ABSTRACT OF THESIS

01-3-3-14

THE EFFECTS OF PRESCRIBED FIRE ON STAND STRUCTURE, CANOPY COVER AND SEEDLING POPULATIONS IN OAK DOMINATED FORESTS ON THE CUMBERLAND PLATEAU, KY

Prescribed fire has become an increasingly common management tool in upland oak forests throughout the southern Appalachians. While prescribed fire's potential for enhancing advanced oak regeneration through temporary reductions in understory competition is widely accepted, its long-term efficacy for promoting oak seedling success has not been demonstrated. This thesis describes research on two components of the fire-oak hypothesis: 1) the long-term effects of repeated prescribed fire on survival and growth of oak and red maple seedlings, and 2) the initial effects of prescribed fire on stand structure, canopy cover, and seedling survival. After five years of periodic prescribed fire, red maple seedlings suffered from higher mortality than oak seedlings, yet showed larger growth responses. In a related study, initial effects of prescribed fire included reduced understory stem density and basal area and increased basal sprouts. Canopy cover was temporarily reduced, but returned to pre-treatment levels within one year. Sassafras seedlings had the highest survival, followed by *Erythrobalanus* oak species, *Leucobalanus* oak species, and finally, maples. Together, these studies demonstrate that while some effects of prescribed fire, such as decreased stem density and increased light can be seen immediately, long-term data are needed to adequately understand fire's role in oak forests.

KEYWORDS: Oak regeneration, prescribed fire, *Quercus* spp., *Acer rubrum*, stand structure

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THE EFFECTS OF PRESCRIBED FIRE ON STAND STRUCTURE,
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Director of Thesis

Director of Graduate Studies

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2005

THE EFFECTS OF PRESCRIBED FIRE ON STAND STRUCTURE, CANOPY COVER AND SEEDLING POPULATIONS IN OAK DOMINATED FORESTS ON THE CUMBERLAND PLATEAU, KY

THESIS

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Forestry at the University of Kentucky

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2005

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CHAPTER ONE

Introduction to the Fire-Oak Hypothesis

Fire is an important disturbance agent that can have direct impacts on the structure, species composition, and successional patterns of natural communities (Pyne 1982, 1984; Abrams and Nowacki 1992). While natural fire is a climate driven disturbance, humans have effectively manipulated it for thousands of years, and therefore played an important role in the manipulation of those ecosystems affected by fire (Pyne 1982). The oak-hickory (*Quercus-Carya* spp.) dominated forests of the southern Appalachians are an artifact of such human-fire interactive manipulations.

Oak (*Quercus* spp.) species have dominated the southern Appalachians since the mid-Holocene (6,000 yr BP) (Delcourt and Delcourt 1987). Pollen and charcoal records, coupled with evidence of prehistoric human occupation show that aboriginal use of fire influenced the proliferation of oaks in southern Appalachian forests (Delcourt and Delcourt 1996). Historical documents confirm that aboriginal use of fire continued until European settlement (Pyne 1982; Russell 1983; Lorimer 1985; Elliott et al. 1999). European settlers also promoted the creation of today's oak-dominated stands through burning, logging, and agricultural practices (Lorimer 1993).

Oaks are typically described as a mid-successional species and their continued dominance is dependant upon frequent disturbances (Abrams 1992). Oaks are considered to be fire-adapted because of their relatively thick bark, resistance to fire scars, and prolific sprouting capabilities (Lorimer 1985). These traits enable oaks to survive low intensity fires, thus providing a competitive advantage over faster growing, shade tolerant species which are more vulnerable to fire (Lorimer 1985; Van Lear and Watt 1993; Abrams 1998; Huddle and Pallardy 1999; Elliot et al. 1999). It is believed that the frequent, low intensity fire regime believed to have been present at the time of European settlement, and the continued disturbance by settlers through extensive clear cutting and fire, selected for the dominance of oak species.

While oaks currently dominate overstory strata in upland southern Appalachian forests, studies have shown increases in fire-sensitive, shade tolerant species such as red maple (*Acer rubrum* L.) and sugar maple (*A. saccharum* Marshall.) (Abrams and Nowacki 1992; Lorimer et al. 1994; Abrams 1998; Delcourt and Delcourt 1998; Harrod et al. 1998; Blankenship and Arthur 1999). This proliferation of shade tolerant, fire-sensitive species has been accompanied by a documented decline in advanced oak regeneration and subsequent recruitment into larger size classes (Carvel and Tryon 1961; Lorimer 1985; Abrams 1992; Lorimer 1993; Van Lear and Watt 1993). This evidence suggests that with continued succession, and in the absence of disturbance, stands presently dominated by oak will not replace themselves. Based on the understood relationship between historical fire regimes and the oak-dominated stands of today, it has been hypothesized that prescribed fire has the potential to be an effective tool for reducing competition caused by fire-sensitive, shade tolerant species, thus improving natural oak regeneration.

The widespread acceptance of the fire-oak hypothesis has led to an increase in the use of prescribed fire as a management tool in the oak forests of the eastern United States over the past twenty years (Yaussy 2000). In light of this increase in the use of prescribed fire, data regarding the effectiveness of fire for enhancing oak regeneration is vital. This thesis describes research on two related components of the fire-oak hypothesis: 1) the long term effects of prescribed fire on survival and growth of oak and red maple seedlings, and 2) initial effects of prescribed fire on stand structure, canopy cover, and seedling survival.

CHAPTER TWO

The Effects of Periodic Prescribed fire on Oak (*Quercus spp.*) and Red Maple (*Acer rubrum* L.) Seedlings on Xeric Ridgetops in Eastern Kentucky

1. Introduction

In 1995, a study was initiated in collaboration with U.S. Forest Service personnel to address current questions regarding the potential role of fire to modify stand structure in a manner that promotes oak seedling success. To date, this research has documented the effects of prescribed fire on: 1) soil nutrients and microbial biomass (Blankenship 1999), 2) encroaching eastern white pine (*Pinus strobus* L.) (Blankenship and Arthur 1999), 3) the physiology and growth of oak (*Quercus* spp.) and red maple (*Acer rubrum* L.) seedlings (Gilbert et al. 2003), 4) understory light availability (Chiang et al. in review), and 5) stand structure (Blankenship and Arthur in preparation).

While other studies have looked at the short term effects of fire on seedlings through in field and greenhouse methods (Reich et al. 1990; Huddle and Pallardy 1996, 1999), there is a lack of information regarding the long term effects of fire on seedling populations. To address this gap in information, a long term seedling population study was initiated in 1998 on the Red River Gorge Geological Area (RRGGA) of the Daniel Boone National Forest (DBNF). Utilizing the experimental design of the previously mentioned study on the long-term effects of low-intensity prescribed fires, my objectives were to 1) quantify the effects of repeated prescribed fires on survival and growth of oak and red maple seedlings, and 2) determine if prescribed fire can enhance oak seedling survival and growth.

2. Site description

This study was conducted on two non-contiguous ridges in the RRGGA on the Stanton District of the DBNF. The RRGGA is included in Braun's (Braun 1950) Cliff Section of the Cumberland Plateau. The two ridges, Klaber and Whittleton, are located in Menifee and Powell Counties, Kentucky, USA. Stands on both ridges are dominated by chestnut oak (*Q. prinus* L.) and scarlet oak (*Q. coccinea* Muenchh.) in the overstory strata, with some white oak (*Q. alba* L.) and black oak (*Q. velutina* Lam.) present. Pines (*P. rigida* Miller and *P. echinata* Miller) and red maple also comprise a minor part of the overstory. The midstory is dominated by red maple, with white pine, sassafras (*Sassafras albidum* (Nutt.) Nees.), sourwood (*Oxydendrum arboretum* (L.) DC.), black gum (*Nyssa sylvatica* Marshall), and some oaks (Kuddes-Fischer and Arthur 2002). Oak species are well represented in the seedling stratum, with scarlet and chestnut oak being the two dominant species (1800 and 900 stems/ha, respectively); red maples also comprise a large portion of the seedling layer (10675 stems/ha). These stands are estimated to be 70 years old, regenerated from a forest that was described as "heavily culled" in the 1930's (Collins 1975; Blankenship and Arthur, in preparation).

The underlying geological substrate is comprised of siltstones and shales of the Upper and Lower members of the Lee formation (Weir and Richards 1974). Soils of Klaber Ridge are classified as well drained, Latham-Shelocta silt loams of the Typic and Aquic Hapludults subgroups (Avers et al. 1974). Soils of Whittleton Ridge are classified as well drained, Gilpin silt loams of the Typic Hapludult subgroup (Hayes 1993). The regional climate is humid, temperate, and continental with warm to hot summers and mild winters (Hill 1976). Mean annual temperature is 12• C, with mean daily temperatures ranging from 31° C in July to 0° C in January. The mean growing season is 176 days. Mean annual precipitation is 130 cm (Foster and Conner 2001).

3. Methods

3.1. Experimental design and fire prescription

In 1995, a long-term study of the effects of prescribed fire was established in cooperation with USDA Forest Service personnel on the Stanton District of the Daniel Boone National Forest. Klaber and Whittleton ridges were selected as replicates based on: 1) perceived need for prescribed fire to control white pine encroachment, 2) accessibility to Forest Service prescribed fire personnel, and 3) lack of recent timber management or fire disturbance (Blankenship and Arthur, in preparation; Jorge Hershel, U.S. Forest Service, Stanton, Ky., pers. comm.). Each ridge was divided into three treatments areas: 1) fire-excluded, 2) to be burned initially in 1995, and 3) to be burned initially in 1996. Throughout the successive eight years, prescribed fires were applied to each ridge periodically as weather and logistics allowed in an "adaptive management" approach (Blankenship and Arthur, in preparation). Because of Whittleton ridge's location between an important regional highway and Natural Bridge State Park, and concerns about smoke management, the treatment area on this ridge, originally slated for burning in 1996, was burned instead in 1997. As of the year 2003, treatments on each ridge included: one fire-excluded site, one site burned twice (2x), and one site burned three times (3x, Table 1).

All prescribed fires were conducted by USDA Forest Service personnel of the Stanton Ranger District. Backing and flanking fires were ignited using drip torches. If fire intensities were too low (<0.3 m flame length), point-source and strip fires were utilized (Richardson 1995). Date, weather, and fire conditions were recorded for each prescribed fire (Table 1).

3.2 Seedling Population

To test oak (*Quercus* spp.) and red maple seedling response to prescribed fire treatments, a long-term seedling population study was established in the winter of 1998-99. Prior to selection of seedlings, all treatment areas (1x and 2x burned) had been burned once. Within the three treatment areas on each ridge, well established seedlings (non-germinants) were identified, mapped, and marked with a small (<5 cm radius) aluminum wire ring and an aluminum tag engraved with a unique identifying number. A total of 781 chestnut oak, scarlet oak, and red maple seedlings were tagged. Oak seedlings often dieback repeatedly to ground level and then resprout, resulting in root systems that are older than above ground stems (Merz and Boyce 1956; Crow 1994). For the purposes of this study, "seedlings" were defined as stems <50 cm tall.

3.2.1. Seedling survival

Marked seedlings were classified as "dead" or "alive" in each year. Seedlings were classified as dead if no stem was visibly present, or if the remaining stem showed no visible signs of life. A few seedlings resprouted after being classified as "dead." This problem has been cited in similar seedling population studies, especially with oak species (Jones and Sharitz 1998; Marks and Gardescu 1998). Seedlings that resprouted after they were classified as "dead" were reclassified as "alive" and maintained in the population study. Due to thick understory vegetation cover and the propensity for top kill after fire, some seedlings were not located during annual censusing. These seedlings were classified as "lost." Seedlings that were classified as "lost" and then found alive the following year were reclassified as "alive." Those not found again were eliminated for all years of the study to avoid confusion between seedlings that were known to have died, and those whose status was unclear.

3.2.2. Seedling growth parameters

Measurements of seedling height and annual growth were taken at the end of each growing season (August-September) for five years (1999-2003). Basal diameter was measured annually starting in 2001. In most cases oak and red maple seedlings experience stem dieback after fire and are capable of resprouting. Due to this physiological adaptation to fire disturbance, relative growth rates were not calculated. Seedling height was measured as the total length (cm) of live stem tissue above ground level (if multi-stemmed, the longest leader). Annual growth was defined as stem elongation (cm) from the last visible bud scar (if multi-stemmed, the longest leader). Although seedlings are capable of undergoing multiple flushes or growth periods during the course of a growing season, this is uncommon under conditions of limited resources, such as those found under the intact forest canopies of this study (Johnson et al. 2002). For the purpose of this study, basal diameter was defined as the diameter (mm) of the stem where it exited the soil. Based on the long-term nature of this study and the annual recurrences of measurements, the organic layer was not removed to expose the true root collar. Basal diameter was the average of two perpendicular measurements taken with a digital caliper.

4. Data analysis

4.2. Seedling survival

Seedling survival was analyzed in 2003, after five years of monitoring. A small number of seedlings (41, total) were lost throughout the course of this study and were not included in analyses, leaving 740 seedlings in the analysis of survival. Seedling survival was analyzed by a Chi-Square using the FREQ procedure of SAS (SAS Institute 1995). Due to unequal numbers of species within each treatment area on each ridge, seedling survival was analyzed using individual seedlings within treatment areas as the experimental unit.

4.3. Seedling growth parameters

Seedling growth parameters were analyzed in two different ways. Due to the previously mentioned logistical constraints, prescribed fires were applied to each ridge in different years (Table 1). To equalize the effects of time since burn on seedling response, seedling height, annual growth, and basal diameter were analyzed two growing seasons after the last fire event on each ridge. For this analysis Klaber seedlings were analyzed using the 2001 data, while Whittleton seedlings were analyzed using the 2003 data. To examine long term trends in seedling responses to prescribed fire, another analysis was run using data at the end of five years (2003 for both ridges). Growth parameter data were found to be non-normally distributed and so were transformed using the natural logarithm. For presentation purposes, data were back transformed.

The experimental design was a randomized block design, with ridge as the blocking factor. Treatment design was a 3 x 3 factorial, with three treatments and three species. While prescribed fires were applied to entire treatment areas, seedlings were treated as the experimental unit, resulting in a pseudoreplicated design. Mortality of 140 seedlings over the course of the study resulted in statistical analyses of growth parameters based on 600 seedlings. Seedling height, annual growth, and basal diameter were analyzed by analysis of variance (ANOVA) using the MIXED procedure of SAS (SAS Institute 2000). A Bonferroni-Holm post-hoc analysis was conducted between pairs of means when an ANOVA F-test was significant. P values <0.05 were considered to be statistically significant.

5. Results and Discussion

5.1. Effects of periodic prescribed fire on seedling survival

On fire-excluded sites, 11.9 % of seedlings died and survival rates among the three seedling species were similar (p>0.09), suggesting that in the absence of fire, forest conditions did not afford any noticeable advantage to red maple seedlings in terms of survival.

After five years, prescribed fires have had significant negative effects on seedling survival (p=0.0010, Figure 1). While neither chestnut nor scarlet oak survival was affected by prescribed fire (Figure 1), red maple survival was reduced (p=0.0007, Figure 1). On 2x and 3x burned sites, red maple survival was lower than on fire-excluded sites (p=0.0417 and p=0.0001, respectively). Although red maple survival tended to decrease with additional burns, survival on 2x and 3x burned sites was not significantly different (p=0.0721, Figure 1). Red maple survival on 2x burned sites was similar to the survival of both scarlet and chestnut oaks (p=0.1410 and p= 0.3104), but on 3x burned sites red maple survival was lower than scarlet oak survival (p=

0.0177), and to a lesser extent chestnut oak, although not significant (p=0.1876, Figure 1).

Although differences among years and between ridges were not tested statistically, Figure 2 suggests that on burned sites, seedling mortality occurred immediately after fire events but remained relatively constant in the intervening years. One exception to this trend was chestnut oak. Although not significant, chestnut oak survival declined across the years on both fire-excluded and burned sites (Figure 2), suggesting sources of mortality not directly related to prescribed fire. In a previous study, Chiang (2002) found that although chestnut oak seedlings on the two ridges had significantly higher root starch concentrations than both scarlet oak and red maple seedlings, they had relatively lower total root masses, and a high percentage of chestnut oak seedlings had hollow root systems. Low root mass could affect a seedling's long term survival, both in the presence and absence of fire disturbance (Sander 1977; Pope 1993; Walters and Reich 1996).

My results show that three periodic prescribed fires significantly reduced the survival of red maple seedlings in comparison to oak species, concurring with other studies which have shown the effectiveness of repeated fires in preferentially reducing the relative abundance of red maple seedlings in favor of oak species (Reich et al. 1990; Huddle and Pallardy 1996; Arthur et al. 1998; Huddle and Pallardy 1999; Elliott et al. 2004; Jackson and Buckley 2004). These results further support the hypothesis that repeated prescribed fires can selectively reduce the abundance of competitor species, such as red maple, in the seedling stratum.

5.2. Seedling height and annual growth

Analyses of seedling height and annual growth two years after the last fire (Figure 3) showed the same general trends as the analyses of height and annual growth after five years of periodic prescribed fires, regardless of time since fire (Figure 4). This suggests that the intervals at which fire disturbances occurred on these study areas were not important; only the analyses of seedling height and growth at the end of five years are discussed here.

Oak and red maple seedling heights were similar on fire-excluded sites (p>0.05, Figure 4a). When seedling heights within species were compared across treatments, chestnut oaks had similar heights on all treatments, while scarlet oaks were taller on 3x burned sites than 2x burned sites (p=0.0020, Figure 4a). Red maple seedlings on 3x burned sites were taller than their counterparts on 2x burned sites (p=0.0020) and fire-excluded sites (p< 0.0001, Figure 4a). When species were compared within treatments, red maple seedlings were significantly taller than oaks on both the 3x and 2x burned sites (p<0.0005); oak seedling heights were similar to each other on the burned sites (Figure 4a). Prescribed fire increased average annual growth for all species. While oak seedlings on burned sites had larger annual growth than their counterparts on fire-excluded sites ($p<0.0002$, Figure 4b), red maple annual growth increased with added burns: 3x > 2x > fire-excluded sites (Figure 4b). The differences observed in seedling height were driven by seedling annual growth (Figure 4a, 4b).

My results showed that prescribed fires resulted in higher annual growth for all species. While all oak seedlings on burned sites were as tall as or taller than their counterparts on fire-excluded sites, red maples were significantly taller than oaks following five years of periodic prescribed fire. These results were comparable to other studies on the responses of oak regeneration to fire. In a study previously conducted on the Whittleton ridge, Gilbert et. al. (2003) found that height relative growth rates for both oaks and red maples were higher on burned sites, and that red maple seedlings were taller than oaks; these differences in height lasted up to three growing seasons after a single prescribed fire. Brose and Van Lear (1998) reported that fire enhanced height growth in oak, and Rieske (2002) found that one year after fire chestnut oak seedlings on burned sites had four times higher height growth than on non-burned sites.

The fact that surviving red maple seedlings were taller than oaks on burned sites, but not on fire-excluded sites suggested that the increase in height was due to fire. Both red maple and oak seedlings are capable of resprouting after fire. Oak seedlings tend to allocate more energy to storage, and red maples allocate more to height and root growth following fire or other disturbances (Huddle and Pallardy 1999; Brose and Van Lear 2004). While this difference in carbon allocation, and the vigorous sprouting ability afforded to oaks by carbon storage, was responsible for relatively high oak survival rates after disturbances such as fire (Reich et al. 1990; Huddle and Pallardy 1999; Brose and Van Lear 2004). Red maples' opposite pattern of allocating more energy to shoot growth gave surviving red maples a height advantage over oaks after fire.

5.3. Basal diameter

When basal diameter was analyzed two growing seasons after the last fire, there were no differences among species (Figure 5a). Seedlings on 2x burned sites had significantly smaller average basal diameters than seedlings on both fire-excluded and 3x burned sites ($p= 0.01$ and $p < 0.0001$ respectively). When analyzed after five years, repeated prescribed fires increased basal diameters across all three species, yet resulted in larger basal diameters in red maple compared to oak species (Figure 5b). Seedlings on 3x burned sites had significantly larger basal diameters than their counterparts on the fire-excluded and 2x burned sites (both $p < 0.0001$); and red maple seedlings had larger basal diameters than both chestnut oak and scarlet oak seedlings across all treatments ($p < 0.0001$ and $p = 0.0155$ respectively). These results were consistent with an earlier study conducted on Whittleton ridge in which Gilbert et al. (2003) found that basal diameter relative growth rates (RGR) were higher for seedlings on burned sites and attributed the increase to higher foliar nitrogen after fire. However, since basal diameters were not measured in this study until 2001, there is no empirical evidence that seedling in this study had similar basal diameters at the start of the study; estimates of initial seedling basal diameter, determined by a linear regression between seedling height and basal diameter in 2003, suggested that red maple basal diameters were larger than basal diameters of oak species on burned sites in 1999.

Basal diameter can be an important indicator of future success in the presence of prescribed fire. In central Virginia, Brose and Van Lear (2004) found that as both the depth and diameter of seedling root collar increased, survival after fire increased. Oak seedlings tend to have larger root systems and deeper root collars than red maple due to silvical characteristics, such as hypogeal germination and an emphasis on root development over shoot growth (Kolb et al. 1990; Brose and Van Lear 2004). In the absence of disturbance, maple seedlings can establish well developed root systems that are large enough and located far enough below the surface to allow for their continued survival after fire (Brose and Van Lear 2004). This might account for why after three prescribed fires, red maple survival, while significantly reduced, was still moderately high at 68%.

6. Conclusions

In summary, repetitive, low-intensity prescribed fires effectively decreased red maple survival, yet those red maples that did survive tended to be more effective at reallocating resources to above ground growth after fire than oak seedlings (Figure 3, 4, and 5). It has been shown that prescribed fire can be used as an effective management tool for the control of shade tolerant, fire sensitive competitor species such as red maple (Lorimer 1985; Arthur et al. 1998; Barnes and Van Lear 1998; Loftis 2004). The results in this study raise concern that the current prescribed fire regime practiced by many land managers is acting as a selection agent for a cohort of more competitive red maple seedlings that will contribute to the exclusion of new oak recruitment into the advanced regeneration pool. Most prescribed fires used by managers in this region are dormant season, low intensity surface fires. Such was the case with the prescribed fires applied to these research sites; all were relatively low in intensity and took place in early spring, before seedlings became physiologically active. Since repeated fires negatively affected red maple survival, yet increased their overall growth, and oak seedlings showed few detrimental outcomes in response to multiple fires, my recommendation is that to further reduce competitor species, attempts be made to use more intense, early growing season fires. Fires of higher intensity would have greater potential to kill larger competitor stems, and result in lower growth responses in surviving stems due to the additional stress caused by burning after seedlings are physiologically active. Table 1. Median fire temperature ranges (°C) and weather conditions of each prescribed fire event. Dashed lines indicate years without burning.

Klaber 3x burn (1995, 1999, 2000)	**1995**	**1996**	**1997**	**1999**	**2000**	**2002**
Burn date	3/17	-	-	3/26	3/30	-
T° @ 15 cm	204-315^0	-	-	400-499^0	500-659^0	-
T° @ surface	316-398^0	-	-	198-249^0	250-399^0	-
Air T°	21^0	-	-	8^o	9^o	-
Rel. Hum.	36%	-	-	40%	35%	-
Flame length	0.3-0.9 m	-	-	0.3-0.9 m	0.3-0.9 m	-

Klaber 2x burn (1996, 2000)	**1995**	**1996**	**1997**	**1999**	**2000**	**2002**
Burn date	-	3/13	-	-	3/30	-
T° @ 15 cm	-	399-481^0	-	-	500-569^0	-
T° @ surface	-	204-315^0	-	-	250-399^0	-
Air T°	-	18^0	-	-	9^o	-
Rel. Hum.	-	25%	-	-	35%	-
Flame length	-	0.3-0.9 m	-	-	0.6-0.9 m	-

Whittleton 3x burn (1995, 1999, 2002)	**1995**	**1996**	**1997**	**1999**	**2000**	**2002**
Burn date	3/15	-	-		-	4/15
T° @ 15 cm	316-398^0	-	-	500-659^0	-	149-197^0
T° @ surface	204-315^0	-	-	198-249^0	-	253-398^0
Air T°	23^0	-	-	11^0	-	na
Rel. Hum.	29%	-	-	40%	-	41%
Flame length	0.3-0.9 m	-	-	0.3-0.9 m	-	0.15-0.6 m

Whittleton 2x burn (1997, 2002)	**1995**	**1996**	**1997**	**1999**	**2000**	**2002**
Burn date	-	-	3/24	-	-	4/15
T° @ 15 cm	-	-	na	-	-	149-197^0
T° @ surface	-	-	na	-	-	253-398^0
Air T°	-	-	13^0	-	-	na
Rel. Hum.	-	-	45%	-	-	41%
Flame length	-	-	0.3-0.6 m	-	-	0.3-0.6 m

Figure 1: Seedling survival after five years of monitoring (2003, n=740). Different uppercase letters denote significant differences among treatment sites within species. Different lowercase letters denote significant differences among species and within treatment sites

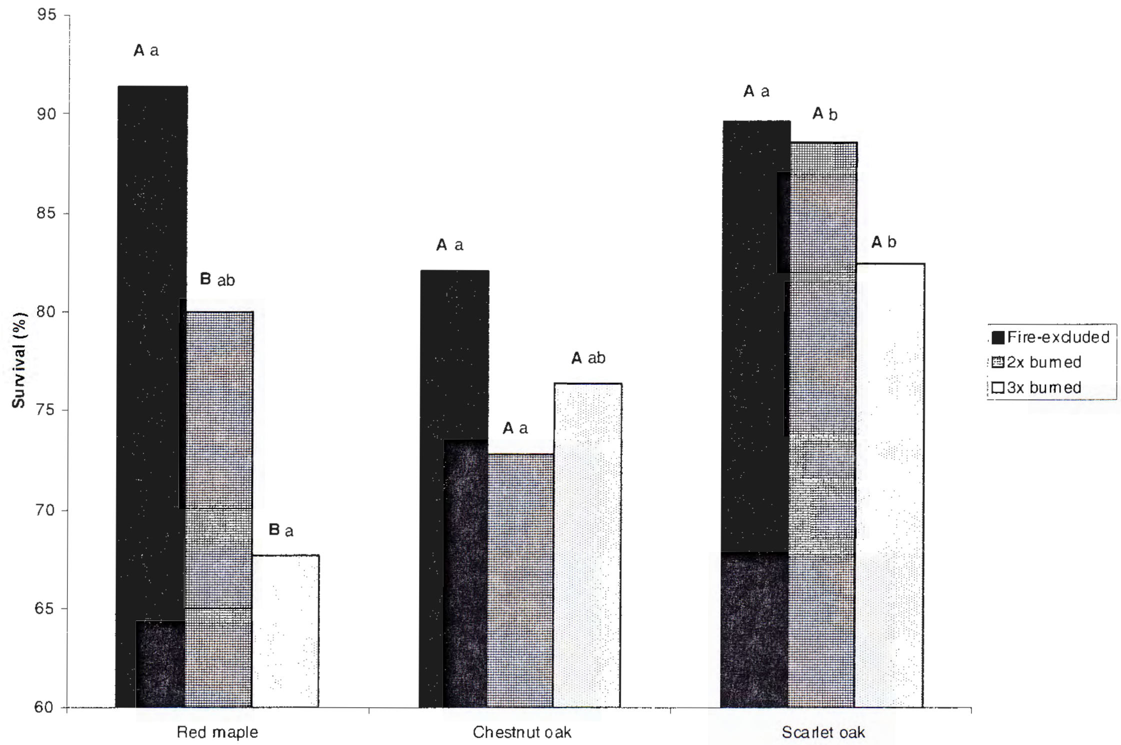

95
90
85
80
75
70
65
60
Survival (%)
A a
B ab
B a
A a
A a
A ab
A a
A b
A b
Red maple
Chestnut oak
Scarlet oak
Fire-excluded
2x burned
3x burned

Figure 2: Seedling survival on Klaber and Whittleton ridges and treatment sites across fire years of monitoring. Arrows indicate prescribed fire events.

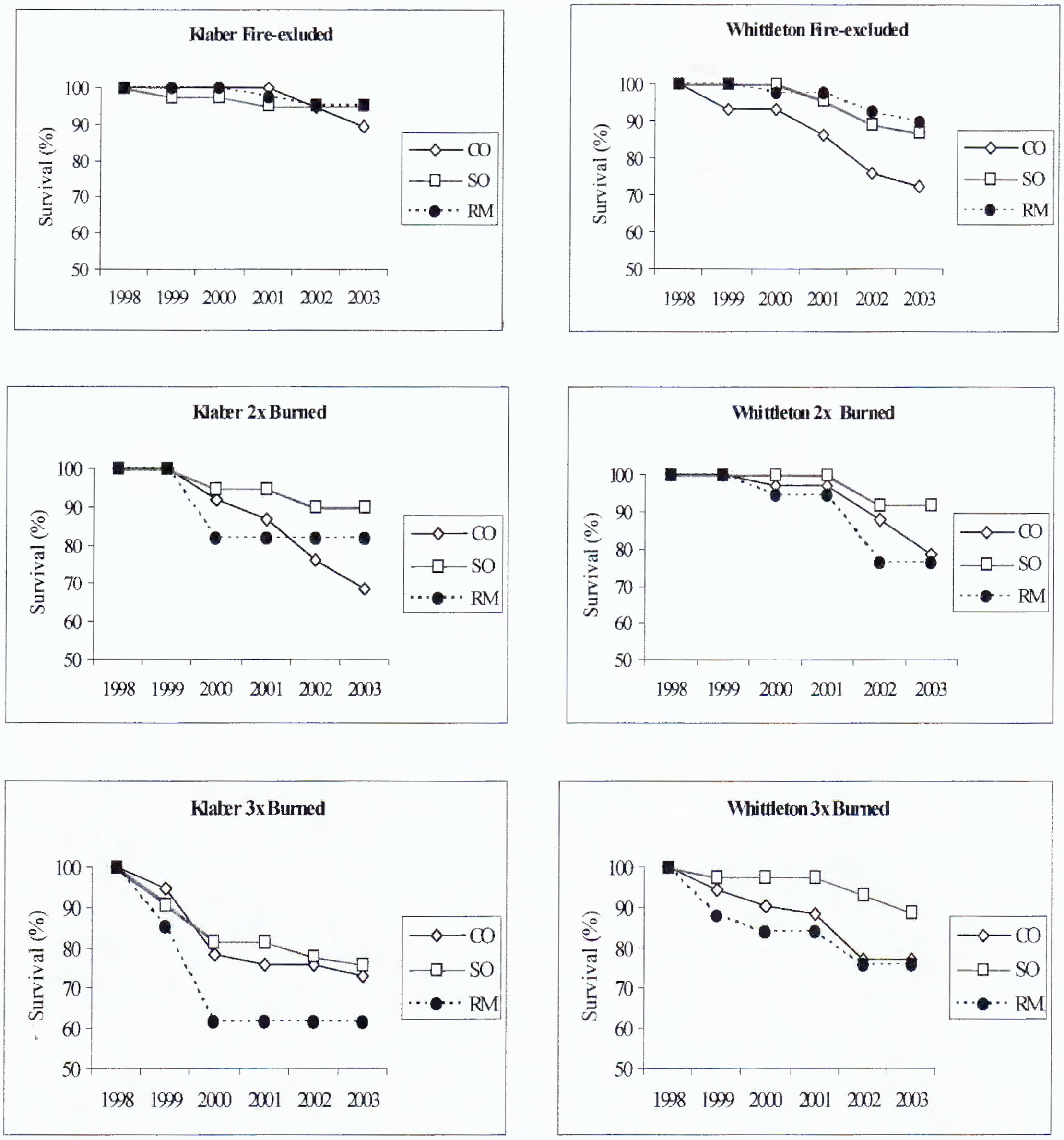
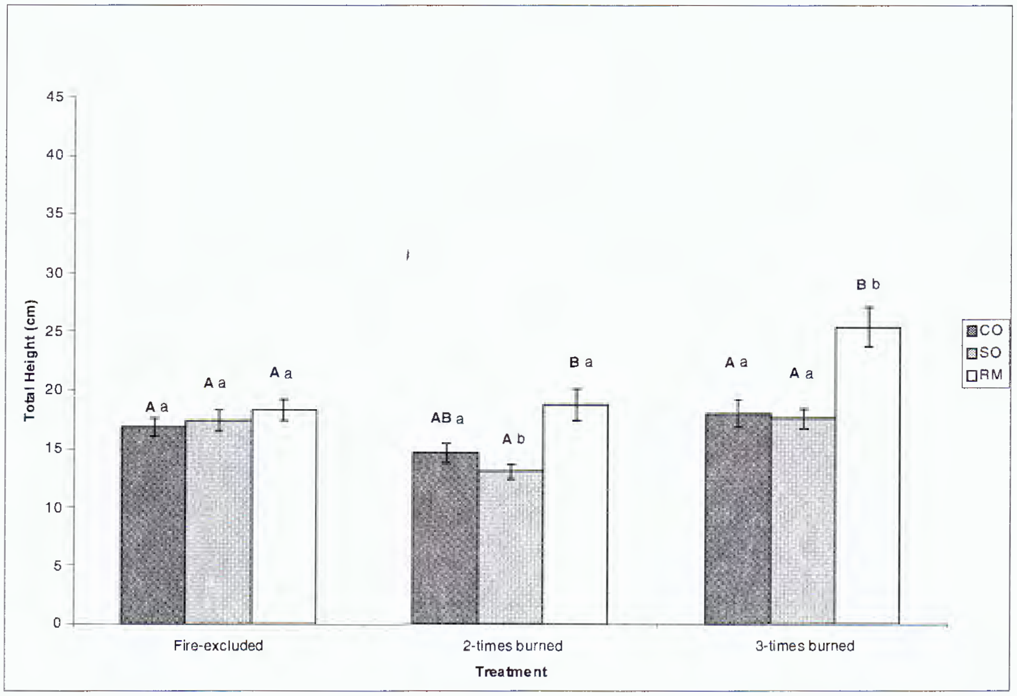


Figure 3: Seedling height (a) and average annual growth (b), two growing seasons after the last prescribed fire treatment (n=600). Different uppercase letters denote significant differences among species within treatment sites. Different lowercase letters denote significant differences within species among treatment sites.

a.



b.

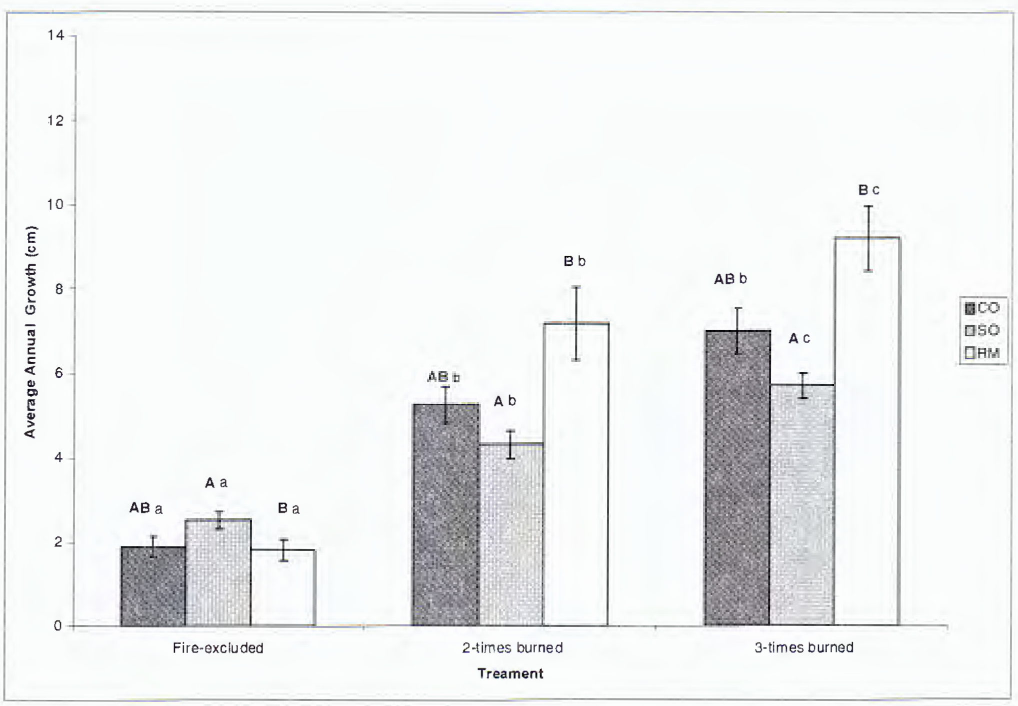
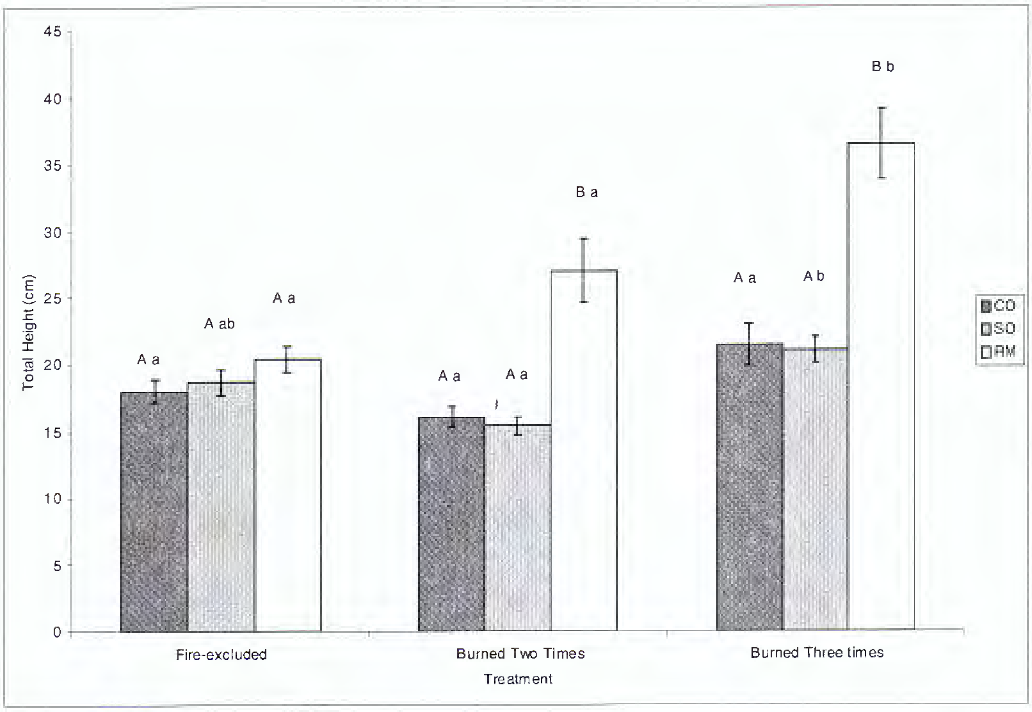


Figure 4: Total height (a) and average annual growth (b), after five years (2003, n=600). Different uppercase letters denote significant differences among species within treatment sites. Different lowercase letters denote significant differences within species among treatment sites.

a.



b.

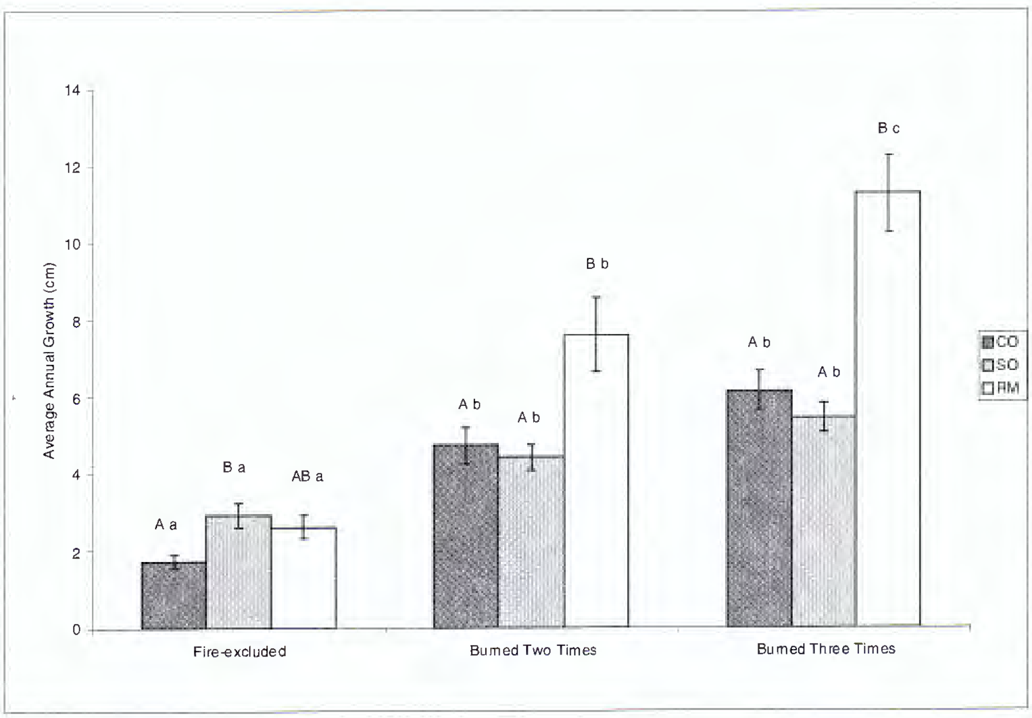
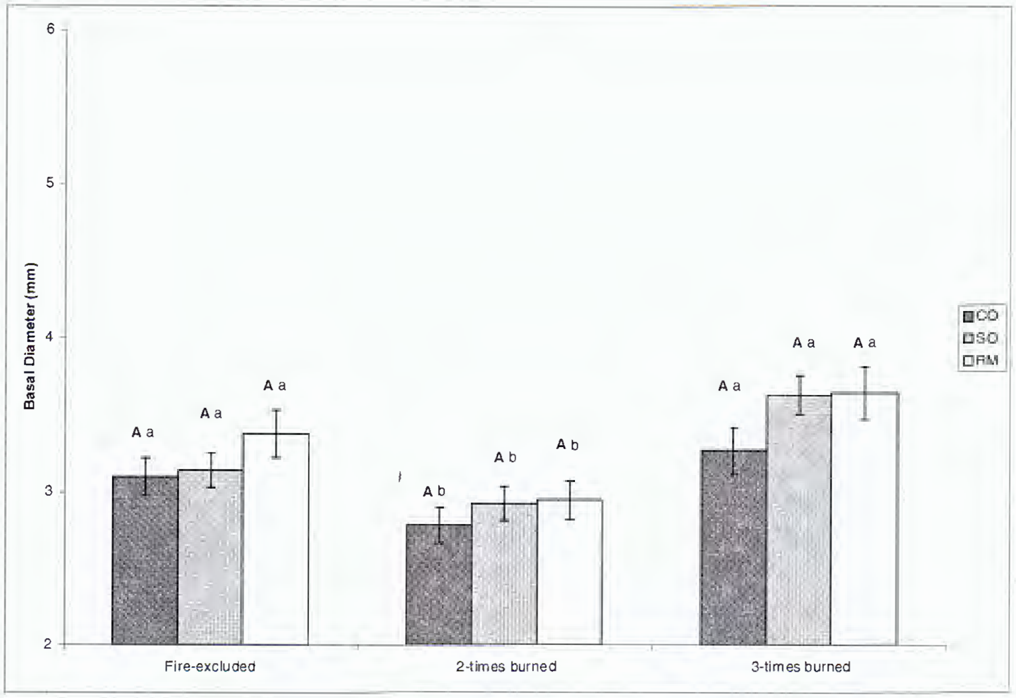


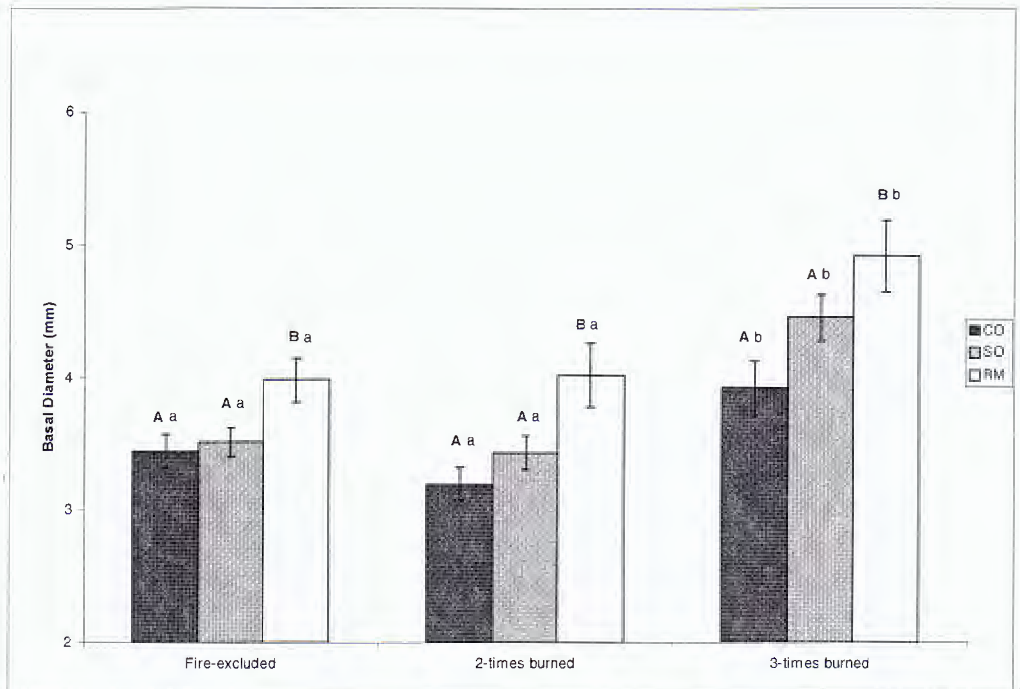
Figure 5: Seedling basal diameter two growing seasons after last prescribed fire treatment (a) and seedling basal diameter after five years (b, n=600). Different uppercase letters denote significant differences among species within treatment sites. Different lowercase letters denote significant differences within species among treatment sites.

a.



Basal Diameter (mm)
A a
A a
A a
A b
A b
A b
A a
A a
A a
CO
SO
RM
Fire-excluded
2-times burned
3-times burned

b.



Basal Diameter (mm)
A a
A a
B a
A a
A a
B a
A b
A b
B b
CO
SO
RM
Fire-excluded
2-times burned
3-times burned

CHAPTER

THREE

The Effects of Prescribed Fire on Stand Structure, Canopy Cover, and Seedling Survival in an Appalachian Hardwood Forest on the Cumberland Plateau.

1. Introduction

Research has shown that decreases in tall understory and midstory vegetation may help increase the vigor and overall competitiveness of oak seedlings (Lorimer et al. 1994). While management of midstory and understory stems may require cutting and herbicide treatment (Lorimer 1985; Van Lear and Watt 1993; Lorimer et al. 1994), fire may offer an additional tool for preferentially reducing oak competitor species, increasing available light through reductions in stem density and canopy cover, and thereby enhancing advanced oak regeneration. This approach is appealing to forest managers because of its cost effectiveness; larger acreage can be treated with relatively lower costs. The research described here was initiated as a long-term study to examine the effects of frequent and infrequent prescribed fire on stand structure and oak regeneration. In this study, I describe the short-term effects of one and two prescribed fires on stand structure, light environment, and seedling survival in an oak dominated forest on the Cumberland Plateau.

2. Site description

This study was conducted on the Morehead District of the Daniel Boone National Forest. The area is included in Braun's (Braun 1950) Cliff Section of the Cumberland Plateau of eastern Kentucky. Stands are dominated by oak (*Quercus* spp.) and hickory (*Carya* spp.) in the overstory (stems≥ 20 cm DBH; >70% relative density). The understory (stems 2-10 cm DBH) is dominated by maples (*Acer* spp., >45% relative density), downy serviceberry (*Amelanchior arborea* (Michx. f.) Fern), black gum (*Nyssa sylvatica* Marshall), and sourwood (*Oxydendron arboreum* (L.) DC.) (Figure 1). Oak species constitute an extremely small portion of the understory stratum (< 3%). The climate is humid, temperate, and continental with warm to hot summers and mild winters (Hill 1976). Mean annual temperature is 12° C, with mean daily temperatures ranging from 31° C in July to 0° C in January. Mean annual precipitation is 130 cm, with up to 15 cm in the form of snow. The mean growing season is 176 days (Foster and Conner 2001). The topography is highly dissected, consisting of ridges and mesic coves, ranging in elevation from 240 to 390 m and with slopes commonly exceeding 60%. The unglaciated, well drained loamy soils are classified as Typic Hapludalfs of the Brookside, Cranston, and Donahue series and Typic Dystrochrepts of the Berks series (Avers et al. 1974).

3. Methods

3.1. Experimental design

Three study areas (197-300 ha), Buck Creek (Bath and Menifee County), Chestnut Cliffs (Menifee County), and Wolf Pen (Bath County) were chosen based on research requirements for experimental design and on pre-existing Forest Service plans for large-scale burning. Each area was divided into three treatments for use in a long-term study of the effects of prescribed fire on oak regeneration. Fire treatments were applied non-randomly to accommodate management constraints related to burning and included: 1) fire excluded, 2) to be burned frequently, and 3) to be burned less frequently. Boundaries were based on available fire breaks such as drainages, cliff lines, roads or trails. As of 2004. less frequent treatment sites had been burned once, while frequent treatment sites had been burned twice. For the purposes of this study treatment sites will be referred to as fire excluded, once burned (1x), and twice burned (2x).

A sampling grid overlain on a topographic map was used to establish eight to twelve permanent plots (10 m x 40 m) in each treatment area, resulting in a total of 93 plots. The long axis of each plot was aligned parallel to the slope. Plots represented a wide variety of slopes, aspects, and species assemblages. To account for these differences, plots were categorized into three landscape positions (sub-xeric, intermediate, and sub-mesic). These classifications were based on species composition, slope position, aspect, and hill-shading (Table 1). No fire disturbance has been recorded on these sites in the past 30 years (Michael Colgon, U.S. Forest Service, Morehead, Ky., pers. com.); information on fire occurrence predating this period are not available.

3.2 Fire prescription

Prescribed fires were conducted by USDA Forest Service personnel in late winter and early spring of 2003 and 2004. Due to the size and topography of each study area, treatments were applied on different dates and ignited either by hand (drip torch), helicopter, or a combination of both. Dates, descriptions of the burn conditions, and source of ignition are described in Table 2.

3.3 Stand structure

The effects of prescribed fire on stand structure were measured through changes in stem density (stems/ha), basal area (m^2/ ha), and basal sprout density (sprouts/ha). Stand density and basal area were divided into three main size classes: understory (stems 2-10 cm DBH), midstory (stems 10-20 cm DBH), and overstory (stems $\geq$ 20 cm DBH). Prior to treatment, overstory and midstory stems ($\geq$ 10 cm DBH) located within each established plot (400 m^2) were identified to species, tagged with a unique number and mapped according to their distance and azimuth from the lower left corner of the plot. Understory stems were tagged and identified in the same manner, but due to the abundance of stems in this size class, only those understory stems located within the first quadrant (100 m^2) of each plot were censused. Annual measurements included DBH, status (dead or alive), and number of basal sprouts on each stem. Since understory stems were only measured on the first quadrant of each plot, basal sprout density was calculated as the sum of the sprout density of stems 2-10 cm DBH on the first 100 m^2 of each plot and the sprout density of stems $\geq$ 10.0 cm DBH on the entire plot (400 m^2).

3.4 Canopy cover

Canopy cover is described as the proportion of an area covered by the vertical component of vegetation (Vora 1988; Barbour et al. 1999). Thus, changes in canopy cover can be used to describe changes in the forest light environment. In order to characterize the effects of prescribed fire on understory light conditions, canopy cover (%) was measured annually with hemispherical photography, and starting in 2003, a spherical densiometer. Sampling transects were established on each plot, two meters up and parallel to the bottom plot boundary. Measurements were taken at three points along the long axis of the plots, at 10, 20, and 30 meters.

Hemispherical photographs must be taken either before sunrise, after sunset, or on a uniformly overcast day to eliminate direct sunlight which can cause uneven exposure and difficulties with analysis (Rich et al. 1999). Due to these technical constraints and the scale of this study, it was not feasible to take hemispherical on all three study areas. The Buck Creek (n= 33) area was chosen based on its accessibility. A spherical densiometer does not have the same technical restraints as hemispherical photography, and so measurements of canopy cover using this technique were taken on all three study areas.

3.4.1. Hemispherical photography

Hemispherical photographs were taken 80 cm above the forest floor using a Nikon CoolPix 950 digital camera during the 2002 and 2003 field seasons and a Nikon Coolpix 4500 digital camera during the 2004 field season. Each camera was fitted with a Nikon FC-E8 183° fisheye converter. For each photograph, the camera was oriented toward North and leveled using a Delta-T Devices self-leveling mount.

All images were analyzed using HemiView 2.1. Prior to analysis, images were edited using Adobe Photoshop® to remove lighted direction markers and to help clarify contrasts between open sky and canopy (Rich et al. 1999). To prevent systematic error, images were randomly sorted before threshold establishment. Threshold establishment was conducted in this manner three times, and the average threshold value for each image was used in the analysis. Threshold level is defined as an intensity for which an individual pixel will become black (blocked sky) or white (open sky). The intensities range from 0-255 (Frazer et al. 1999). Depending on the light levels and exposure, threshold values can vary from image to image, and therefore must be established separately for each image. Although there are inherent biases in this subjective method of classification, a study conducted by Robison and McCarthy (1999) showed no significant difference in estimated light values obtained by ten inexperienced volunteers, after proper training was provided. In this study all hemispherical photographs were analyzed by the same person.

3.4.2. Spherical densiometer

A convex spherical densiometer was used. The densiometer was held 80 cm off the forest floor, far enough away from the body so that the observer's head did not obstruct the view. Measurements were taken in four cardinal directions, averaged and then divided by 96. Canopy cover was the inverse of this number (Englud et al. 2000). Vales and Bunnell (1988) found significant differences between measurements taken by different observers. To minimize this source of error, all measurements were taken by the same person.

3.5. Seedling survival

To test seedling response to prescribed fire treatments, a long-term seedling population study was established in June of 2002. Seedlings were marked with a small (<5 cm radius) aluminum wire ring and an aluminum tag engraved with a unique identifying number. Where possible, ten seedlings each of the two dominant oak species (*Quercus* spp.) and ten seedlings of the most abundant competitor species on each plot were identified. Oak species included white oak (*Q. alba* L.), northern red oak (*Q. rubra* L.), black oak (*Q. velutina* Lam.), chestnut oak (*Q. prinus* L.), scarlet oak (*Q. coccinea* Muenchh.), and chinkapin oak (*Q. muhlenbergii* Engelm.). Competitor species included red maple (*Acer rubrum* L.), sugar maple (*A. saccharum* Marshall), sassafras (*Sassafras albidum* (Nutt.) Nees.), white ash (*Fraxinus americana* L.), hickory (*Carya* spp.), and eastern redbud (*Cercis canadensis* L.). The landscape scale of this study produced a large amount of variation in species assemblages, and so seedling species were not equally represented across treatments sites. To correct for this variation, seedling species were grouped by genera and ecophysiological traits (Table 3). Oak species were grouped by subgenera into *Erythrobalanus* spp. (red oaks) and *Leucobalanus* spp. (white oaks). Sugar and red maple seedlings were grouped together because they filled a similar ecological niche on mutually exclusive landscape positions. Sassafras was not grouped with any other species due to its abundance on all treatments and sites. White ash, eastern redbud, and hickory seedlings were grouped together, but were not included in analysis because they were not present on all treatments and sites. A total of 2,749 seedlings were tagged.

Marked seedlings were classified as "dead" or "alive" in each year. Seedlings were classified as dead if no stem was visibly present, or if the remaining stem showed no visible signs of life. A few seedlings resprouted after being classified as "dead." This problem has been cited in similar seedling population studies, especially with oak species (Jones and Sharitz 1998; Marks and Gardescu 1998). Seedlings that resprouted after they were classified as "dead" were reclassified as "alive" and maintained in the population study. Due to thick understory vegetation cover and the propensity for top kill after fire, some seedlings were not located during annual censusing. These seedlings were classified as "lost." Seedlings that were classified as "lost" and then found alive the following year were reclassified as "alive." Those not found again were eliminated for all years of the study, to avoid confusion between seedlings that were known to have died, and those whose status was unclear.

4. Data analysis

4.1. Stand structure

Although vegetation was censused annually, this study analyzed the effects of prescribed fire on stand structure by comparing pre-treatment (2002) and post-treatment (2004) measurements. Statistical analyses were based on 93 plots. Stem density and basal area for each plot were divided into understory, midstory, and overstory size classes and analyzed by a repeated measures, split-plot analysis using PROC MIXED in SAS (SAS Institute 2000), with treatment blocked within site. The effects tested were: year (2002 or 2004), treatment (fire-excluded, 1x, 2x), landscape position (sub-xeric, intermediate, and sub-mesic), and their respective interactions. The Shapiro-Wilk test was used to test for Normality, and sprout densities were found to be non-normally distributed, and so were transformed using the natural logarithm. When retested, the log transformed data met normal distribution assumptions, and were analyzed using the model described above. All post-hoc pairwise comparisons were made using the least squares difference (LSD) test (SAS Institute 2000). P-values of <0.05 were considered statistically significant.

4.2. Canopy cover

Statistical analysis of canopy cover measured by a spherical densiometer was based on 91 plots. In 2004, canopy cover was not estimated on two plots in the Wolf Pen and Chestnut Cliffs areas. These plots were therefore removed from analyses. Statistical analysis of canopy cover measured by hemispherical photography was based on 33 plots. Since hemispherical photographs were taken only on the Buck Creek area, prescribed fire treatments were not replicated, thus individual plots were treated as replicates, resulting in a pseudoreplicated design. The Shapiro-Wilk test was used to test for Normality, and mean canopy cover estimates for each type of measurement were found to be non-normally distributed, and so were arcsine transformed (Sokal and Rohlf 1981). When retested, the transformed data met normal distribution assumptions and were analyzed by repeated measures analysis using PROC MIXED in SAS (SAS Institute 2000), as described above.

Relationships between changes in canopy cover measured by hemispherical photographs and stand structure were tested by correlation analyses. Correlations were conducted to determine if there were relationships between canopy cover and: 1) stem density (understory, midstory, and overstory); 2) basal area (understory, midstory, and overstory); and 3) sprout density. All correlations were run using JMP IN® statistical software (SAS Institute 2005). Relationships between canopy cover measured by a spherical densiometer and stand structure were not tested because preliminary results indicated no effect of prescribed fire treatment on spherical densiometer canopy cover estimates, and that this measurement technique was subject to considerable user error.

4.3. Seedling survival

Seedling survival was analyzed after two years of treatments (2004). As previously mentioned some seedlings (105 total) were lost throughout the course of this study and were not included in analysis of survival. Although white ash, eastern redbud, and hickory species were grouped (174 total), they were not represented across all treatments, and therefore were not included in the analysis. A total of 2,470 seedlings in four groups (red oaks, white oaks, maples, and sassafras) were used in the analysis of survival which was based on n=3 sites per treatment.

Survival percentages for seedling species groups on each treatment were averaged and then arcsine transformed (Sokal and Rohlf 1981; Huddle and Pallardy 1999). Data were analyzed by a split-plot fixed effects model using PROC MIXED (SAS Institute 2000), with treatment blocked within study area. Effects tested were species group, treatment and their respective interactions. All pairwise comparisons were made using the least squares difference (LSD) test. P-values of <0.05 were considered statistically significant.

5. Results

5.1 Stand structure

5.1.1. Stem density

Total stem density was reduced by fire, but repeated prescribed fire did not have an additional effect on stem density (Figure 2). Sub-mesic sites had higher total stem density (1,368 stems/ha) than both intermediate (1,091 stems/ha, $p=0.0319$) and sub-xeric (904 stems/ha, $p=0.0026$) sites. Total stem densities on landscape positions differed regardless of treatment or year.

Stem density in the understory and midstory were reduced by fire, but the largest reduction in density occurred in understory stems (Table 4); there was no effect of fire on stem density in the overstory stratum. After two years, understory stem density had decreased on both 1x ($p<0.0001$) and 2x ($p<0.0001$) burned sites, but the burned sites did not differ from each other ($p=0.90$) (Figure 2). Understory stem density differed across landscape positions. Sub-mesic plots had higher understory stem density (956 stems/ha) than both intermediate (663 stems/ha, $p=0.0235$) and sub-xeric (472 stems/ha, $p=0.0013$) plots, regardless of treatment or year. There was a significant interaction between year, treatment, and landscape position ($p=0.0142$) on midstory densities. Midstory densities on 1x, sub-xeric sites and 2x, sub-xeric sites were significantly reduced from pre-treatment levels ($p<0.0001$, both). Twice burned, sub-xeric sites had a lower midstory stem density than 2x, sub-mesic sites ($p=0.01$). Midstory stem density on fire-excluded sites remained stable (Table 3).

5.1.2. Basal area

Total basal area and its largest component, overstory basal area, were not affected by fire.

The understory basal area, though a small component of total stand basal area, was reduced on burned sites ($p<0.0001$). In 2004, both 1x and 2x burned sites had lower understory basal areas than fire-excluded sites ($p=0.0367$ and $p=0.0476$, respectively); burned sites did not differ from each other. Sub-mesic plots had higher understory basal area (1.99 m^2/ha) than sub-xeric (1.09 m^2/ha, $p=0.0013$) plots, regardless of treatment or year (Table 3).

As with midstory density, there was an interaction between treatment and landscape position on midstory basal area. Midstory basal area on 1x, sub-xeric and 2x, sub-xeric sites was significantly reduced from pre-treatment levels ($p<0.001$ and $p<0.0001$, respectively). Midstory basal area was lower on 2x, sub-xeric sites than 2x, sub-mesic sites ($p=0.002$, Table 3). Midstory basal area on fire-excluded sites remained stable (Table 3).

5.1.3. Sprouting

There was a significant interaction ($p=0.04$) between year, treatment, and landscape position on basal sprout density. For both burn treatments, sprout densities increased significantly across all landscape positions. On fire-excluded treatments, sprout densities remained stable on sub-xeric and intermediate sites, but increased ($p<0.0001$) on sub-mesic sites (Figure 3).

5.2. Canopy cover

When measured with a spherical densiometer, canopy cover was higher in 2003 than 2004 ($p=0.0121$), regardless of treatment. Also, canopy cover was higher on sub-mesic sites, as compared to both sub-xeric and intermediate sites ($p\leq 0.001$).

When canopy cover was measured with hemispherical photography (Buck Creek area only), there were significant interactions between year and treatment ($p=0.0041$). Canopy cover was similar across all treatment sites prior to treatment ($p>0.10$, Figure 4). In 2003, after only one prescribed fire on both the 1x burned sites and 2x burned sites, canopy cover was reduced on burned sites, but only the reduction on the 2x burned site was significant ($p<0.0001$; 1x burned sites: $p=0.0971$). Also, canopy cover on the fire-excluded site was higher than on the 1x burned site ($p=0.1711$) and significantly higher than on the 2x burned site ($p=0.0007$). Canopy cover on the 1x burned site was higher than canopy cover on the designated 2x burned site ($p=0.0156$; Figure 4). In 2004, after the application of a second fire to the 2x burned site, canopy cover on the fire-excluded sites was higher than on the 1x burned sites ($p=0.4386$) and 2x burned sites ($p= 0.0023$); and canopy cover on the 1x burned sites was higher than canopy cover on the 2x burned sites (p=0.0120; Figure 4). Canopy cover on the fire-excluded sites increased with time, resulting in significantly higher (p=0.0102) canopy cover after two years. In contrast, canopy cover on 1x and 2x burn sites decreased in 2003 (p=0.0971 and p<0.0001, respectively), and then increased in 2004 (p=0.0187 and p<0.0008, respectively; Figure 4).

When canopy cover was measured by hemispherical photography an interaction between year and landscape position (p=0.0206) was also detected. In 2002 and 2003, canopy cover on sub-mesic plots was higher than intermediate (p=0.0026 and p=0.0401, respectively) and sub-xeric plots (p=0.0007 and p=0.0006, respectively; Figure 5). In 2003, intermediate plots also had higher (p=0.0327) canopy cover than sub-xeric plots. In 2004, sub-mesic (p=0.0001) and intermediate (p=0.0037) plots had higher canopy cover than sub-xeric plots (Figure 5). Although not statistically significant, canopy cover on sub-mesic plots increased slightly over three years of measurement. In contrast, canopy cover on intermediate and sub-xeric plots decreased in 2003 (p=0.0002 and p=0.0521, respectively) and then increased in 2004 (p=0.0025 and p=0.0027, respectively; Figure 5).

All components of stem density were significantly correlated with measures of canopy cover by hemispherical photography. The strongest correlation was between understory stem density and canopy cover (r=.4451, p<0.0001), whereas midstory stem density was weakly correlated with canopy cover (r=.2272, p=.0237). There was a very weak, negative correlation between overstory density and canopy cover (r= -.1980, p=0.0494). Understory (r=.4353, p <0.0001) and midstory (r=.2028, p=0.0441) basal areas were also significantly correlated with measures of canopy cover by hemispherical photography. Sprout density was negatively correlated with canopy cover (r= -.3312 p=0.0017).

5.3. Seedling survival

In the absence of fire, all species groups had similar survival rates. Maple and white oak group seedling survival rates were reduced by fire, but repeated prescribed fires did not have a greater effect on their survival. Maple seedlings on 2x burned sites survived at lower rates than those on 1x burned and fire-excluded sites (p=0.0428 and p<0.0001, respectively), but survival rates on burned sites did not differ from each other. Although not significantly different from each other, survival rates of white oaks on 1x burned (p=0.0169) and 2x burned sites (p=0.0240) were lower than fire-excluded sites. Sassafras and red oak seedlings did not have significantly different survival rates among treatments. Although not always significant, survival rates across species followed the same pattern on less frequent and frequent sites (maples < white oaks < red oaks < sassafras; Figure 6).

6. Discussion

6.1. Effects of prescribed fire on stand structure and canopy cover

Prescribed fire affected stand structure in the oak-hickory dominated forests of this study, as shown by reductions in stem density, basal area, and increases in basal sprout densities on burned sites. However, two successive prescribed fires did not produce greater effects on stand structure compared to a single fire. A comparable study conducted on ridgetop forests in eastern Kentucky found similar reductions in stem density and basal area after fire, but in that study, each fire had an increased effect on stem mortality (Blankenship and Arthur in preparation). In both studies, the reductions in density and basal area were attributed mainly to a loss of understory stems.

Although a decrease in density and basal area of large stems was not expected in this study, landscape position negatively influenced midstory density and basal area in the presence of prescribed fire. Higher fire intensities have the potential to result in increased mortality among larger stems. Studies such as the one done by McCarthy and Sims (1935) have shown that there is a negative correlation between stem size and mortality due to fire, attributed to increasing bark thickness with growth (Brown and Davis 1973). When exposed to low-intensity surface fires, large diameter trees have lower mortality than smaller stems due to their thicker barks (Harmon 1984; Van Lear and Waldrop 1989). Using the fire temperature data collected on this study as an estimate of fire intensity could help explain why reductions in midstory density and basal area were seen only on sub-xeric, burned sites. Average fire temperatures at 20cm above the forest floor were higher on sub-xeric sites than both sub-mesic and intermediate sites (Loucks 2004). Thus, hotter fires on sub-xeric sites resulted in a significant decrease in midstory stem density. It appears that fire intensities on sub-mesic sites were not high enough to result in a reduction of midstory stems.

Sprouting is a trait that helps ensure persistence and can affect the community composition and structure after disturbance (Bond and Midgley 2001). In this study, sprout density increased on both once burned and twice burned sites after fire treatment. Increased stump sprouting after fire by wounded stems is a common occurrence in hardwood forests (Van Lear and Watt 1993; Arthur et al. 1998; Kuddes-Fischer and Arthur 2002; Blankenship and Arthur in preparation). Although it is generally thought that increased fire frequency results in decreased sprout densities (Van Lear and Waldrop 1989; Van Lear and Watt 1993), studies in this region have shown mixed results. In this study, sprout densities on 1x and 2x burned sites were not significantly different from each other. In contrast, similar studies on the effects of prescribed fire showed that sprout densities increased with repeated fires (Arthur et al. 1998; Blankenship and Arthur in preparation).

The initial design of this study proposed characterizing the effects of prescribed fire on canopy cover across all three study areas using only hemispherical photography, but for the reasons previously outlined, this was not feasible. To compensate for the missing canopy cover data, a spherical densiometer was used to estimate canopy cover on all areas starting in 2003. The spherical densiometer is a relatively inexpensive and simple method of estimating canopy cover. Studies that compared these two methods of estimating canopy cover (Comeau et al. 1998; Englud et al. 2000) have shown that they provide reliable and comparable estimates.

Measurements of cover using a spherical densiometer require the user to estimate the fraction (in fourths) of squares on a mirror that are not blocked by vegetation. Although a relatively easy procedure, there is a high amount of subjectivity involved in the estimation of canopy cover by a spherical densiometer (Comeau et al. 1998). Englud et al. (2000) found that observer consistency improved with practice, and that inexperienced users initially overestimate canopy cover. Canopy cover, as measured by the spherical densiometer, was higher in 2003 than 2004. My inexperience with the use of a spherical densiometer at the time of initial measurements (2003), could account for the differences in estimates between years. Although no effects of prescribed fire treatment on canopy cover were found, there were differences in canopy cover across landscape positions. These differences in canopy cover mirrored the differences observed in understory stem density; sub-mesic plots had higher canopy cover and understory stem densities than both sub-xeric and intermediate plots.

Hemispherical photography also detected significant effects of landscape position on canopy cover across the years. However, since yearly differences were based on canopy cover averaged over burned and unburned sites, and treatments were added yearly (1x-2003, 2x-2004), it is difficult to discern this early in the study, which changes in canopy were only due to differences within landscape positions.

Canopy cover measured with hemispherical photographs was reduced by prescribed fire, at least initially. In the absence of fire, canopy cover increased over the three years, gaining 2.4 percent. On the burned sites, the trend was for canopy cover to decrease in 2003 (-6.4%) and then increase (+5.4%) to pre-treatment levels in 2004. This temporal loss in canopy cover is comparable to the results of a similar study (Chiang 2002), where canopy cover increased continuously over five growing seasons following an initial decrease after prescribed fire. It is important to note that in 2004, only one growing season had passed since the twice burned sites on this study had been burned, yet they still showed an increase in canopy cover. Also, temperatures on the 2004 Buck Creek burn were lower than temperatures on the 2003 Buck Creek burn (Table 2) and the 2004 burn was discontinuous in nature (Lyons, per. communication). It is possible that the non-continuous, low intensity nature of this fire had limited additionally stress on stems, allowing for the quick recovery of canopy cover.

There are several other possible explanations for this rapid canopy closure. The 2004 growing season was very wet, thus promoting greater leaf areas and therefore higher canopy cover. Another possible explanation is that individual trees responded to stress caused by prescribed fire by producing epicormic branches. Although not quantified, this "bushy" growth was observed on many trees, especially sourwood and black gum. Finally, the considerable increase in sprout densities recorded on burned sites could account for the rapid increase in canopy cover, even after an additional fire. Not only did the number of sprouts increase on burned sites, they also grew to heights greater than 80 cm, as evidenced by their appearance in the hemispherical photographs.

While canopy cover measured by hemispherical photographs was correlated with all components of stem density, the strongest relationship was between understory stem density and canopy cover. As understory stem density decreased, canopy cover also decreased. Chiang (2002) found the same relationship between understory stem density and canopy cover one year after prescribed fire treatment in an oak-dominated stand. This relationship suggests that the removal of understory stems can result in an increase in the amount of light reaching the forest floor.

The most surprising relationship was that of sprout densities and canopy cover. Chiang (2002) found that high basal sprout densities after fire effectively canceled out the effects of decreased stem density on canopy cover; meaning that damaged stems sprouted prolifically and the increase in sprout density resulted in an increase in total canopy cover at the seedling level. As discussed previously, increased sprout densities is one explanation for the increase in canopy cover seen in this study; however, sprout density was found to be negatively correlated with canopy cover. This unexpected negative relationship would suggest that increased basal sprout density was not responsible for the increase in canopy cover seen in 2004. But when this seemingly contradictory relationship was examined further, it was determined that the negative relationship was being driven by the 2003 data. There were plots on the burned areas that although they had high sprout densities, had low canopy cover due to high stem mortality. The reductions in canopy cover on these plots, as much as -21% in some cases, was too great to be masked by high sprout densities, and so resulted in the negative relationship between canopy cover and sprout densities.

6.2. Effect of fire on seedling survival

Prescribed fire resulted in higher seedling mortality for maples and white oak, but not for sassafras or red oaks. There was a trend in survival among species groups (sassafras> red oaks > white oaks > maples). Overall, oak seedlings survived fire better than maples. While seedlings in the white oak group experienced significantly higher mortality on burned sites, successive fires did not negatively affect their survival. In contrast, maple survival rates decreased with each fire treatment. However, there was large variability in maple survival that can be explained by differences between sugar maple and red maple survival rates. Although not statistically analyzed due to lack of complete representation across treatments, red maple seedlings had higher rates of survival on burned sites compared to sugar maples (52% versus 3%). When red maple survival was compared to white (58%) and red oak group (78%) seedlings, the trend in survival among species mentioned earlier is still present, but not as large.

These results are comparable to other studies. Huddle and Pallardy (1999) found that survival of northern red oak (*Erythrobalanus*) was significantly higher than survival of both white oak (*Leucobalanus*) and red maple seedlings when exposed to fire. In a long-term seedling population study conducted on ridgetops in eastern Kentucky, scarlet oak (*Erythrobalanus*) and chestnut oak (*Leucobalanus*) seedling survival rates were significantly higher than red maple after five years of periodic prescribed fires (Green, this thesis), but in contrast to the preliminary results of this study, scarlet oak seedlings did not survive at significantly higher rates than chestnut oak seedlings. My results support the hypothesis that repeated prescribed fires can selectively reduce the abundance of competitor species such as maples in the seedling stratum.

Sassafras maintained very high survival rates across all treatments (> 86%), probably due to clonal sprouting response. Across all three sites in this study, prescribed fire prompted vigorous sprouting by sassafras and there were some areas in which sassafras formed a nearly continuous layer. The abundance of sassafras sprouts on these study sites was not expected in light of other studies that have been conducted in similar stands on the Daniel Boone National Forest, although Arthur et al. (1998) reported positive sprouting responses due to prescribed fire for sassafras. Jackson and Buckley (2004) reported significant increases in sassafras densities after prescribed fires in Tennessee and described sassafras as an important competitor for oak seedlings. While the duration of this competition is unknown, it is possible from the preliminary data of this study that sassafras has the potential to be a significant oak competitor, at least at the seedling level.

7. Conclusions

Significant reductions in density and basal area in response to prescribed fire occurred only in the understory size class and in midstory size classes on sub-xeric, burned sites. These reductions were sufficient enough to cause temporal decreases in canopy cover, resulting in a presumed increase in light. Seedling survival decreased with prescribed fire, with maple seedlings suffering from higher mortality than both oak seedling groups. The time scale for this study was too short to make many conclusive statements regarding the relationship between seedling responses and changes in light or stand structure, but it does provide a solid argument for the long-term monitoring of this seedling population, especially in regards to the unexpected response of sassafras seedlings.

Table 1: Total numbers of plots arranged by site, treatment, and landscape position.

Site	Sub-mesic	Intermediate	Sub-xeric	Total
Buck Creek				
Fire-excluded	3	4	3	10
Less Frequent	3	4	4	11
Frequent	2	7	3	12
Chestnut Cliffs				
Fire-excluded	3	6	2	11
Less Frequent	4	4	1	9
Frequent	3	4	3	10
Wolf Pen				
Fire-excluded	5	3	2	10
Less Frequent	1	6	5	12
Frequent	2	5	1	8
Total	26	43	24	93

Table 2: Dates, method of ignition (helicopter or hand), ambient conditions on the day of burn, and mean maximum temperatures surpassed at three heights above forest floor (0, 20, and 40 cm). Mean maximum temperatures across landscape position are also shown. The treatment sites are Buck Creek less frequent (BCLF), Buck Creek frequent (BCF), Chestnut Cliffs less frequent (CCLF), Chestnut Cliffs frequent (CCF), Wolf Pen less frequent (WPLF), and Wolf Pen frequent (WPF).

	BCLF	BCF		CCLF	CCF		WPLF	WPF
Date	4/14/2003	4/14/2003	3/26/2004	3/25/2003	3/24/2003	4/7/2004	4/16/2003	4/16/2003
Time of Ignition	1130	1130	1300	1130	1230	1200	1230	1230
Method of Ignition	Aerial	Aerial	Hand	Hand	Hand	Aerial	Aerial	Aerial
Air temperature °C[1]	26.3	26.3	25.2	25	24	23.4	26.8	26.8
RH (%)[1]	23.2	23.2	44.6	32.5	35	39.7	38.7	38.7
Wind Direction	NW	NW	SW	SW	W	W	W	W
Wind Speed (km/hr)[1]	0-9.7	0-9.7	3.2-9.7	0-9.7	0-9	3.2-6.4	0-14.48	0-14.48
Mean Temp @ 0 cm	**473.2**	**563.7**	**132.7**	**533.1**	**475.0**	**411.5**	**583.5**	**561.3**
Sub-xeric	554.6	621.9	28.3	549.8	554.1	486.2	574.1	617.0
Intermediate	536.9	553.9	215.3	522.4	493.5	366.5	594.7	560.1
Sub-mesic	306.8	510.7	NA[3]	541.9	371.2	396.7	563.8	536.4
Mean Temp @ 20 cm	**181.7**	**269.5**	**37.9**	**315.7**	**232.5**	**186.7**	**350.6**	**256.7**
Sub-xeric	240.1	325.0	4.4	536.8	281.5	255.5	294.8	332.0
Intermediate	179.3	278.4	63.1	346.0	221.2	166.4	411.0	252.5
Sub-mesic	126.4	155.5	NA[3]	201.6	198.6	145.0	267.0	229.6
Mean Temp @ 40 cm	**138.8**	**186.6**	**11**	**234.5**	**158.2**	**109.0**	**242.0**	**200.2**
Sub-xeric	189.2	269.1	0.0	415.7	194.2	175.8	208.0	255.5
Intermediate	133.6	174.3	18.8	253.5	167.0	67.0	279.3	201.7
Sub-mesic	95.3	105.7	NA[3]	148.9	110.6	98.3	188.2	168.7

[1] Calculated as the average through duration of fire.

[2] Fire on treatment site was originally started on 3/26/2004, but extinguished before burning any plots. Fire was relit 4/7/2004.

[3] BCF sub-mesic and WPF sub-xeric plots did not burn in 2004.Table 3: Seedling species and the number of plots on which they are represented. Numbers in parentheses equal the total number of seedlings. Species with asterisk were included in analyses of survival and growth parameters.

Oak Species	
*Erythrobalanus spp.	**89** *(958)*
*Leucobalanus spp.	**85** *(819)*
Competitor Species	
**Acer* spp.	**39** *(356)*
**Sassafras albidum*	**35** *(337)*
Fraxinus americana	**14** *(134)*
Carya spp.	**3** *(30)*
Cercis canadensis	**1** *(10)*

	Density (stems/ha)						Basal Area (m^2/ ha)				
	2-9.9 cm		10-19.9 cm		≥ 20.0 cm		2-9.9 cm		10-19.9 cm		≥ 20.0 cm
	2002	2004	2002	2004	2002	2004	2002	2004	2002	2004	2002
Fire-excluded (n=3)	**835**	**823**	**228**	**225**	**218**	**221**	**1.85**	**1.80**	**3.72**	**3.05**	**23.2**
Sub-xeric	671	729	243	225	214	214	1.47	1.62	4.06	3.52	17.5
Intermediate	823	792	248	246	235	244	2.03	1.89	4.02	3.43	22.2
Sub-mesic	955	918	195	200	200	198	1.88	1.82	3.16	2.31	27.9
Less Frequent (n=3)	**963**	**319**	**253**	**184**	**195**	**180**	**2.03**	**0.90**	**4.01**	**3.03**	**19.6**
Sub-xeric	770	90	283	138	210	170	1.74	0.38	4.48	2.53	17.2

Sub-mesic	943	614	264	254	204	186	1.86	1.56	4.12	4.36	23.1

Table 4: Mean densities and basal areas across treatment and landscape position. Three components of density and basal area are represented (understory, 2-9.9 cm DBH; midstory 10-19.9 cm DBH; and Overstory ≥ 20.0 cm DBH).

Figure 1: Relative pre-treatment species composition averaged over all plots (n=93) in the study area. Species composition is shown by size classes.

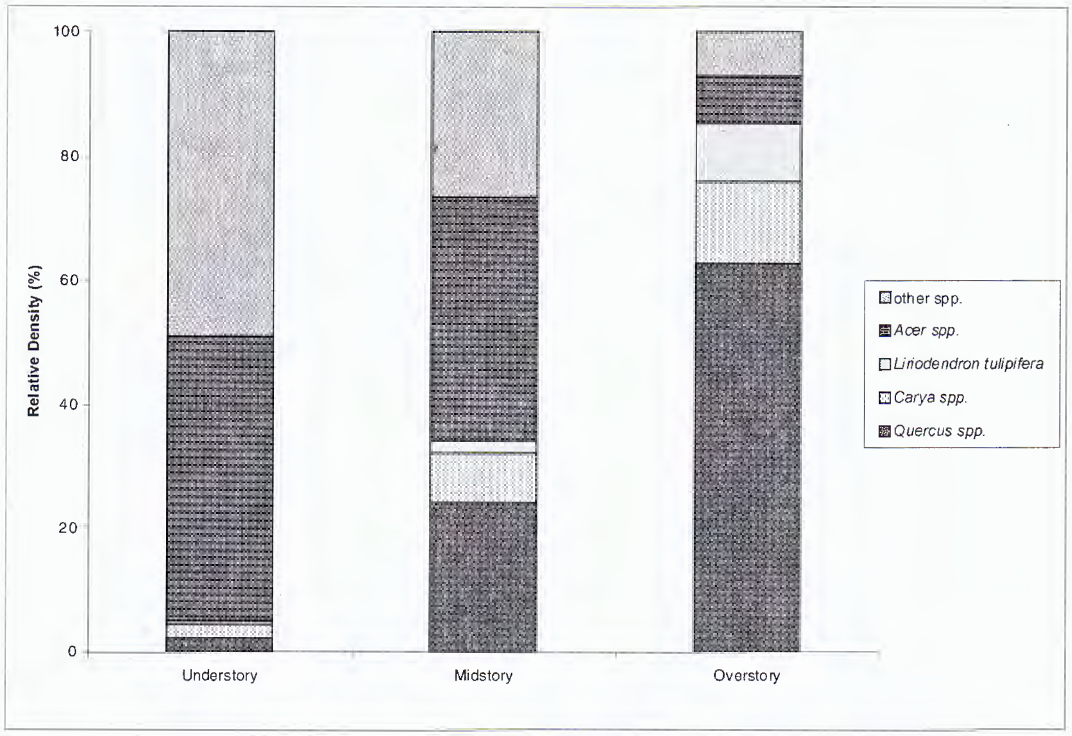


Figure 2: Components of total stem density on the three treatment sites: fire-excluded, less frequent, and frequent. Different lowercase letters denote significant differences at the treatment level within density components and across years. Different uppercase letters denote differences in total density across years within treatment sites.

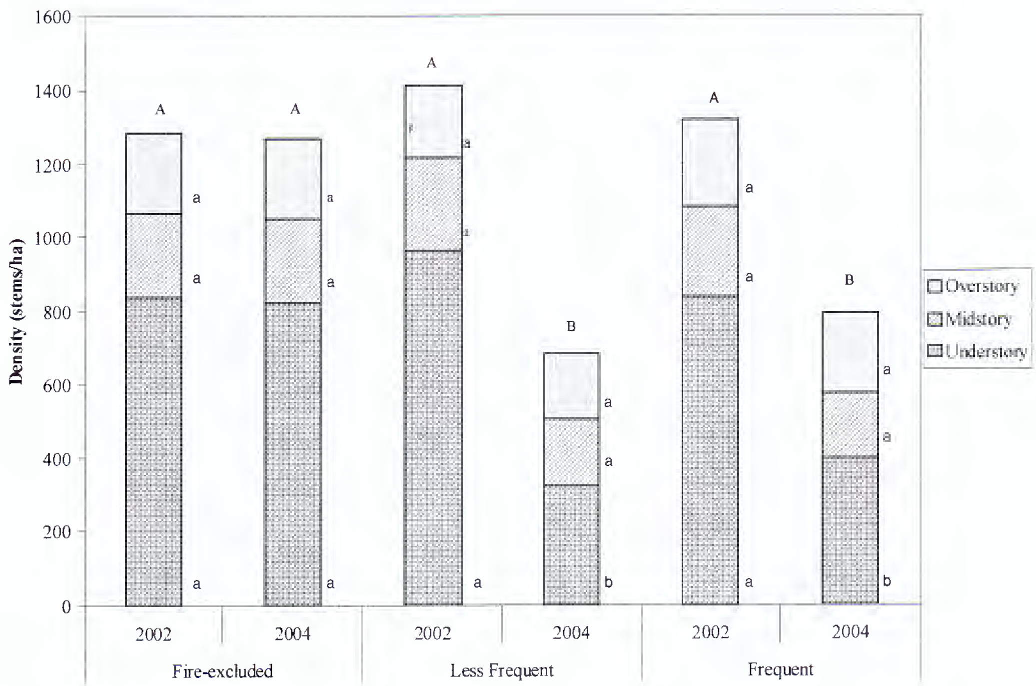


Figure 3: Basal sprout density (stems/ha) across treatments, years, and landscape position. Different letters denote significant differences across all treatments, landscape positions, and both years.

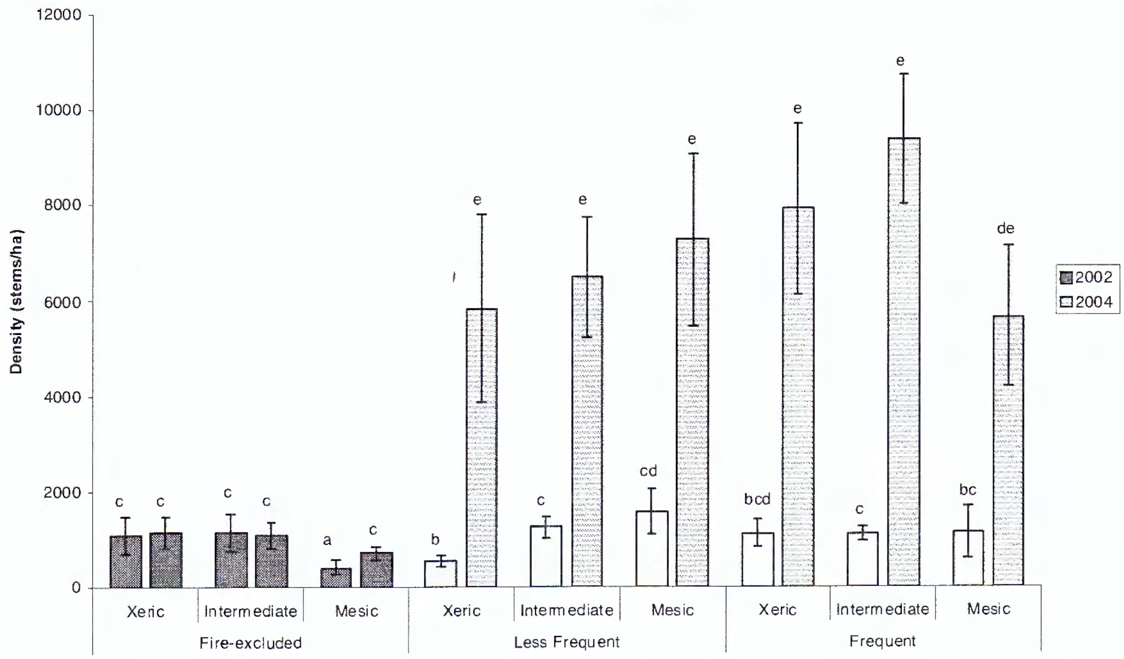


Figure 4: Canopy cover on the Buck Creek site, as measured by hemispherical photography. Canopy cover is shown across treatment and years. Different uppercase letters denote significant differences across treatments within years. Different lowercase letters denote significant differences within treatments and across years.

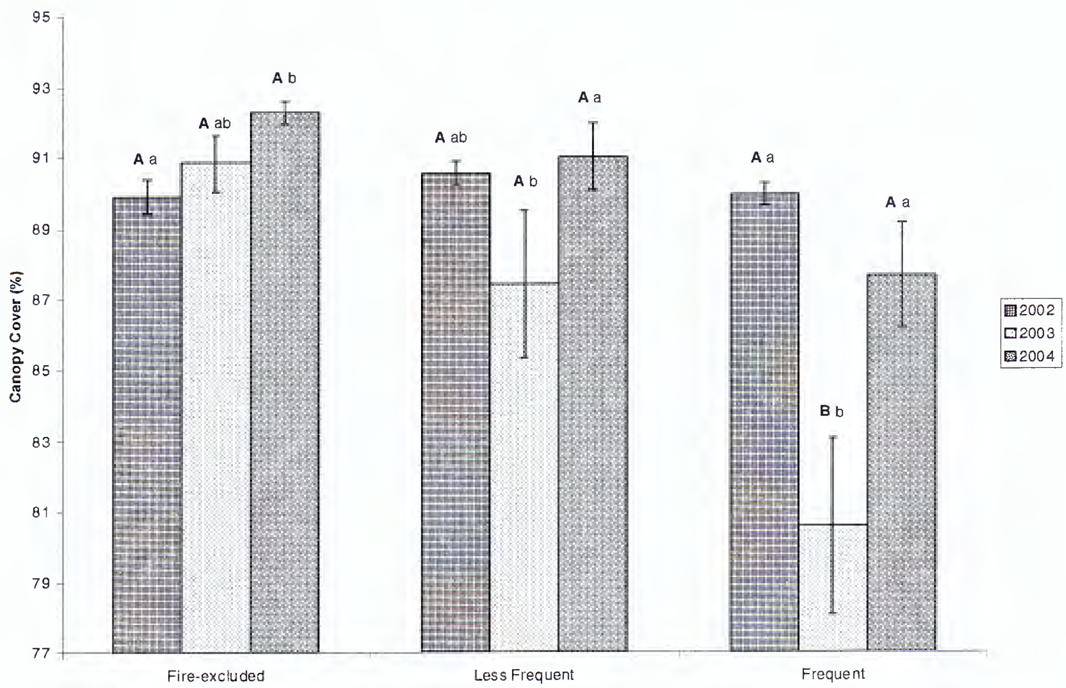


Figure 5: Canopy cover on the Buck Creek site, as measured by hemispherical photography. Canopy cover is shown across years and landscape positions. Different uppercase letters denote significant differences across landscape positions within years. Different lowercase letters denote significant differences within landscape positions across years.

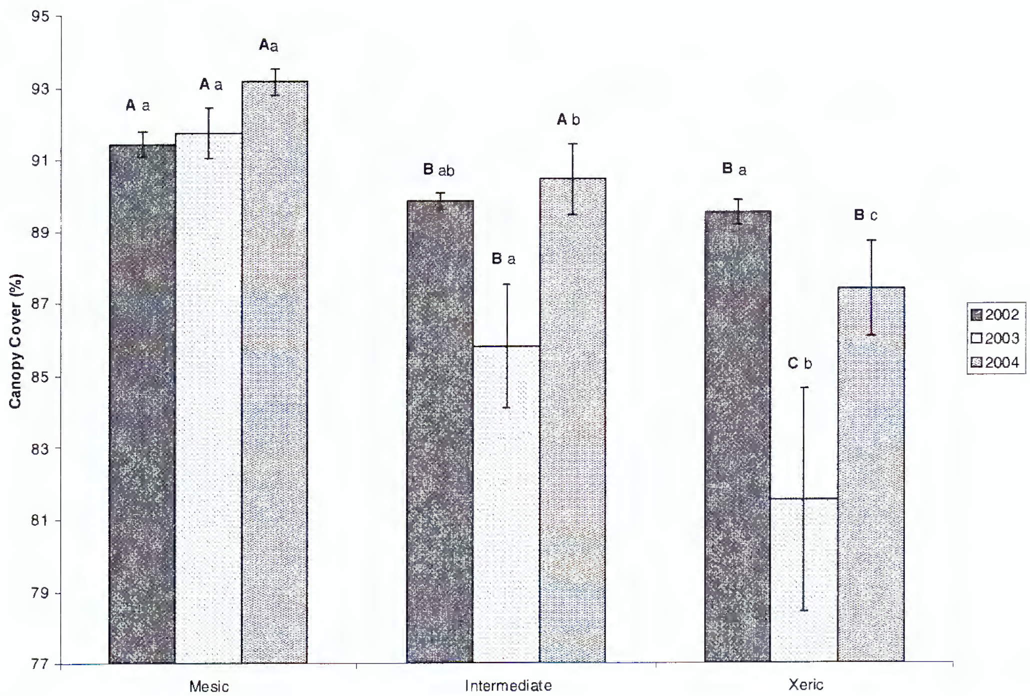
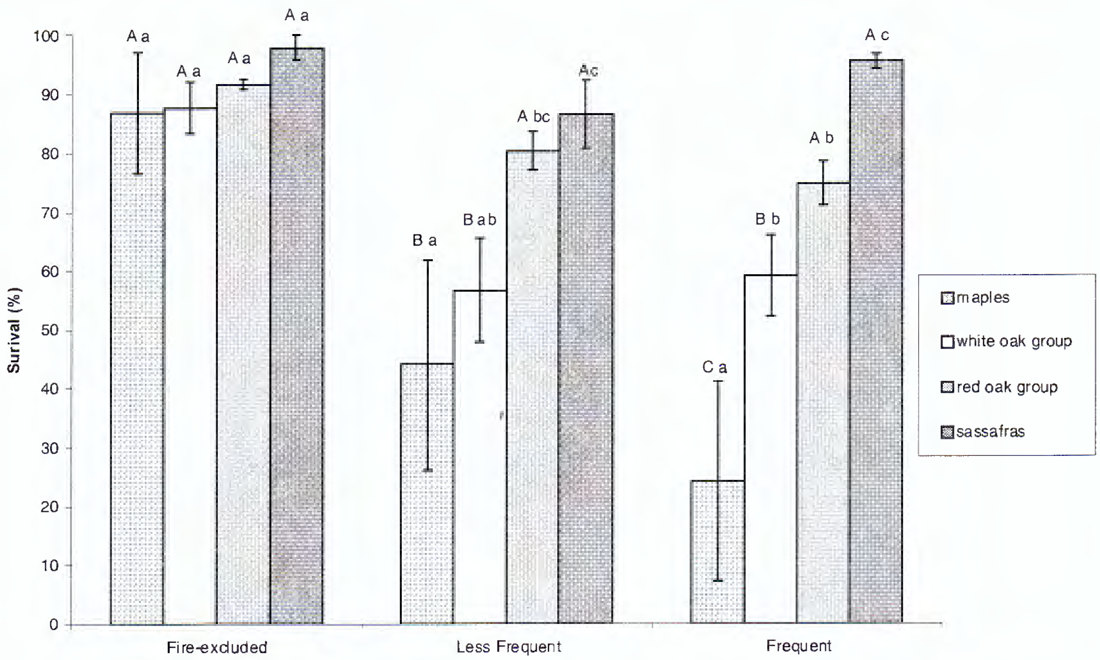


Figure 6: Seedling survival (%) across treatments. Seedlings were grouped into three species groups and sassafras. Different uppercase letters denote significant differences across treatments within species groups. Different lowercase letters denote significant differences between species groups within treatments.



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