicate attempts to maintain historical fidelity or naturalness. With widespread climate change, maintaining viable populations of fire-associated species may be the most practical goal for fire managers: working to conserve particular community types or associations could be futile. Continued monitoring and research, flexibility, and adaptive management will be necessary to determine whether this is the case. In the meantime, using prescribed fire and promoting landscape heterogeneity may be the best way to hedge against climate change when outcomes are so uncertain.

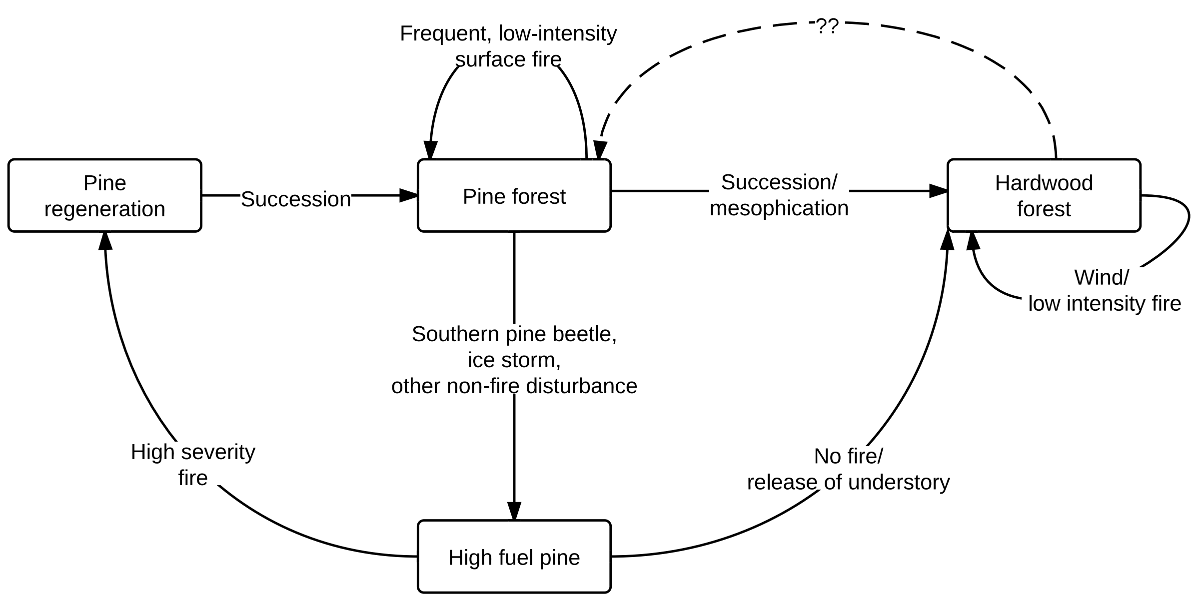
Maintaining pine-oak forests in any substantial portion of their pre-suppression range will certainly require substantial prescribed fire, as these forests were spread and maintained by Native American and early-settlement fire, and lightning fire does not occur frequently enough to maintain fire dependent vegetation on its own. Maintaining fire dependent vegetation in the Southern Appalachians is not a matter of restoring historic conditions and ending fire suppression so that the lightning fire regime can take over again; instead it will require a long-term commitment to continued prescribed fire. Because of this, it is important that management goals are carefully evaluated, that values are explicitly acknowledged, and that decisions about where, why, and how to burn are made scientifically.

Figures and Tables

Table 1: Fire management goals in the Southern Appalachians

Goal	Important in Southern Appalachians?	Remaining questions
1. Prevent catastrophic changes to ecosystems	No	How will climate change affect fire risk in the Southern Appalachians?
2. Ecosystem processes	Unknown	How does fire in the Southern Appalachians affect ecosystem properties/processes within pine forest and at the landscape scale?
3. Biodiversity	Yes	Without prescribed fire, how much would the range of yellow pine forest contract? Where is the edaphic core? Where is the fire core?
4. Naturalness	Somewhat	Where is the fire core? Which sites would have historically supported pine in the absence of anthropogenic fire? Did Native American fire interact with lightning fire and make it less frequent?
5. Historical fidelity	Yes	How much pine is a result of Native American burning vs. wildfire or other anthropogenic fire?
6. Prevent damage to human life and property	No	How will climate change affect fire risk in the Southern Appalachians? Will future land-use and expansion of the wildland-urban interface increase this risk?

Figure 1: The fire-pine hypothesis in the Southern Appalachians (modified from White 1987)



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CHAPTER 3:

EFFECTS OF PRESCRIBED FIRE ON FOREST STRUCTURE AND COMPOSITION IN THE GREAT SMOKY MOUNTAINS NATIONAL PARK

Introduction

Fire suppression in the eastern United States can drive a process called "mesophication," a positive feedback in which fire-prone, open-canopy forests undergo succession towards dense, closed-canopy forests with moist microclimatic conditions that are unfavorable for fire ignition and spread (Nowacki and Abrams 2008). This phenomenon occurs in the fire-dependent pine-oak forests of the Southern Appalachians, which were once dominated by fire dependent yellow pine (*Pinus* subgenus *pinus*) and oak (*Quercus*) species. In the absence of frequent fire, fire-sensitive species such as *Acer rubrum, Nyssa sylvatica,* and *Pinus strobus* can grow large enough to resist fire (Harmon 1984). These trees fill in the canopy and create shady, moist conditions that are conducive to their own reproduction and unfavorable for reproduction of fire-dependent species such as *Pinus rigida* and *Pinus pungens,* which require high light conditions and exposed mineral soil to reproduce (Zobel 1969, Waldrop and Brose 1999). Fire dependent species in these forests are failing to regenerate, resulting in declines in the abundance of yellow pines and oaks and loss of a unique community type across the Southern Appalachians.

Mesophication occurs more quickly at wetter sites where fire-sensitive mesophytic species can establish and grow more quickly than on more xeric sites. This can transform formerly fire-dependent communities to an alternate stable state in which restoration of previous forest structure and composition may be exceedingly difficult (Nowacki and Abrams 2008). While prescribed fire may help slow or reverse the process of mesophication, doing so becomes increasingly difficult as mesophication progresses (Abrams 2005). Understanding how the effects of prescribed fire vary depending on local environmental conditions or the length of time since the start of fire suppression can help managers prioritize sites for prescribed fire, by both identifying conditions under which prescribed fire is most effective at restoring and maintaining fire-dependent communities and places where fire is most urgently needed to prevent a shift to an alternate stable state.

In the Southern Appalachians, fire-dependent forest communities dominated by yellow pines and oaks are found on xeric south and southwest facing slopes and ridgetops at mid elevations (Whittaker 1956, Jenkins 2007). These communities have undergone changes to their structure and composition due to fire suppression and changes in human fire-use patterns (Harmon 1982, 1982, Harrod et al. 1998). Prior to suppression, frequent, low-intensity surface fires occurred at approximately 3 to 13 year intervals (Harmon 1982, White 1987, Aldrich et al. 2010, Feathers 2010). These fires maintained open, park-like stands with a rich herb layer and shallow litter and duff layers dominated by fire resistant pine species such as *Pinus rigida,* and *Pinus pungens*, as well as oak species such as *Quercus coccinea* (Harrod et al. 1998, Lafon et al. 2007). Occasional severe fires often followed a southern pine beetle (*Dendroctonus frontalis*) outbreak, ice storm, or other disturbance (White 1987, Lafon and Kutac 2003). These high severity fires opened up the canopy, increased light availability, reduced the litter and duff layers, and promoted episodes of pine recruitment (Harrod et al. 2000, Wimberly and Reilly 2007, Jenkins et al. 2011). Prior to suppression, most fire in the Southern Appalachians was anthropogenic. Native Americans used low-intensity surface fire to maintain hunting and gathering grounds (Delcourt and Delcourt 1997, 1998, Brose et al. 2001) and early European settlers in the pre-industrial Southern Appalachians used fire to improve common grazing lands and conditions for hunting and gathering, or for pest and disease prevention (Jurgelski 2008).

Fire suppression since the early 20th century has caused these stands to undergo succession towards dense closed-canopy forests dominated by fire sensitive species such as *Acer rubrum* and lacking conditions conducive to regeneration of fire resistant species (Harrod and White 1999). Deep litter and duff layers have accumulated, and conditions have become shady and moist so that many fires are not hot enough to reduce the litter and duff, kill back fire sensitive species, and promote pine or oak regeneration (Vose et al. 1995, Harrod et al. 1998). As a result, management agencies such as the National Park Service and US Forest Service have begun to use prescribed fire to restore pre-suppression conditions.

These efforts have had mixed success. Elliott et al. (1999) found that while *Pinus rigida* seedlings were abundant after a prescribed fire in Nantahala National Forest, North Carolina, most died within two years after fire and *Pinus rigida* seedling density was less two years after prescribed fire than before. This may have been due to competition for light from shrubs like *Kalmia latifolia*, which resprouted vigorously after fire. Welch et al. (2000) found *P. rigida* and *P. pungens* seedlings present after low intensity prescribed fire. However, shrubs and hardwoods were not exposed to lethal temperatures, and resprouted to such an extent that understory density within a year of prescribed fire was double pre-fire density. Jenkins et al. (2011) found that below a certain threshold of fire severity, yellow pine seedlings were not observed following prescribed fire in the Great Smoky Mountains National Park (GSMNP). These studies demonstrate that to successfully promote pine regeneration, prescribed fire must open the canopy, reduce litter and duff, and expose competitors to lethal temperatures. They also suggest that conditions conducive for pine regeneration created by low-intensity prescribed fire may be too short-lived to promote pine regeneration. As data for longer than two years after a prescribed fire is often unavailable, it is difficult to evaluate the effectiveness of prescribed fire at achieving management goals in the long-term.

While there is some agreement about *how* to do prescribed fire in order to achieve stated management goals, less attention has been paid to *where* to focus fire management efforts. Prescribed fire or a particular prescribed fire regime may be more or less effective depending on the environment, climate, or history of a given location. Mesophication is hypothesized to proceed more quickly at more mesic sites (Abrams 2005, Nowacki and Abrams 2008); prescribed fire there, even if conducted under very dry conditions, may not be effective at killing off fire-sensitive species if they have grown large enough to resist fire. On the other hand, xeric, steep, and rocky sites with shallow soils may not require prescribed fire to maintain populations of *Pinus rigida* or *Pinus pungens* (Williams 1998, Barden 2000) and prescribed fires in these areas could have minimal effects and be a poor use of resources. Many studies of prescribed fire in the Southern Appalachians examine the effects of only one fire and thus do not have a gradient of environmental, topographical, and fire conditions across which to compare. In order to better understand where to focus prescribed fire efforts, it is necessary to examine the effects of multiple fires across an environmental gradient.

In this study, I use long-term fire effects monitoring data from GSMNP spanning 16 years and 21 fires to address the following questions:

1) How do community composition and structure change in fire suppressed stands after a prescribed fire?
2) Are these changes maintained over time?
3) Do the magnitude of change and the degree to which changes are maintained over time vary with site environment conditions and/or fire severity?

Methods

Study area

This study focuses on the effects of prescribed fire in xeric ridge forests in the Great Smoky Mountains National Park (GSMNP). The park is located in the southern Appalachian Mountains, along the border between North Carolina and Tennessee. Elevation ranges from 267 m to 2025 m. Mean annual precipitation averages 143 cm and mean annual temperature in nearby Gatlinburg, TN is 13°C (Jenkins et al. 2011). Precipitation is highest at high elevations in the park (Shanks 1954).

Xeric ridge forests cover about 16% of total park area (33,000 ha), largely on ridgetops and exposed south- to west-facing slopes, and are dominated by *Pinus* and *Quercus* species (Jenkins 2007). The yellow pine species *P. rigida* (pitch pine), *P. pungens* (Table Mountain pine), *Pinus echinata* (shortleaf pine), and *Pinus virginiana* (Virginia pine) dominate the most xeric of these forests, while less xeric sites are dominated by *Pinus strobus* (white pine), *Quercus coccinea* (scarlet oak), and *Quercus montana* (chestnut oak). These forests would have historically experienced regular fire, with a fire return interval of approximately 13 years prior to suppression (Harmon 1982). Several of the dominant pine and oak species display adaptations to fire, such as thick bark and the ability to resprout after fire. Table Mountain pine is partially serotinous (Zobel 1969) and some *P. rigida* cones remain closed until exposed to heat (Jenkins et al. 2011). These dominant species also reproduce more successfully in the presence of fire. The yellow pine species in GSMNP require high light conditions and shallow litter and duff in order to germinate and the oaks are shade intolerant and germinate well on recently burned substrate (Zobel 1969, Williams and Johnson 1992, Brose et al. 2001, Brose and Waldrop 2006). Other common tree species are *A. rubrum* (red maple), *N. sylvatica* (black gum), *Tsuga canadensis* (eastern hemlock) and *Oxydendrum arboretum* (sourwood), all of which are relatively shade tolerant and fire sensitive. There is a well-developed shrub layer, with common species including *Kalmia latifolia* (mountain laurel), *Vaccinium* species (blueberry), and *Gaylussacia* species (huckleberry). Prior to fire suppression, these communities may have had a rich herbaceous layer, but this is largely absent today (Harrod et al. 2000). The elevation in the study area ranges from 330 m to 1030 m.

As elsewhere in the Southern Appalachians, fire suppression since the founding of the park in 1931 has resulted in changes to forest structure and composition and decreased regeneration of fire-dependent pine and oak species (Harmon 1994, Harrod et al. 1998, Harrod et al. 2000). To remedy this, GSMNP management has been conducting prescribed burns since 1996 with the main objectives of reducing stand densities and regenerating fire dependent species, especially yellow pines (Jenkins et al. 2011). Since the start of the program, there have been 21 prescribed fires conducted in the GSMNP ranging in size from 18 to 939 ha (Figure 2). Fires have been conducted throughout the park, with the majority in the western portion of the park. In order to monitor vegetation communities' responses to prescribed burning and to provide feedback for adaptive management, the National Park Service (NPS) established the fire effects monitoring program in 1997 (Jenkins et al. 2011). The program uses a standardized set of protocols to monitor conditions during fires and to assess fuel and vegetation conditions prior to burning and at set intervals following fire (USDI National Park Service 2003).

Data collection

Vegetation and fuels data were collected according to the protocols in the National Park Service's fire monitoring handbook (USDI National Park Service 2003). Since 1997, permanent 0.1 ha plots have been established within each prescribed fire in the GSMNP and have been regularly sampled since. Plots are sampled within one year prior to the first burn, and again one, two, five, and ten years after fire. Sampling events are denoted throughout this paper as "pre", "fire1year1", "fire1year2", etc. The monitoring cycle begins again after every burn. A total of 39 of these plots were included in this study: 19 that have burned once, and 20 that have burned twice. Eleven plots total were sampled up to fire2year5, the longest time interval included in this study. Due to differences in the timing of prescribed burns as well as logistical considerations that prevented sampling from occurring on several events, not all plots have been sampled on all sampling events. For example, a plot that was burned for the second time only three years after the first fire will be lacking data for the fire1year5 sampling event, while a plot that was burned for the second time six years after the first burn will have fire1year5 data. Sample events in some plots were occasionally skipped due to safety considerations.

During each sampling event, trees larger than 15 cm dbh were measured and identified to species. Trees larger than 2.5 cm dbh and smaller than 15 cm dbh were measured and identified in a 250 m^2 subplot. Measurements from subplots were multiplied by four to match the full plot scale. Herb, shrub, subshrub, and vine cover were measured along two 50 m transects in each plot. Along each transect, a pole was dropped every 30 cm and the height and species of every individual intersecting the pole were recorded. Cover was calculated by totaling the number of points at which a species was recorded by the total number of points sampled. Cover by lifeform (herb, shrub, subshrub, vine) was calculated by summing percent cover of all herb and shrub species in a plot.

Dead and down woody fuels, litter, and duff were sampled using methods described by Brown (1974). Four randomly oriented 15.2 m transects were established in each plot. Along each transect, fine woody debris were tallied by fuel/size class as follows: 1-hour fuels (0-0.62 cm diameter) and 10-hour fuels (0.62-2.54 cm diameter) were tallied along the first 1.83 m of each transect, 100-hour fuels (2.54-7.62 cm) were tallied along the first 3.66 m, and 1000-hour fuels (>7.62 cm) along the whole transect. Depth of litter and duff were measured at 10 points along each transect. Fuel loadings were calculated based on Brown (1974) and Brown et al. (1982) with values for specific gravity of downed woody material and bulk density of litter and duff from Jenkins et al. (2011). Total fuel reduction after fire and reduction in the average depth of litter and duff were included in analyses as measures of fire severity. Depth of litter and duff was also included in analyses as it is an important factor in determining success of pine and oak regeneration.

Measures of topographic position (elevation, slope, aspect, topographic convergence index) were used as environment variables. A digital elevation model (DEM) for GSMNP was derived from the United States Geological Service nationwide 10-m DEM (Gesch et al. 2002, Gesch 2007). Slope and aspect were derived from the DEM in ArcGIS (ESRI 2011). Aspect is used as a proxy for evaporative demand. Southwest facing slopes experience the highest evaporative demand, while northeast facing slopes experience the lowest. To reflect this, aspect was transformed using the equation $TA = -(\cos 45 - A)$, so that a maximum value of 1 represents southwest facing slopes and a minimum value of -1 represents a northeast facing slope (Pierce et al. 2005). Topographic convergence index (TCI), the ratio of uphill contributing area to slope, was used as a proxy for soil moisture (Beven and Kirkby 1979). TCI was calculated from the USGS DEM using the terraflow package in GRASS (Arge et al. 2003).

Analysis

Management objectives in GSMNP are largely focused on restoring pine and oak dominance. Therefore, I focused on examining and explaining changes in density and species composition of trees. Prior to analyses, all trees in genus *Carya* were grouped together because the majority were missing species identifications, and species found in less than five percent of samples were removed from the dataset.

When analyzing multivariate community data, it is routine to use data transformed in various ways, including relativized by site or by species, square root transformed, or some combination of these. Here I used stem density data and conducted analyses with both raw data and data relativized by site totals. Analysis of raw stem density data captures changes in the number of stems in a plot in addition to changes in species composition, whereas analyses of data for relative density will reflect only changes in composition. Because many plots underwent major changes in tree density after fire, analyses of raw density data predominantly reflected changes in structure and not necessarily in composition. Therefore, I repeated the analyses with both sets of data to assess trends in composition as well as structure after fire.

Ordination is a set of methods used to analyze multivariate data in low dimensional space. To assess changes in community structure and composition, I used non-metric multidimensional scaling ordination (NMS, Kruskal 1964) as implemented in the *vegan* package in R (Oksanen et al. 2007, R Development Core Team 2010). NMS makes no assumptions about the shape of species occurrence distributions or about the relationship between species occurrences and environmental gradients. Individual observations are arranged in a low-dimensional space so that the distances between points in ordination space correspond to observed dissimilarities between samples. Environmental variables are overlaid as biplot vectors after the fact, where the direction of a vector represents the direction in which that variable increases and the length represents the strength of the correlation between the environmental gradient and the ordination axes.

Pairwise dissimilarities between plots were calculated using the Bray-Curtis (Sorenson's) dissimilarity index. All plots that were sampled before and at least once following fire were included in the ordination and all sampling events for these plots were included as separate entries in the data. A step-down procedure was used to determine the proper dimensionality of each NMS solution. Varimax rotation was used to rotate each solution so that NMS axis 1 explained the most variance. Individual environment variables (Table 1) were overlaid on the ordination. Only correlation vectors that were significant at the $P<0.05$ level were plotted. Species scores were calculated as average ordination scores weighted by species abundances. Correlations between the ordination axes and environment variables and species abundances were calculated using the ecodist package in R (Goslee and Urban 2007). To analyze plots' changes over time, change vectors connecting a plot at one time to itself at a later time were visually compared.

There is a longitudinal gradient present in the dataset, with plots spanning the width of the park (Figure 2). Because the mountains to the east are higher than those to the west, elevation is correlated with longitude ($r=0.87$, $P<0.0001$). I expected that the effect of longitude would be largely a reflection of the effect of elevation. To test whether there was an effect of longitude beyond the effect of elevation, I used linear regression to predict ordination scores on NMS axis 1 (the axis with which elevation and longitude were most strongly correlated) from elevation, and then from elevation and longitude. An F-test of the two models demonstrated that longitude does not contribute significantly beyond elevation ($P=0.08$). Therefore, I excluded longitude from analyses.

To analyze how the magnitude of change varies with environment and with fire severity, and whether these changes are maintained over time, I calculated Spearman's rank correlation between the environment variables of interest (Table 2) and plot level fire effects, represented as each plot's dissimilarity to itself before fire and 1, 2, and 5 years after the first fire, and 1 and 5 years after the second fire. A plot's dissimilarity to itself is proportional to the length of the change vectors plotted in ordination space, but is preferable for use in statistical analyses because the ordination does not perfectly represent the dissimilarities between plots in two-dimensional space.

Because not all plots were sampled at all sampling events, it was not possible to examine trends across the entire dataset over time in terms of species abundances or size class distributions. Instead, to qualitatively assess dynamics on the plot level, I examined individual plots that were burned twice and were sampled five years after the second fire and at most time points along the way. For each of these plots I calculated size class distributions in 10 cm dbh bins, grouping trees with dbh > 70 cm together, and examined changes in the size class distribution over time for the entire plot and for selected species within the plot. Additionally, I plotted density over time for the dominant species in each plot. Plots of density over time illustrate trends in terms of both absolute density as well as relative abundance of different species. Downward sloping lines indicate declines in species' density, and if two species' lines cross it indicates a shift in relative dominance.

Results

Ordination of raw density data

The step-down procedure suggested a two-dimensional solution, and the final stress in two dimensions was 0.22. The ordination axes had a cumulative r^2 of 0.71 with NMS axis 1 explaining most of the variation (Table 2). Burned and unburned plots overlap in ordination space, but the unburned plots appear to be largely confined to the right side (Figure 3a). To test if the variation within groups of plots that were all burned or unburned was less than the variation between groups, I conducted a mantel test on the distance matrix between points in ordination space and a pairwise design matrix

(Legendre and Legendre 1998) that contrasts pairs of samples that are in the same group of burned vs. unburned with those that are in different groups. The results of this test were not significant, indicating that between group dissimilarities are not significantly different from the within group dissimilarities despite the visual pattern in ordination space (Mantel r=0.02, P=0.25), possibly due to variation in sample size between the two groups.

A plot of species scores indicated that all species had their maximum expected abundances on the right side of the graph, corresponding with the area occupied by the unburned plots (Figure 3b). This suggests that all species are most abundant in unburned plots and decline in abundance after fire although the fact that burned plots span the range of NMS axis 1 affects the certainty of this interpretation. The species scores for the yellow pines (*Pinus rigida, Pinus echinata,* and *Pinus virginiana*) are the furthest into the side of the graph where burned plots dominate, suggesting that these species' densities change the least after fire.

NMS axis 1 was most strongly correlated with elevation, and most negatively correlated with herbaceous cover and total fuel reduction, while NMS axis 2 was correlated with the fire severity measures (fuel reduction), variables that are affected by fire (shrub and herb cover, total tree density), and with aspect (Table 3, Figure 3c, 3d).

Change vectors drawn from samples taken before fire to one year after fire showed a trend towards lower density, lower shrub cover, and higher herb cover (Figure 4a). Plot dissimilarity with itself between these time periods varies from 0.03 to 0.86, with a mean of 0.33. In most plots, the majority of change takes place within the first year after fire: subsequent change vectors, drawn from samples one year after fire to two years after fire, and from two years after fire to five years after fire, are shorter on average, and do not show any major trend in any direction (Figure 4b, 4c). Change vectors drawn from before fire to five years after the second fire show an overall trend towards lower density, lower shrub cover, and higher herb cover (Figure 4d), suggesting that changes in density after the first fire are maintained over time if there is a second fire. There is substantial variation in the magnitude of change in the time period from before fire until fire2year5, with dissimilarity values ranging from 0.15 to 0.85, with a mean of 0.5.

Results indicate that for the most part, changes after fire are maintained after the first year. If changes to plot structure and composition caused by fire began to revert to prior conditions in the years following fire, I would expect change vectors for years two and five to trend in the opposite direction than the change vectors immediately following fire. Instead, change vectors plotted from fire1year1 to fire1year2, and fire1year2 to fire1year5 (Figure 4b, c) are short on average and no single direction predominates, indicating that changes in later years are much smaller in magnitude than those immediately following fire, and are not necessarily in the opposite direction of the immediate changes.

Ordination of relative density data

The step-down procedure conducted with the relativized density data again suggested a two-dimensional solution. The final stress was 0.22, and the combined r^2 of the axes was 0.71 (Table 4). There did not appear to be any pattern in the distribution of burned and unburned plots in ordination space as there was in the previous ordination (Figure 5a). A mantel test on the distance matrix between points in ordination space and a pairwise design matrix on whether or not two plots had both been burned indicated that between group dissimilarities were not significantly different from within group dissimilarities (mantel r= -0.06, P=0.97).

Species scores were distributed more evenly throughout ordination space than in the previous ordination (Figure 5b). NMS axis 1 was strongly correlated with elevation and with depth of litter and duff, and negatively correlated with herbaceous cover and total fuel reduction (Table 3; Figure 5c). NMS axis 2 was negatively correlated with TCI, herbaceous cover, and most positively correlated with depth of litter and duff and with shrub cover. Herbaceous and shrub cover are again negatively correlated with each other, and herbaceous cover and fire severity (total fuel reduction) are positively correlated and increasing towards the left. The yellow pines are located toward the left side of the graph, suggesting that they have higher relative abundance at lower elevations and at sites that have high herbaceous cover and have experienced hotter fires. The more mesophytic species such as *A. rubrum, T. canadensis,* and *P. strobus* have their maxima towards the top right side of the graph, at higher elevations and in areas with higher shrub cover, deeper litter and duff, and less severe fires. TCI is negatively correlated with NMS axis 2 and is approximately orthogonal with elevation, suggesting an effect of site moisture on composition beyond that of the elevation-moisture gradient. Change vectors plotted over all time periods did not show any clear temporal trends with regards to composition in either the short- or long-term. Figure 5d shows change vectors plotted from before fire to fire2year5; the lack of directional trend pictured there is representative of the trends observed over all time intervals examined.

Correlation analyses

Analysis of the correlation between environmental variables and fire severity with plots' fire effects (dissimilarity to itself) calculated from raw density data found a significant correlation with elevation over all periods examined. Variables associated with fire severity were significantly positively correlated with a plot's fire effects in the shorter term, and less important in the long-term, whereas variables associated with topography (slope and aspect) were not correlated with fire effects in the short term, but were correlated with the longer-term magnitude of change (Table 5). Inference here is limited as a different subset of plots were included in correlation analyses from year to year due to differences in sampling between the plots, and thus a different range of each of the variables tested was represented in each round of analysis. For example, the range of elevation in the analyses for fire1year1 is 331 m to 1047 m, while the range for fire1year5 is only 331 m to 635 m, explaining the switch in the sign of the relationship for pre to fire1year5 dissimilarity.

Correlation analyses conducted with fire effects values calculated from relativized density data showed different patterns. Elevation was negatively correlated with the magnitude of change after fire for some time intervals, but not others. Aspect and TCI were never significant, and slope was significant only in the interval from pre-fire to fire1year5. Variables relating to fire severity were significantly correlated with the magnitude of compositional change from pre-fire to two years after the first fire, and depth of litter and duff was important for the first two years after a prescribed fire.

Individual plot dynamics

I selected four plots to illustrate a range of trajectories that plots may experience after prescribed fire (Table 7). Plot 1 is the highest elevation plot, and is located on a southwest-facing slope in the eastern part of the park. Fires in this plot were of relatively low severity, duff was shallow, and dissimilarity from pre fire to fire2year5 in this plot was the lowest observed. Figure 6a shows stem density over time for six dominant species in this plot. The lines are nearly parallel, indicating little change in relative dominance. The lines slope downwards, but only gently so, indicating minor changes in total plot density. Fire killed some small trees, but the overall size class distribution remained largely unchanged (Figure 6b). Similarly, size class distributions for individual species in the plot were relatively static, although the number of small individuals of several species including *A. rubrum* and *P. strobus* declined (Figure 5c).

Plot 2 is at mid-elevation, located in the western part of the park, and is at a northeast aspect and a mid-slope. The first fire in this plot was very low severity (total fuel reduction=0.97 tons/ha), but the second was high severity (26.7 tons/ha). The plot's dissimilarity to itself over the entire time period was the median value for all plots. The varying severity of the fires is reflected in the plot of stems/ha over time: after the first fire species abundances remained stable but most species abundances declined after the second fire (Figure 7a). The species that were the most stable in density after the severe fire are *Q. coccinea* and *P. rigida*, the two most fire tolerant and fire dependent of the species found in this plot. The second fire in this plot killed a large number of small trees, which altered the shape of the size-class distribution (Figure 6b). There were a large number of *N. sylvatica*, and several *P. strobus* and *A. rubrum* in the smallest size class that persisted after the first fire, but these were almost all killed by the second fire (Figure

6c). By the end of the sample period, the number of *A. rubrum* remaining in the plot was diminished. However, those that remained were in larger size classes and may be large enough to escape future fires.

Plot 3 is located near plot 2. It is also at mid-elevation, on a relatively gentle slope (6.6 degrees) and has the highest TCI and highest species richness of the four example plots. The first and second fires did not differ in severity, which is reflected in the plot of stems/ha over time. Densities of all species declined over time, and most experienced a slight decline after the first fire and another slight decline after the second (Figure 8a). The number of individuals in the smallest size classes declined over time, but gradually, while the number of individuals in the other size classes remain relatively constant (Figure 8b), unlike the quick and drastic change observed in the size class distributions of plot 2 after the second fire. *A. rubrum* and *O. arboretum*, both relatively fire sensitive species, declined continuously after the second fire. This decline appears to be mostly associated with individuals in the smallest size class (Figure 8c). Prior to fire, there were a large number of *A. rubrum* and *O. arboretum* individuals in the smallest size classes, which were nearly all killed over the sample period. Individuals of these species in size class 2 and above remained relatively constant, suggesting that they had grown large enough to escape the moderate severity fires this plot experienced.

Plot 4 is the lowest elevation of the four example plots and has the lowest TCI. The fires it experienced were quite severe, reducing litter and duff to only 1.9 cm after the second fire, and dissimilarity from pre to fire2year5 was high. All species declined sharply after fire (Figure 9a). *N. sylvatica* and *A. rubrum* were eliminated from the plot, and *Q. coccinea* declined to zero density after the second fire but regenerated or resprouted by the end of the sampling period. Relative dominance changed with fire in this plot as well: *P. rigida* was dominant by the end of the sampling period and experienced the least decline with fire. The number of individuals in all size classes declined over time, but this decline was most severe in the two smallest size classes (Figure 9b). Before fire, individuals of *N. sylvatica* and *A. rubrum* were all in the smallest two size classes, and these were all killed by fire. *P. rigida,* whose density remained the most constant over time, had many individuals in larger size classes before fire (Figure 9c). By the fifth year following the first fire, *A. rubrum* started to regenerate or resprout, indicated by an increase in the number of individuals in the smallest size class. The second fire, however, killed these individuals, and they had not returned by the fifth year after the second fire.

Discussion

Reducing stand density is important for creating high light conditions that promote pine and oak regeneration (Elliott et al. 1999, Welch et al. 2000, Jenkins et al. 2011) and is a desired outcome of prescribed fire in the Great Smoky Mountains along with reducing depth of litter and duff. Results from the ordination of raw density data indicate that prescribed fire does consistently reduce stand density, and that these reductions are maintained over the duration of the study particularly if there is a second fire. Plots also move towards higher herbaceous cover and lower shrub cover, indicating that in addition to reducing canopy shading, prescribed fire can reduce shrub layer competition with pine and oak seedlings while promoting the colonization of herbaceous species through competitive release. However, without direct analyses of seedling data, it remains unclear whether these changes will be adequate to result in pine and oak regeneration.

Other studies have found that prescribed fire can cause understory trees and shrubs to vigorously resprout, potentially making prescribed fire counterproductive if the long-term goal is to reduce density in the overstory, understory, and shrub layer (Elliott et al. 1999, Welch et al. 2000). Change vectors plotted over the maximum study duration indicate an overall shift towards higher herbaceous cover and lower shrub cover after two fires (Figure 4d). It is possible that although shrub cover declines, trees killed by fire do resprout, but are not large enough to be sampled (i.e. have dbh < 2.5 cm) within the sample interval. Further monitoring and analyses of existing seedlings data will be necessary to determine if this is the case.

Fire severity is important in determining post-fire effects in the Southern Appalachians (Groeschl et al. 1992, Harrod et al. 1998, Elliott et al. 1999, Wimberly and Reilly 2007, Jenkins et al. 2011). I found that particularly during the time immediately following fire, fire severity (measured as reduction of fuels, litter, and duff) is positively correlated with the magnitude of compositional and structural change after fire (Table 5, Table 6). The dynamics in plot 2 highlight the importance of fire severity. The first fire in this plot is very low severity and reduces litter and duff by only 0.1 cm and total fuels by only 0.97 tons/ha, close to the minimum values observed in the data. Accordingly, composition and structure remain stable in this plot after the first fire. The second fire is far more severe (litter and duff reduction=7.1 cm, total fuel reduction=26.7 tons/ha) and with it, we observe a shift back towards pine dominance and declines in fire sensitive species (Figure 7).

The differences between the ordinations of raw density data versus relativized density data demonstrate that while there is a trend in terms of density (highlighted by the raw density ordinations), there is not a clear-cut pattern with regards to composition. Some stands do appear to be shifting towards increased pine and oak dominance (such as plot 4), while others change very little or change in unexpected directions. There are several possible explanations for this. First, fire-related mortality varies between plots, and the magnitude of compositional change (i.e. the length of the change vector in the relative density ordination) may be just a reflection of the amount of mortality. If this were the case, I would expect the lengths of the change vectors in both ordinations to be correlated. Another possible explanation is the different rate at which fire suppression affects plots or the length of time since fire before prescribed fire. In plots where succession away from pine-oak dominance proceeds slowly or where fire occurred relatively recently before suppression, fire-sensitive hardwood species are confined to smaller size classes. In plots where succession has proceeded more quickly or in plots where the time since fire prior to prescribed fire is longer, individuals of these species have grown large enough to escape fire and be present in the canopy and there may be few pines and oaks remaining in the canopy. In these plots, fire may kill small trees and reduce stand density without restoring pine or oak dominance or driving a significant shift in the relative abundance of species. If this were the case, I would expect no correlation between the lengths of the change vectors in both ordinations. To test this, I calculated Spearman's correlations between a plot's dissimilarity to itself from before fire to one year following the first fire calculated from raw density data and the same calculated from relative density data. I found that the two were correlated (r=0.85, *P*<0.0001), suggesting that variation in the magnitude of compositional change is due to variation in mortality and in the magnitude of changes in density. Further research should investigate drivers of variation in the direction of compositional change after fire.

Changes that take place after prescribed fire are maintained over the duration of the study. Change vectors do not trend back towards pre-fire conditions in later years following fire (Figure 4b, c). In the four example plots, most of the declines in density occur in the first year following fire, but species' densities either remain low or continue to decline in later years. This is not to say that regeneration is not occurring; rather this pattern is likely because few seedlings or resprouts of any species get large enough to be detected in sampling within the five year monitoring period. Continued monitoring would probably show abundant regeneration once enough time since fire has elapsed to allow new individuals or resprouts to grow large enough to be included in sampling. Species composition would depend on post-fire stand characteristics such as stand density, shrub cover, and depth of litter and duff.

My results suggest that mesophication is occurring in GSMNP, and that it is occurring more rapidly in wetter sites. In both ordinations, elevation was the variable most strongly correlated with NMS axis 1, meaning that elevation is the most important variable in explaining variation in site composition and density. Elevation is strongly correlated with precipitation in GSMNP (Shanks 1954), and the highest elevation plots in this dataset are located in the eastern part of the park. Plots at higher elevations change less after fire (Table 5, Table 6), have deeper litter and duff, higher shrub cover, higher stem density, lower herbaceous cover, and experience the least severe fires (Table 3, Table 4). Nowacki and Abrams (2008) demonstrate that mesophication occurs more rapidly and is more steadfast on mesic than xeric sites, as these sites tend to be more fertile and have more favorable growing conditions. In the GSMNP, this would mean that sites that once supported fire dependent vegetation at higher elevations would grow denser and accumulate litter and duff more rapidly than the drier, lower elevation sites in the western part of the park, and so when prescribed burns occur they have less of an effect. This appears to be the case in these results, but further research comparing pre-suppression stand structure to current stand structure is needed to confirm this.

The dynamics in plot 1 further illustrate the above point. This plot is one of the high elevation plots described above, and its density and composition remain largely unchanged even after two prescribed fires (Figure 6a). In this plot, many individuals of *A. rubrum* and *O. arboretum* are in size classes 2 and 3 (dbh >10 cm; Figure 6c). While the number of individuals of these species in the smallest size classes declines over time, the number in larger size classes remains fairly constant. The other three plots, which experience sharper declines in fire-sensitive species, do not have larger individuals of these species, suggesting that hardwoods and fire sensitive species have been invading plot 1 more quickly, or that plot 1 had a longer time since fire before suppression. Encroachment of hardwoods and other fire sensitive species will be easier to reverse on sites while individuals of these species are still small and susceptible to fire, but prescribed fire may be more urgently needed on the more mesic high elevation sites in order to prevent a shift to an alternate stable state.

Climate is another factor that may predict the effects of prescribed fire. Climate affects the frequency of fire in GSMNP, with fires occurring more frequently in dry years (Flatley et al. 2011). Fire severity is indirectly related to climate and weather: it is correlated with measures of drought such as the Keetch-Byram Drought Index (KBDI; Jenkins et al. 2011) and is related to site moisture (Wimberly and Reilly 2007). In this study, I did not consider climate and weather directly. I consider the effects of variation in precipitation by including elevation in my analyses, and TCI, aspect, and slope are all indirectly related to site moisture. Direct consideration of weather conditions preceding prescribed fires and of spatial and temporal variation in climate could further help to identify times and locations for which prescribed fire in GSMNP and other Southern Appalachian pine-oak forests will be most effective, and could help in predicting how these forests and fire regimes might change with future climate change.

Because lightning fire is rare in GSMNP (Harmon 1982, Cohen et al. 2007) and historic fire regimes were dominated by anthropogenic fire, maintaining pine-oak forests will require an enduring commitment to prescribed fire. This study includes some plots that have experienced two prescribed fires, but does not explicitly consider characteristics of the fire regime other than fire severity, such as fire frequency, fire season, or fire size. These variables can be important in driving the response to prescribed fire in terms of fuel consumption, tree dynamics, stand diversity and other post-fire effects (Glitzenstein et al. 1995, Sparks et al. 1998, Peterson and Reich 2001, Andersen et al. 2005, Knapp et al. 2005). Future research in the Southern Appalachians should investigate the role of all aspects of a fire regime, including frequency, burn season, and fire severity, in determining effects of prescribed fire. Understanding how to maintain and restore pine- and oak- dominated xeric ridge forests in the Southern Appalachians is a long-term project and managers should likewise pursue flexible, adaptive, long-term management planning accompanied by continued monitoring and research.

Figures and Tables

Figure 2: Prescribed fires in the Great Smoky Mountains National Park. Prescribed burns have been conducted throughout the park, but efforts have been focused on the western portion of the park (inset).

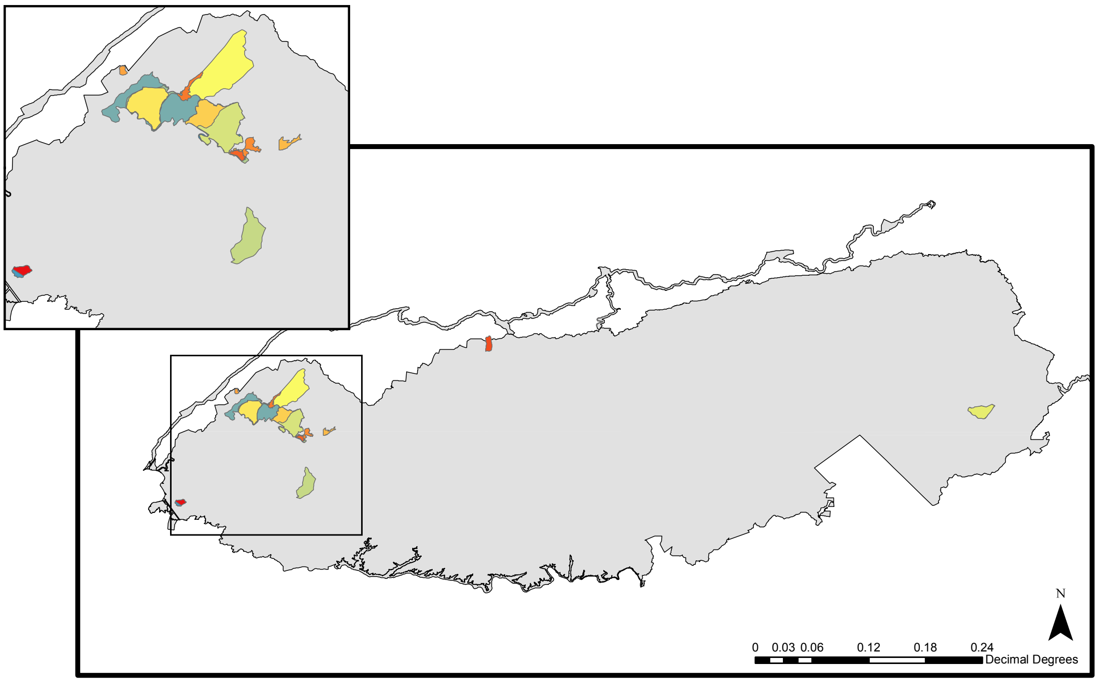

Figure 3: Results from NMS ordination of abundance data. a) samples. b) species scores. c) environment vectors. Only significant vectors (p>0.05) are displayed. d) vectors for burn characteristics. d.all=total fuel reduction, d.totdepth=litter and duff reduction, TotDep=depth of litter and duff at time of sampling

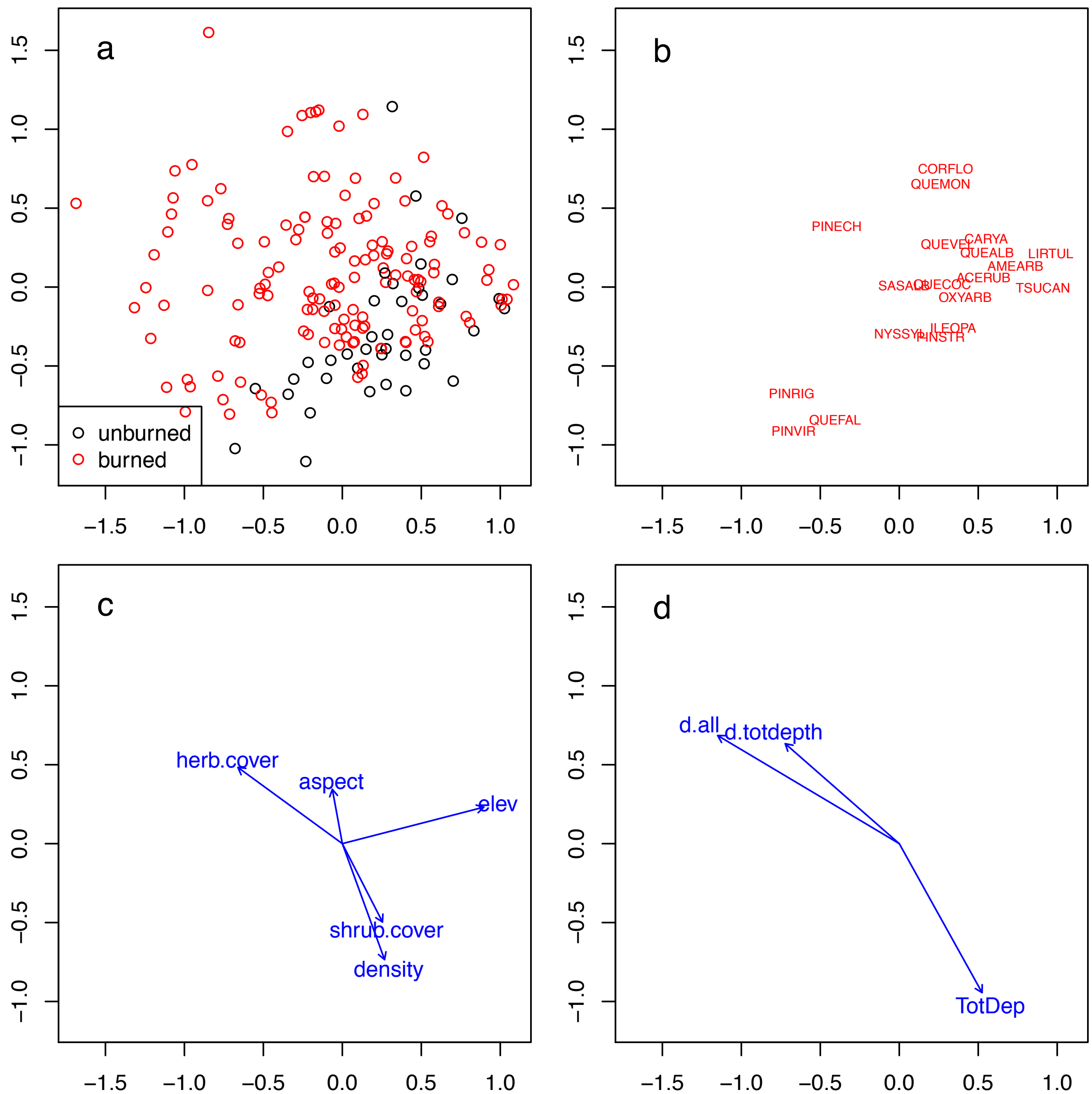

Figure 4: Change vectors. Red arrows represent the average vector.

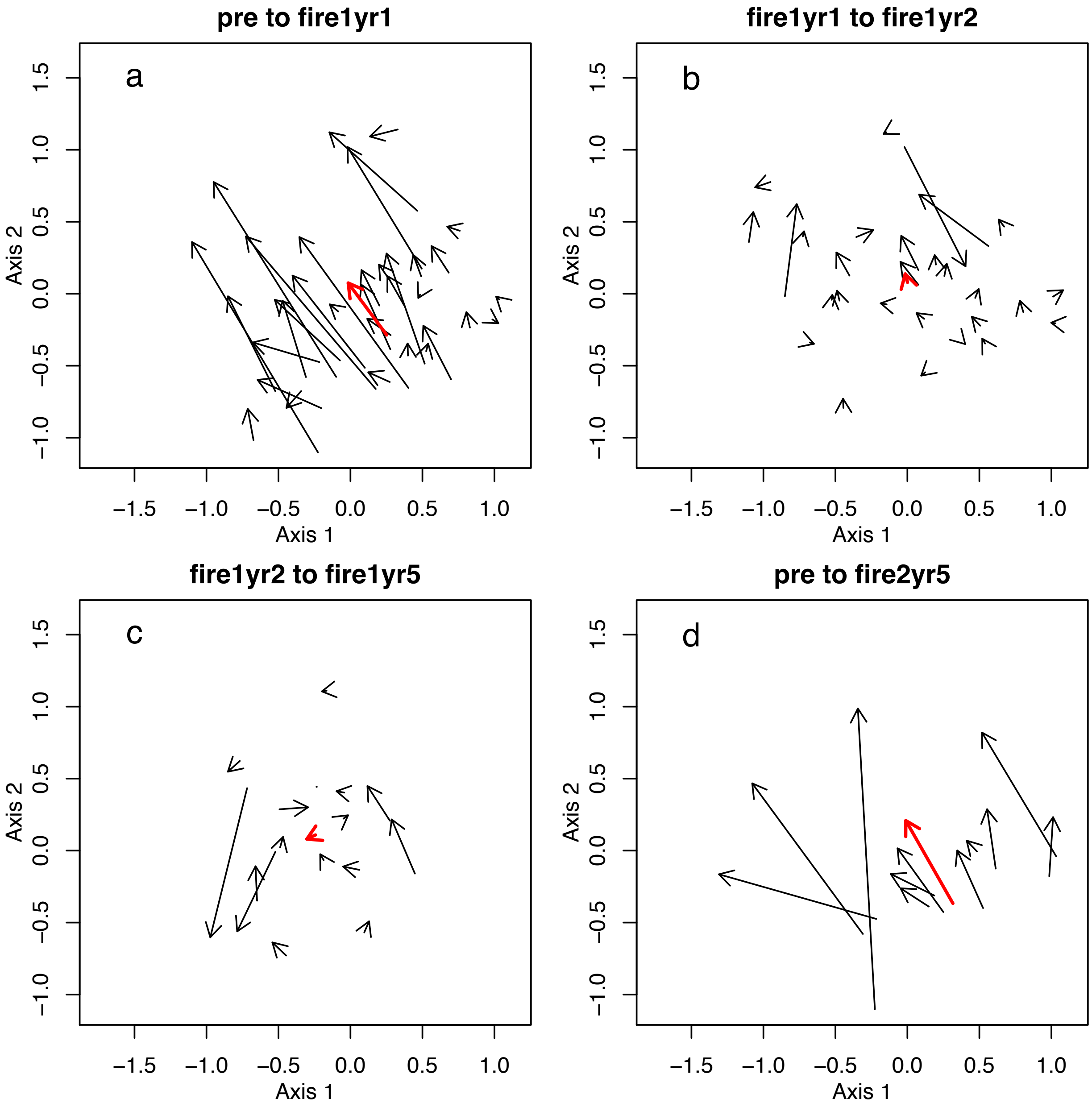

Figure 5: Results from ordinations with relative abundance data. a) plots color-coded by burned vs. unburned. b) species scores. c) environment vectors. Only significant vectors (p>0.05) are displayed. d) change vectors drawn from before fire to 5 years after the second fire. The red vector is the average vector.

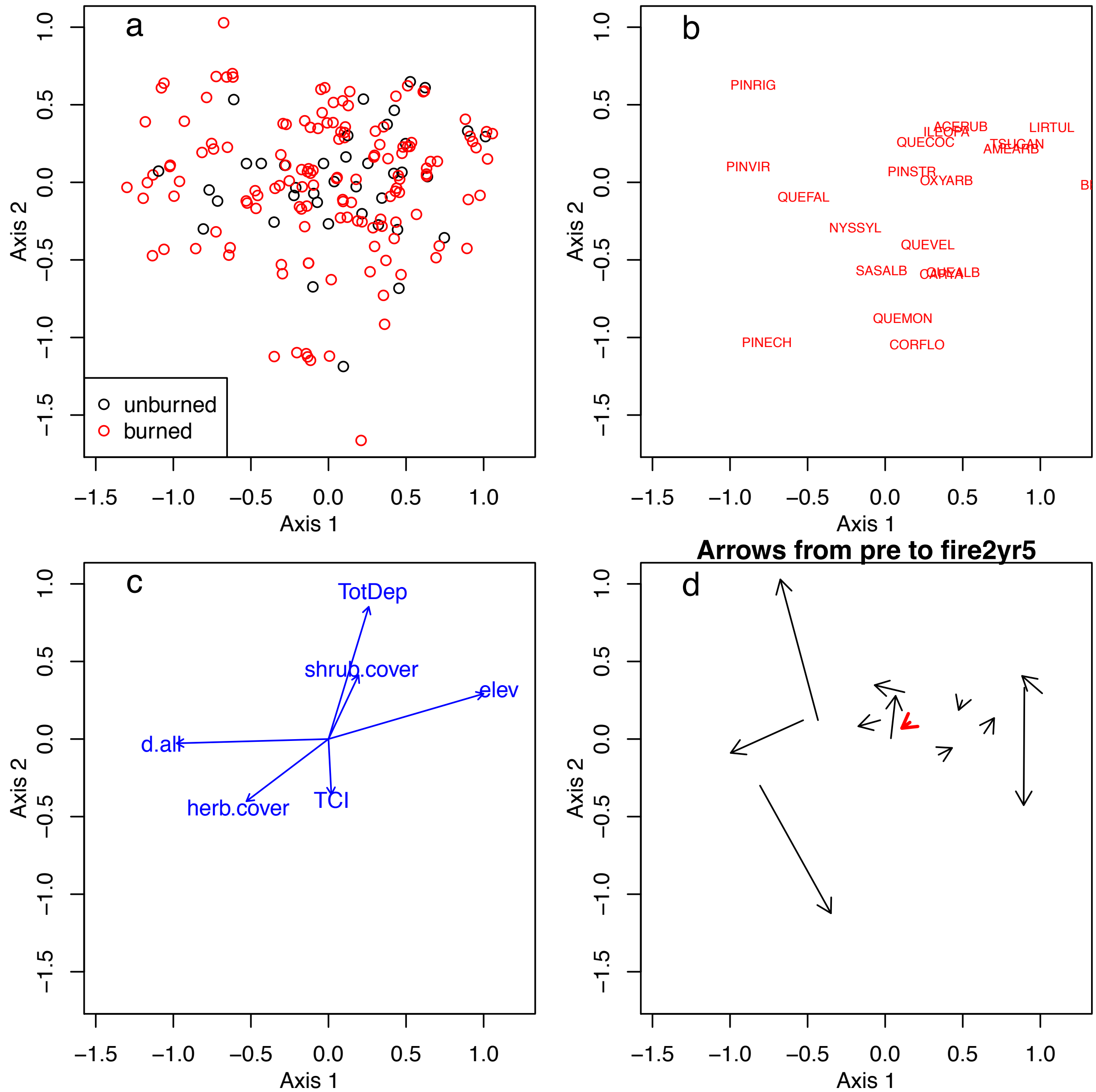

Figure 6: Example plot 1 dynamics. a) density over time. b) size class distributions over time for the whole plot. c) size class distributions by species over time.

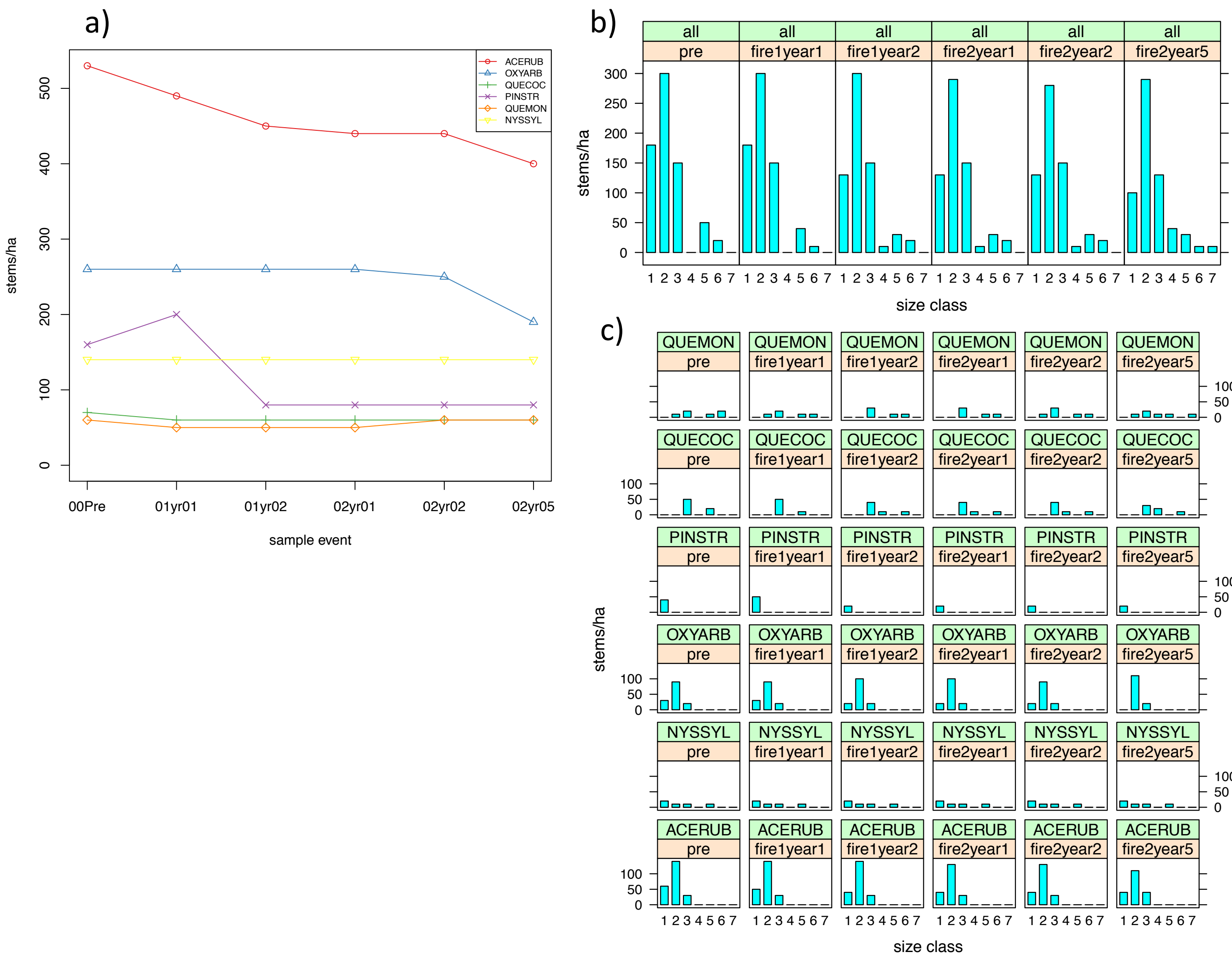

Figure 7: Example plot 2 dynamics. a) density over time. b) size class distributions over time for the whole plot. c) size class distributions by species over time.

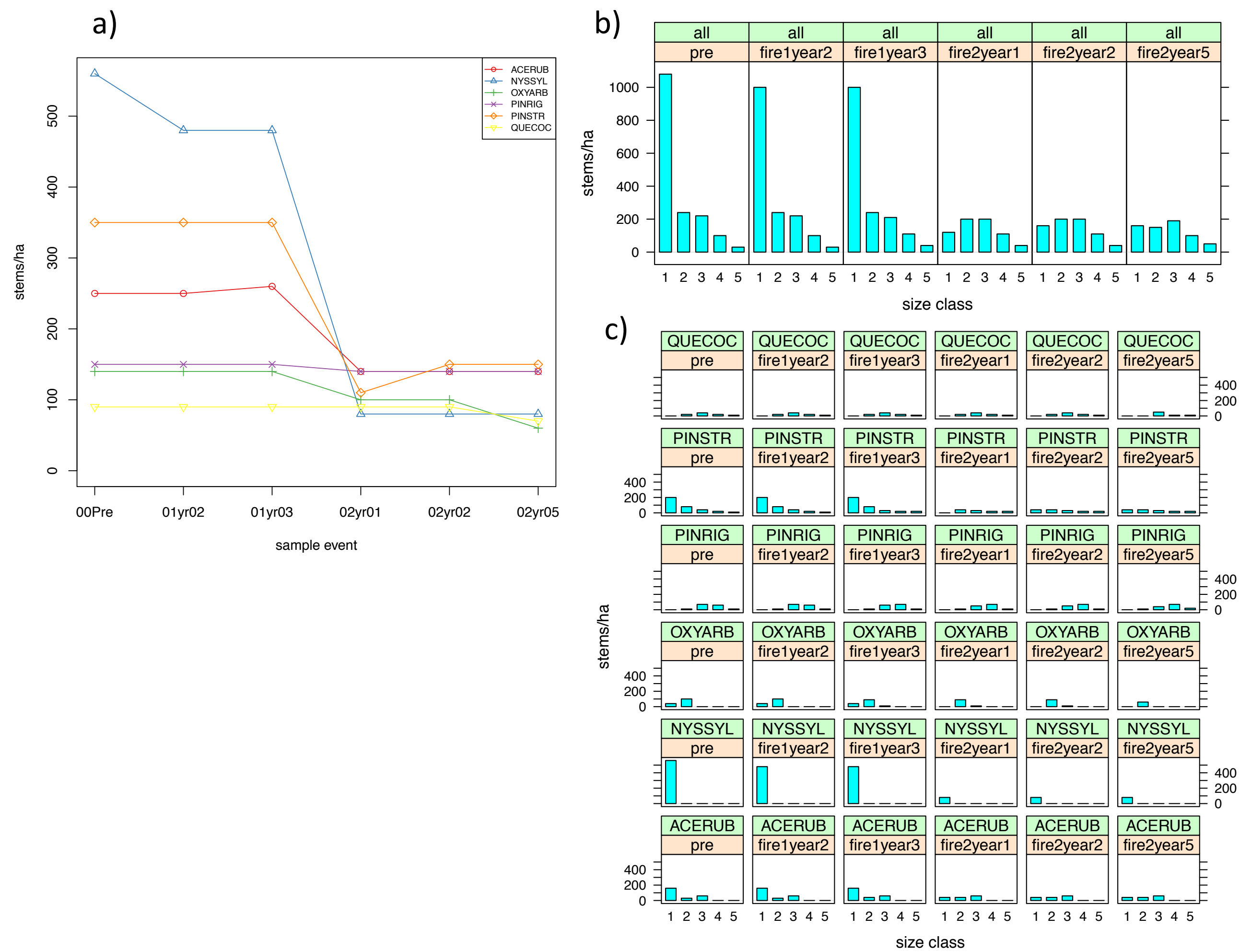

Figure 8: Example plot 3 dynamics. a) density over time. b) size class distributions over time for the whole plot. c) size class distributions by species over time.

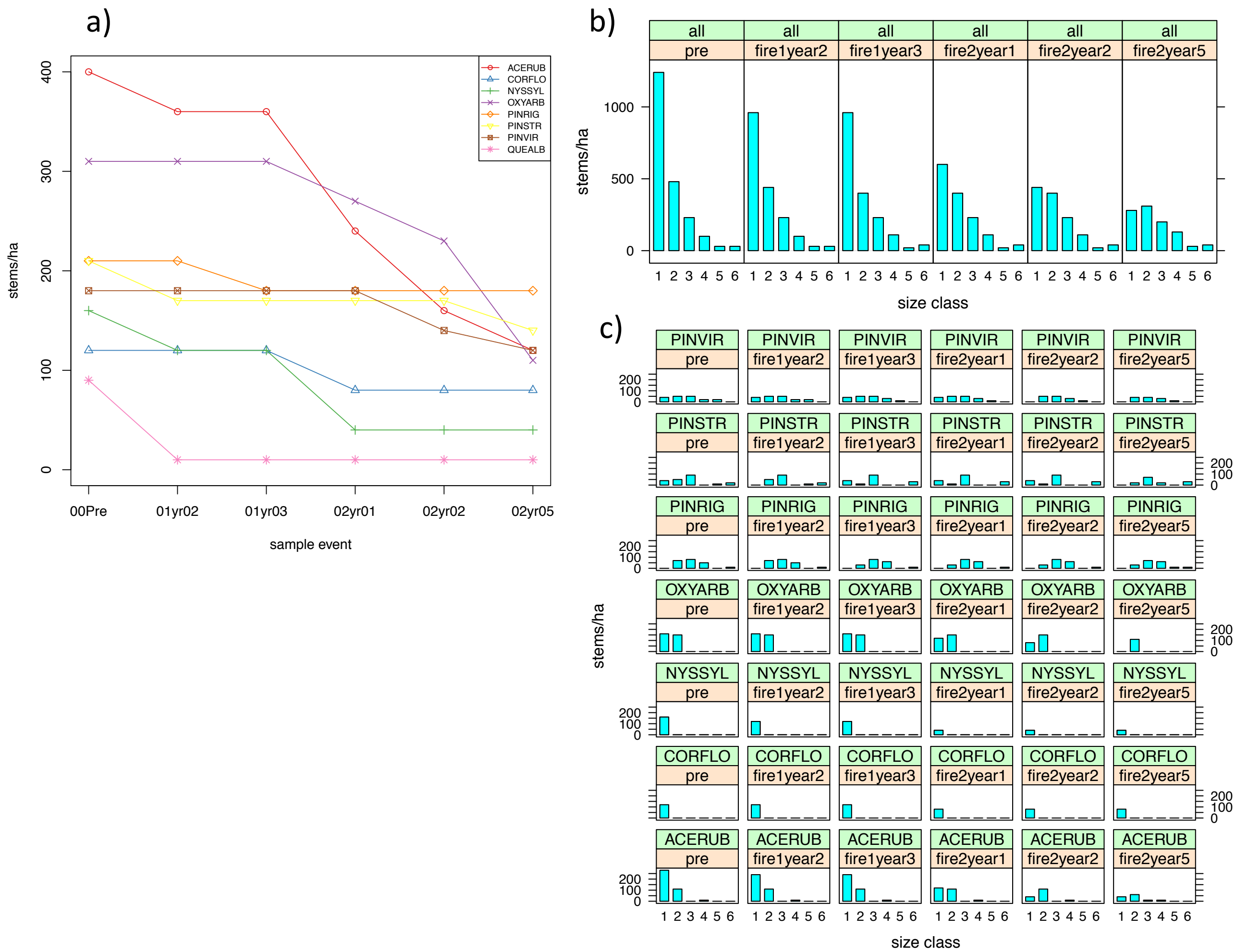

Figure 9: Example plot 4 dynamics. a) density over time. b) size class distributions over time for the whole plot. c) size class distributions by species over time.

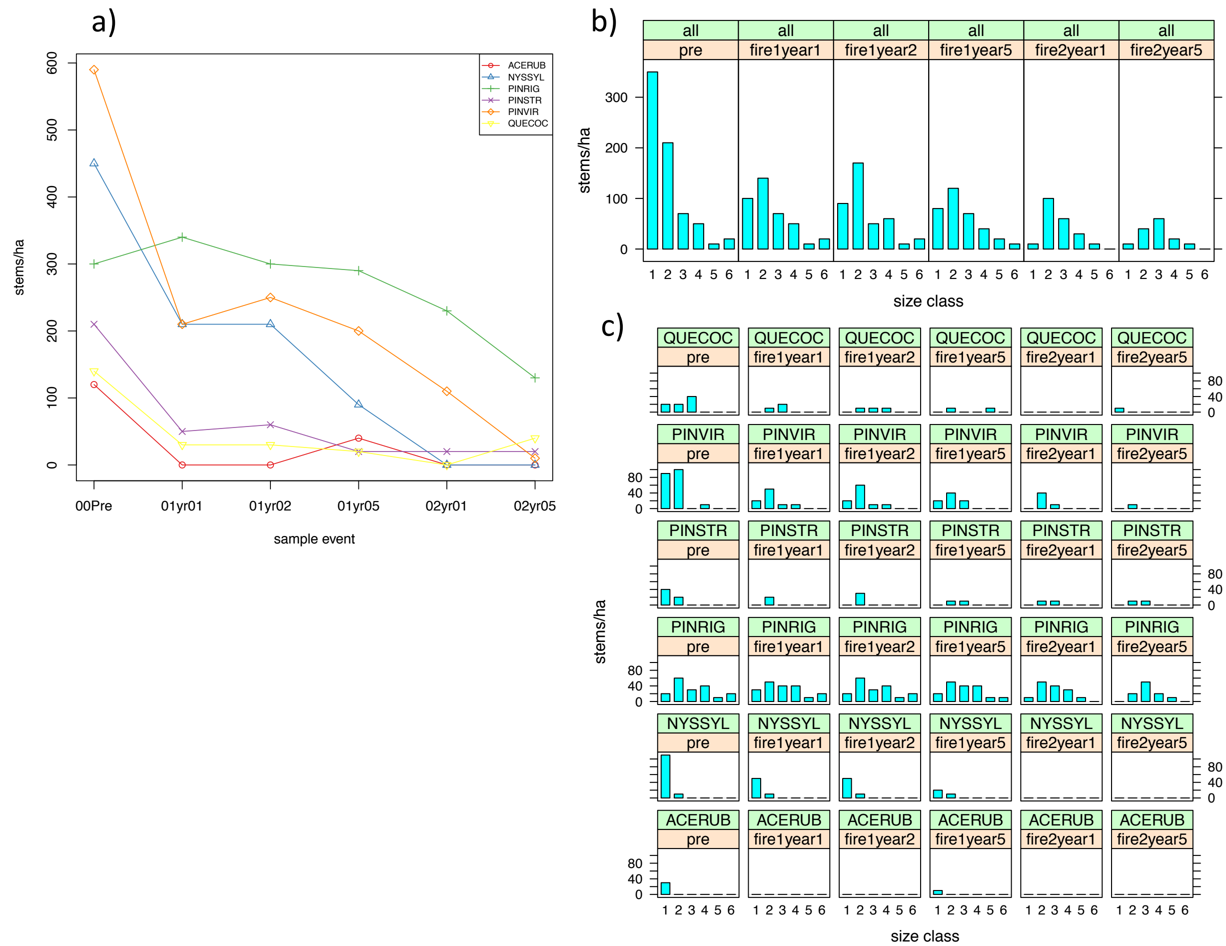

Table 2: Environmental variables used in analyses

Variable (abbreviation)	**Description**	**Mean (range)**
Elevation	Plot elevation (m)	599.0 (331.3-1046.6)
Slope	Plot slope angle (degrees)	20.5 (6.2-39.2)
Aspect	Transformed aspect (-1 to 1)	-0.03 (-1, 1)
TCI	Topographic convergence index	4.3 (3.2-7.5)
Total fuel reduction	pre-fire fuel loadings – post-fire fuel loadings (tons/ha)	12.4 (-7.1-27.0)
Litter and duff reduction	pre-fire litter and duff depth – post-fire litter and duff depth (cm)	4.3 (0.1-8.1)
Litter and duff depth	Depth of litter and duff (cm)	7.7 (1.0-21.1)

Table 3: Results from 2-dimensional NMS ordination of raw density data. Only significant correlations (*P*<0.05) are displayed.

	Axis 1	**Axis 2**
% variance explained	0.42	0.29
Cumulative variance explained	0.42	0.71
Species correlations with ordination axes		
Acer rubrum	0.618409933	-0.128551778
Amelanchier arborea	0.222045101	0
Betula lenta	0.360254789	0
Carya spp.	0.294841938	0
Cornus florida	0	0.128592439
Ilex opaca	0.161930746	-0.165641058
Liriodendron tulipifera	0.346577702	0
Nyssa sylvatica	0	-0.401680007
Oxydendrum arboretum	0.450921388	-0.239221528
Pinus echinata	-0.151168866	0
Pinus rigida	-0.362547133	-0.472927019
Pinus strobus	0.290711461	-0.399382794
Pinus virginiana	-0.323170391	-0.543135739
Quercus alba	0.329593766	0
Quercus coccinea	0.336999127	-0.165592885
Quercus falcata	0	-0.231982459
Quercus montana	0.279346872	0.278446231
Quercus velutina	0.232856056	0
Sassafras albidum	0	0
Tsuga canadensis	0.525028518	0
Environment variables correlations with ordination axes		
elevation	0.627890762	
slope		
aspect		0.237587951
TCI		
Total fuel reduction	-0.4	
Litter and duff reduction	-0.22	0.20
Depth of litter and duff	0.43	-0.44
Shrub cover	0.20	-0.35
Herbaceous cover	-0.49	0.27
Total plot density (stems/ha)	0.21	-0.47

Table 4: Results from 2-dimensional NMS ordination of relativized density data. Only significant correlations (*P*<0.05) are displayed.

	Axis 1	**Axis 2**
% variance explained	0.45	0.28
Cumulative variance explained	0.45	0.73
Species correlations with ordination axes		
Acer rubrum	0.609684673	0.408092751
Amelanchier arborea	0.24787302	0
Betula lenta	0.350567692	0
Carya spp.	0.188653614	-0.30133719
Cornus florida	0	-0.337280975
Ilex opaca	0.158331157	0.163709372

Liriodendron tulipifera	0.336041324	0
Nyssa sylvatica	-0.225864731	-0.352274936
Oxydendrum arboretum	0.394944638	0
Pinus echinata	-0.324260668	-0.440857516
Pinus rigida	-0.599578961	0.433913458
Pinus strobus	0.188581929	0.22438042
Pinus virginiana	-0.763294961	0
Quercus alba	0.255215174	-0.30211785
Quercus coccinea	0.211779247	0.204135031
Quercus falcata	-0.128683873	0
Quercus montana	0	-0.723148666
Quercus velutina	0.183569579	-0.256062881
Sassafras albidum	0	-0.210504044
Tsuga canadensis	0.57451724	0
Environment variables correlations with ordination axes		
elevation	0.665788683	0
slope		
aspect		
TCI	0	-0.250301446
Total fuel reduction	-0.32	
Litter and duff reduction		
Depth of litter and duff	0.29	-0.37
Shrub cover	0.136671	0.314555826
Herbaceous cover	-0.371464788	-0.167911677
Total plot density (stems/ha)	0	0.15

Table 5: Spearman's correlation between environment variables and plots' dissimilarity to itself calculated from raw density data for given monitoring statuses. Only significant results are displayed (significance codes: . P<0.1, * P <0.05, ** P <0.01, *** P <0.001)

Environment/fire variable	**Pre to f1y1 (n=33)**	**Pre to f1y2 (n=34)**	**Pre to f1y5 (n=20)**	**Pre to f2y1 (n=16)**	**Pre to f2y5 (n=11)**
Elevation	-0.3904*	-0.4494**	0.3925 .	-0.6912**	-0.6091 .
Aspect		0.3112 .	0.4211 .		
Slope			-0.5122*		
TCI					
Total fuel reduction	0.3431 .	0.4297*			
Litter and duff reduction		0.4154*			
Depth of litter and duff	-0.6484*	-0.7909***			-0.6242 .

Table 6: Spearman's correlation between environment variables and plots' dissimilarity to itself calculated from relativized density data for given monitoring statuses. Only significant results are displayed (significance codes: . *P*<0.1, * *P* <0.05, ** *P* <0.01, *** *P* <0.001)

Environment/fire variable	**Pre to f1y1 (n=33)**	**Pre to f1y2 (n=34)**	**Pre to f1y5 (n=20)**	**Pre to f2y1 (n=16)**	**Pre to f2y5 (n=11)**
Elevation		-0.31 .		-0.73**	-0.54 .
Aspect					
Slope			-0.48*		
TCI					
Total fuel reduction		0.40*			
Litter and duff reduction		0.42*			
Depth of litter and duff	-0.60*	-0.75***			

Table 7: Characteristics of plots used as illustrative examples.

Variable	**Plot 1**	**Plot 2**	**Plot 3**	**Plot 4**
Elevation (m)	1046.6	594.7	598.1	367.3
TCI	3.6	3.5	5.9	3.35
Aspect	0.99	-0.99	-0.91	0.49
Slope (degrees)	24.3	17.1	6.6	22.9
Total fuel reduction fire1	10.7	0.97	12.1	22.5
Total fuel reduction fire2	6.4	26.7	14.9	10.5
Litter and duff reduction fire1	5.1	0.1	1.8	6.4
Litter and duff reduction fire2	2.8	7.1	2.6	3.7
Pre-fire litter and duff depth (cm)	10.2	17.8	20.8	9.1
Post-fire1 litter and duff depth (cm)	5.3	17.6	16.0	2.6
Post-fire2 litter and duff depth (cm)	6.4	12.3	11.7	1.9
Dissimilarity pre-fire to fire2year5 (raw density data)	0.15	0.44	0.36	0.82
Dissimilarity pre-fire to fire2year5 (relative density data)	0.09	0.28	0.26	0.63
Tree species richness	10	9	15	9

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CHAPTER 4:
CONCLUSION

The research presented in the previous chapters provides an analysis of the values and goals that underlie fire management in the Southern Appalachians and an investigation into the implications of these goals for management actions, and expands our understanding of the effects of prescribed fire in the Southern Appalachians and of what drives variation in these effects. Chapter 2 illustrates how the appropriate course of action for managing fire in the Southern Appalachians depends on the goals that have been articulated. For example, managing for historical fidelity on a landscape scale would require extensive prescribed fire targeted to places where Native American burning would have been frequent, while managing solely for biodiversity could allow for a departure from historical fidelity as managers try to maintain viable populations of species of interest. This would be true even if there were no other environmental challenges to these ecosystems, but will be even more important in the face of climate change.

The results of Chapter 3 indicate that in the Great Smoky Mountains, prescribed fire is creating conditions favorable for pine and oak regeneration at some sites. Burned plots have lower density, lower shrub cover, higher herb cover, and shallower litter and duff than unburned plots. While there are not any major compositional trends apparent in the data as a whole, some plots are shifting back towards pine and oak dominance.

Prescribed fire is particularly effective at high severity and at lower elevation sites where fire sensitive species are still confined to smaller size classes and mesophication has proceeded more slowly.

Although the body of knowledge on the historic and contemporary role of prescribed fire in the Southern Appalachians has grown in recent years, there are still many questions that if answered, could improve management efforts. My results corroborate those from other studies that suggest that high severity prescribed fire is most effective at promoting pine regeneration (Elliott et al. 1999, Harrod et al. 2000, Welch et al. 2000, Jenkins et al. 2011). However, conducting high severity prescribed fires is often operationally impractical, and managers in GSMNP instead aim to conduct repeated low- to medium-severity fires (Jenkins et al. 2011). Although one or two low-severity prescribed fires do not appear to create conditions for pine regeneration, future research should examine whether this is a viable strategy for regenerating oak and pine in the long term, and should investigate the optimal fire regime (in terms of frequency, fire season, fire size, etc. over long time periods) to maintain and restore fire-dependent vegetation in the Southern Appalachians. Work to identify whether a “fire core” exists, as I hypothesize in chapter 2, would help locate sites where pine and oak forests could be maintained with minimal prescribed fire effort. More research to quantify mesophication, and particularly to identify the threshold at which prescribed fire is no longer effective in reversing mesophication could help to ensure that managers reach sites at risk of crossing this threshold before it is too late and that efforts are not wasted at sites where they will not be effective. More research into the effects of climate change on fire regimes and vegetation in the Southern Appalachians will help managers better plan for an uncertain future.

Promoting fire-dependent vegetation in the Southern Appalachians will require a long-term commitment to active management and prescribed fire. Because of the history of anthropogenic fire, it is unlikely that restoring vegetation structure and composition and then allowing "background levels" of wildfire to resume will be enough to maintain fire dependent vegetation. Continued monitoring, like that of the fire effects monitoring program in GSMNP, is and will remain essential for understanding the effects of prescribed fire, for determining effective fire regimes for management, for assessing and observing the effects of climate change in real time, and for practicing adaptive management. Efforts should be made to continue and expand such long-term monitoring programs in GSMNP and elsewhere in the Southern Appalachians to ensure that regardless of the goals and values managers embrace, they have the data and the science to achieve those goals successfully.

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