2017

Vegetation Response to Repeated Prescribed Burning and Varied Wildfire Severity in Upland Forests on the Cumberland Plateau, Kentucky

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Digital Object Identifier: https://doi.org/10.13023/ETD.2017.094

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VEGETATION RESPONSE TO REPEATED PRESCRIBED BURNING AND VARIED WILDFIRE SEVERITY IN UPLAND FORESTS ON THE CUMBERLAND PLATEAU, KENTUCKY.

THESIS

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in the College of Agriculture, Food, and the Environment at the University of Kentucky

By

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2017

ABSTRACT OF THESIS

Vegetation response to repeated prescribed burning and varied wildfire severity in upland forests on the Cumberland Plateau, Kentucky

As a result of decades of fire suppression, oaks (*Quercus* L.) and other disturbance-dependent tree species are experiencing widespread regeneration failure. Today, fire takes the form of relatively low to moderate intensity prescribed fire, used to restore fire adapted ecosystems, and wildfires which often vary in severity. I investigated long-term changes to forest structure and composition in response to repeated prescribed burning followed by an extended period of no fire. Burning reduced total basal area, midstory stem density and sapling stem density. However, the fire-free interval significantly increased sapling layer stem densities of oaks and competitor species. This research shows that repeated prescribed fire, followed by a fire-free interval, can allow oak seedlings to grow into sapling sized stems, but competitors also increase in density. I also investigated relationships between varying wildfire severity and stand structure, basal area, and tree recruitment. Both stem density and total basal area were significantly and negatively related to fire severity. Oak and pine recruitment was significantly and positively related to fire severity whereas competitors had no relationship. The positive relationships with fire severity and oak or pine sapling recruitment could have important implications for managers using prescribed fire or managing areas after wildfire.

Keywords: fire, wildfire, oak, maple, succession, prescribed fire

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April 4, 2017

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DEDICATION

I dedicate my work to my twin brother, Dylan Scott Poynter, and my two children, Nevaeh and Harrison.

ACKNOWLEDGEMENTS

First and foremost, I would like to thank my mother and my little brother for always supporting my dreams. I also give thanks to my wonderful girlfriend, Jade, for being my biggest fan and always encouraging me. I also would not be here if it were not for two high school teachers who inspired me to love science. I give credit to Mr. Gerst and Mrs. Weber for spurring my curiosity in biology, environmental science, chemistry, and physics. I would also like to thank Larry Brewer for his mentorship and friendship.

I have met and worked with many people at the University of Kentucky and they have all taught me or helped me in some way. In particular, I'd like to think my colleague, Devin Black, for her help with research, but also for being a role model to me.

Finally, I would like to thank my committee members, Dr. Mary Arthur, Dr. John Lhotka, and Rob Paratley, for supporting me and giving thoughtful advice. Dr. Mary Arthur deserves my utmost thanks because she has believed in my abilities and has graciously offered her constructive criticism and words of encouragement during my time in her lab.

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Chapter One

Introduction

Introduction

Fire has been part of the eastern North American landscape for thousands of years (Clark et al., 1996). Most of these fires were started by Native Americans (Delcourt and Delcourt, 1997; Delcourt and Delcourt, 1998; Whitney, 1994), who used fire to increase the quality of hunting grounds, make traveling easier, and to support agriculture (Ruffner, 2006). The forest species composition of eastern North America was influenced by this burning, as evidenced in charcoal and paleoecology data which indicate a close relationship between fire and oak dominance (Delcourt and Delcourt, 1998; Patterson, 2005). The frequency of fire increased when Europeans arrived on the landscape and used fire to clear land. As harvesting increased unregulated logging activities and the resulting debris led to widespread and severe fire (Brose et al., 2001; McEwan et al., 2007). In response, federally-regulated fire suppression was initiated in the early 20th century. Due to the suppression of fire, ecosystems that were historically maintained by periodic burning, such as oak forests and woodlands, have been changing in structure and composition (Fei et al., 2011; McEwan et al., 2011; Nowacki and Abrams, 2015). Nowacki and Abrams (2008) have coined this process “mesophication” and described it as the increase in abundance and density of shade-tolerant species resulting in a positive feedback loop in which conditions become increasingly shady and moist, further decreasing flammability and favoring shade-tolerant, fire-intolerant species regeneration. In particular, oak (*Quercus* spp. L.) dominated ecosystems in the eastern United States have become denser with more fire-sensitive, mesophytic species such as maple (*Acer* spp. L.) in the understory and midstory (Hart and Grissino-Mayer, 2008; Hart et al., 2012). Under closed canopy conditions and without fire to reduce stem density and kill the fire-intolerant species in the understory, species that depend on higher light environments to persist are failing to regenerate (Fei et al., 2011; Hutchinson et al., 2008; Nowacki and Abrams, 2015).

Depending on burn severity, the effects of fire can range from reduction of stem density of smaller understory stems to mortality of overstory trees. Today, fire in the eastern U.S. takes varied form, from low to moderate intensity prescribed burns to wildfires that vary widely in intensity and severity. Managers are increasingly using prescribed fire as a tool to restore fire-dependent ecosystems (USDA Forest Service Daniel Boone National Forest, 2005; USDA Forest Service Mark Twain National Forest, 2005; USDA Forest Service Ozark-St. Francis National Forests, 2005). These prescribed fires are often low intensity ground fires which consume leaf litter and top kill seedlings and saplings. Because of the long period of fire suppression and minimal impacts to stand structure, a single prescribed fire generally does not put oaks at a competitive advantage (Brose et al., 2013). As a result, many researchers have experimented with repeated fires, but with mixed results, sometimes finding oak benefits but other times finding competitor species benefit as well (Arthur et al., 2015; Blankenship and Arthur, 2006; Hutchinson et al., 2012a; Keyser et al., 2017). Recurring fire top kills oaks as well as competitor species, preventing oaks from growing to a large enough size to resist subsequent fires (Knapp et al., 2016; Green et al., 2010). Thus, a long fire-free interval is needed to allow oak seedlings to reach a size that is resistant to fire (Knapp et al., 2017; Arthur et al., 2012). Long-term prescribed fire studies incorporating a decadal long fire-free interval after burning are rare in this region. It is unclear whether a fire-free interval after repeated burning will facilitate oak advance regeneration to grow to a larger size.

While prescribed burning is conducted under controlled conditions, wildfires are inherently less controlled, often occurring during conditions which may enable their spread, intensity, and severity. As a result, wildfire severity and intensity can vary widely across the landscape depending on topography, fuel properties, weather conditions, and control tactics (Franklin et al., 1997). With climate warming and changing weather patterns, wildfire may become more common as fuels become increasingly dry and flammable (Lafon and Quiring, 2012; Mitchell et al., 2014). As wildfires become more prevalent it will be important to understand how forested areas burned by wildfire recover. Vegetation development after wildfire can also give us insights into how disturbance dependent species, such as oaks, respond to a fire that includes areas burned with greater severity than typical prescribed burns in the region.

In Chapter Two of this thesis, I investigated the effects of a long-term prescribed fire study in an upland oak dominated forest in the Cumberland Plateau of eastern Kentucky in which repeated fire implemented over 10 years was followed by ~10 years of a fire-free interval. I examined changes to stand structure and species composition in response to burning but also to a subsequent fire-free interval ranging from 10-11 years. This study provides critical insights into using prescribed fire over a decade and is among the first studies in this region to include measurements following an extended fire-free interval. Prior research demonstrated that fire can reduce the density of competitor species, although fire sensitive species also responded positively (Blankenship and

Arthur, 2006). This study addresses the question, “How has forest structure and composition changed due to twenty years of fire-exclusion?” and also, pertaining to burn units, “will the cessation of fire following repeated burning allow oak seedlings to grow to a competitive size and how has overall stand structure and composition changed?”

Chapter Three assesses changes in forest stand structure and species composition of an oak-pine dominated forest in response to a wildfire which burned in 2010 with varied fire severity across the 674 ha containment area. Low intensity prescribed fire may indiscriminately top-kill small diameter stems, but how does wildfire of varying severity affect regeneration? Do species or species groups respond differently to different severities of wildfire? These questions are important to answer for managers working in a future where wildfire will likely be more prevalent.

Chapter Two

The response of an upland oak forest to repeated burning followed by an extended fire-free interval

Introduction

Oaks (*Quercus* spp. L.) are a foundational genus in the eastern United States (Hanberry and Nowacki, 2016), but their continued dominance is threatened by fire suppression (Fei et al., 2011; Nowacki and Abrams, 2015). In the central hardwood forest and central Appalachians, most oak species have low to intermediate shade tolerance (Burns and Honkala, 1990). Therefore, they cannot successfully compete in the understory of forests which are dense and shady because of fire suppression and the lack of other disturbances (McEwan et al., 2011; Nowacki and Abrams, 2008). In comparison to other hardwood species in the region, oaks as a group have several physical attributes which suggest that they are adapted to an environment in which fire is a key disturbance agent and that fire disturbance in the past contributed to the creation or maintenance of conditions such as canopy openness, that supports oak domination (Abrams, 1996; Burns and Honkala, 1990).

Physical adaptations of oaks that are beneficial in an environment where fire is a periodic disturbance include hypogeal germination of acorns (Brose and Van Lear, 2004), dormant buds below the root collar (Brose and Van Lear, 1998), high root-to-shoot ratios allowing vigorous sprouting after top-kill (Merz and Boyce, 1956), the generally thick bark of large trees of many species (Hare, 1965; Hengst and Dawson, 1994), and the ability to rapidly and effectively compartmentalize wounds (Smith and Sunderland, 1999). Not only are many oaks fire resistant, but burning can also facilitate germination and oak seedling establishment by reducing the litter layer, destroying seeds of competitor species, and reducing the size and density of competitors in the understory (Brose et al., 2013). Although oaks have several adaptations which make them able to establish and persist after fire, they are still vulnerable at different stages of their life cycle, namely as acorns and small seedlings which are not able to resist fire damage and may reside in an environment without sufficient light. This suite of oak adaptations to burning, then, should allow oak seedlings and saplings to respond positively to prescribed burning, but only in circumstances in which the canopy is sufficiently open and fire cessation occurs long enough for oak seedlings to grow large enough to resist future fire. Yet, the effects of prescribed burning on oak regeneration are mixed and long-term studies that include a significant fire-free interval are rare. Even rarer are studies which incorporate a long-term fire-free interval after multiple fires to allow oak regeneration to reach a size which is resistant to future fire (Arthur et al., 2012).

Single prescribed fires have little effect on reducing stem densities of larger stems (Brose et al., 2013). Thus, it is recognized that repeated fire will be needed to influence the structure and composition of many stands which have not burned in decades. Repeated burning reduces sapling stem density dramatically, but kills small oak stems as well as mesophytic species (Brose et al, 2013). Some targeted species, such as red maple, can sprout after top kill and stay competitive with oak despite repeated burning (Blankenship and Arthur, 2006; Green et al., 2010). A major canopy disturbance after repeated burning, such as a natural gap formation or partial harvest, provides the most benefit to oak regeneration (Brose, 2014; Hutchinson et al., 2012b; Johnson et al., 2009).

However, natural disturbance is unpredictable in timing and intensity, and mechanical manipulation is not always possible where harvesting is restricted or costs to treat large areas are prohibitive. In these situations, prescribed fire may be the only tool available to facilitate oak regeneration, but studies have shown that repeated burning can limit recruitment by repeatedly killing small stems before they are large enough to resist fire and be recruited to the canopy (Blankenship and Arthur, 2006; Knapp et al., 2015; Knapp et al., 2017). Moreover, it is unlikely that surviving oaks will be recruited to the canopy unless current conditions are conducive to oak recruitment or there is a subsequent canopy disturbance caused by fire or another disturbance agent (Arthur et al., 2012). Several authors have called for a long fire-free interval after burning to allow oak seedlings to establish and grow large enough to resist future fire (Arthur et al., 2012; Knapp et al., 2015; Knapp et al., 2017). Unfortunately, there are no published studies that investigate a decade-long fire-free interval after repeated burning in this region.

This research takes advantage of permanent plots established in 1995 in the Red River Gorge Geological Area within Daniel Boone National Forest, Kentucky that allowed me to investigate changes in stand structure and species composition after ~10 years of periodic prescribed burning followed by a ~10 year fire-free interval. The specific hypotheses for this study are as follows:

H1: A fire-free interval will significantly increase the density of oak saplings and oak seedling regeneration in burn treatments.

H2: After repeated burning, burned treatments will have lower basal area, midstory stem density, and sapling stem density than the control.

Methods

Site Description:

An investigation of the effects of prescribed fire on forest structure and species composition was initiated in 1995 in collaboration with the USDA Forest Service. Three non-conterminous ridges were initially selected as replicates in the Cumberland Ranger District of Daniel Boone National Forest, in the Red River Gorge Geological Area which is in the Cliff Section of the Cumberland Plateau (Braun, 1950). Over the two decades of research, one of the three ridges was abandoned as a study site due to impacts of hiking and camping in the area, leaving two study sites, Whittleton Ridge and Klaber Ridge. The geological substrate of the sites are shales and siltstones of the Upper and Lower members of the Lee formation (Weir and Richards, 1974). Klaber Ridge soils are classified as Latham-Shelocta silt loams which are moderately deep and well-drained, slowly permeable clayey soils of the subgroups Typic and Aquic Hapludults (Avers et al., 1974). Whittleton Ridge soils are classified as Gilpin silt loam, which is moderately deep and well-drained, with a lower subsoil of silty clay loam of the subgroup Typic Hapludult (Hayes, 1993). Mean annual precipitation is 130cm, with a mean growing season of 176 days. Mean annual temperature is 12° C, with mean daily temperatures ranging from 0° C in January to 31° C in July (Foster and Conner, 2001).

The sites are mature, second-growth forests dominated by chestnut oak (*Quercus montana* Willd.) and scarlet oak (*Q. coccinea* Muenchh.), with black oak (*Q. velutina*

Lam.), white oak (*Q. alba* L.), and hickories (*Carya* spp. Nutt.) frequently present. Pitch pine (*Pinus rigida* Mill.), shortleaf pine (*P. echinata* Mill.), and Virginia pine (*P. virginiana* Mill.) are also common in the overstory. The understory is dominated by red maple (*Acer rubrum* L.), white pine (*P. strobus* L.), blackgum (*Nyssa sylvatica* Marsh.), and sourwood (*Oxydendrum arboreum* L.). Prior to this study, no forest management activities or fire had occurred in these stands for at least the past 35 years (Blankenship and Arthur, 2006). As of 2016, the overstory oaks are approximately 80 years old and red maples are approximately 60 years old (Washburn and Arthur, 2003).

Experimental Design and Fire Application:

Whittleton and Klaber Ridges each had three treatment units: a fire-excluded control and two units planned for frequent and less frequent burning, with plans to apply prescribed fire to the burn units periodically in an adaptive management approach as deemed necessary by USFS personnel. Weather conditions prevented the planned 1996 burn treatment unit on Whittleton Ridge from being burned until 1997. As of 2016, treatment units include a fire-excluded unit, a less frequent (3 burns) burn unit, and a more frequent (4 burns) burn unit on both sites. Burn conditions and timing can be found in Green et al. (2010).

All prescribed burns were performed by USDA Forest Service personnel. Using drip torch ignition, firing began from the highest points on the ridges and was pulled downslope from the ridges into the wind, burning across the whole treatment area. Strip-firing and point-source were used if backing fires were not sufficiently intense (>0.3m flame length; Richardson, 1995). Flame temperatures were measured with pyrometers made by painting Tempilac temperature-sensitive paints on aluminum tags attached to stakes at 15 cm above ground and at the soil surface. Temperatures were measured at four locations in each plot. Fires from 1995 through 2000 were performed between March 13 and 31; fires after 2000 were conducted later in the year (April) in recognition of a need for early growing season burning to further control competing woody vegetation. All fires were low intensity surface fires with flame length between 0.3 m and 2 m (Green et al., 2010).

In 1995, eight permanent circular plots of 500m^2 were established in each treatment unit resulting in 48 plots across the two study ridges. Since then, recreational impacts from hiking and camping as well as wildfires from escaped campfires resulted in the loss of 2 plots in the frequent burn unit and 5 plots in the less frequent burn unit of Klaber Ridge, for a total sample size of 41 plots across the two ridges at the time of last sampling, in 2015.

Data collection:

Circular plots of 500 m^2 area were sampled in 1995 prior to burning and then intermittently. Plots were sampled most recently in July and August of 2015. Data were collected in the same manner as in Blankenship and Arthur (2006). For all stems ≥2 cm diameter at breast height (DBH), the DBH, number of basal sprouts, number of browsed sprouts, and species were recorded. Regeneration stems (<2cm DBH) were tallied by species in a circular 25 m^2 area centered about the original plot center.

Data Analysis:

I analyzed data collected before treatment (1995), after cessation of burning (2007), and after a fire-free period (2015) to assess changes in stem density or basal area for all stems and select species and species groups by overstory (≥20cm DBH), midstory (10-20cm DBH), and sapling (2-10cm DBH) size classes. These categories were based upon previous literature which demonstrates that prescribed fire differentially affects stems of different diameters with low mortality of stems ≥ 20 cm DBH (Arthur et al., 2015; Blankenship and Arthur, 2006; Hutchinson et al., 2005). For the regeneration (stems <2cm DBH), I analyzed stem density for all stems and select species and species groups in 1995, 2003 and in 2015. I used data from 2003 (before the last burn) because regeneration data were not collected in 2007 (after burning). I am limited to analyzing changes from 2003 and 2015 to see if an additional burn followed by an extended fire-free period influenced regeneration density.

Total stem density and stem density by species or species group were analyzed for saplings, midstory, and overstory stems when data were sufficient for analysis (distribution approximately normal and lacking an abundance of zero counts). Species groups include oaks (*Quercus* spp.), red maple (*Acer rubrum*), sourwood (*Oxydendrum arboreum*), blackgum (*Nyssa sylvatic)* and white pine (*Pinus strobus*). For statistical analysis, a repeated measures ANOVA was utilized in the linear mixed model framework including treatment, year, site, and treatment*year as fixed effects and plots as a random effect. PROC GLIMMIX in SAS™ Statistical Software (SAS, 2011) was used for analysis. I used unstructured covariance to model the correlation of plots through time. When there was a significant treatment effect or interaction, contrasts were used to test for differences among treatments in 2007 (after burning) and in 2015 (after fire-free interval) as well as between years within each treatment to assess change. Data were transformed when necessary to meet ANOVA assumptions.

Results

Burning led to a significant decrease in total basal area for the more frequent burn treatment but there was no significant changes in the less frequent and fire-excluded control (Table 2.1; Figure 2.1; Appendix B). During the fire-free interval (2007 to 2015), total basal area significantly decreased in the frequent burn treatment. However, both the less frequent and frequent burn treatments had significantly less total basal area in 2015 when compared to the control (Figure 2.1). The frequent burn treatment had the lowest total basal area of all treatments being significantly lower than the less frequent treatment and fire-excluded control (Figure 2.1).

		1995	2007	2015
Total Basal Area (m^2 ha^{-1})				
	Fire-excluded	32.3 (3)^{a}	30.8 (2)^{aA}	31.9 (1)^{A}
	Less Frequent	23.4 (3)^{a}	27.4 (2)^{aA}	26.9 (1)^{A}
	Frequent	34.3 (3)^{a}	26.3 (3)^{bA}	22.3 (2)^{B}
Overstory stem density (stems ha^{-1})				
Total	Fire-excluded	262 (40)^{a}	243 (15)^{aA}	243 (14)^{A}
	Less Frequent	218 (35)^{a}	251 (16)^{aA}	262 (19)^{A}
	Frequent	286 (35)^{a}	234 (27)^{aA}	201 (22)^{A}
Oak	Fire-excluded	188 (42)^{a}	186 (15)^{aA}	176 (15)^{A}
	Less Frequent	209 (37)^{a}	200 (20)^{aA}	215 (21)^{A}
	Frequent	257 (34)^{a}	211 (27)^{aA}	167 (21)^{B}
Red maple	Fire-excluded	13 (9)^{a}	33 (8)^{bA}	46 (9)^{A}
	Less Frequent	0^{a}	13 (6)^{aA}	24 (9)^{A}
	Frequent	7 (7)^{a}	9 (5)^{aA}	14 (6)^{A}
Midstory stem density (stems ha^{-1})				
Total	Fire-excluded	338 (46)^{a}	304 (21)^{aA}	280 (23)^{A}
	Less Frequent	355 (41)^{a}	209 (32)^{bA}	136 (19)^{B}
	Frequent	350 (56)^{a}	109 (17)^{bA}	73 (12)^{B}
Oak	Fire-excluded	144 (47)^{a}	49 (10)^{aA}	33 (9)^{B}
	Less Frequent	191 (34)^{a}	87 (12)^{aA}	60 (9)^{B}
	Frequent	114 (56)^{a}	109 (16)^{aA}	73 (11)^{B}
Red maple	Fire-excluded	125 (23)^{a}	195 (18)^{bA}	188 (15)^{A}
	Less Frequent	91 (34)^{a}	71 (22)^{aA}	31 (9)^{B}
	Frequent	114 (43)^{a}	14 (4)^{bA}	10 (5)^{A}
White pine	Fire-excluded	13 (9) ^{a}	19 (7)^{aA}	23 (7)^{A}
	Less Frequent	9 (9)^{a}	9 (7)^{aA}	11 (7)^{A}
	Frequent	50 (25)^{a}	10 (6)^{aA}	0^{B}
Sapling stem density (stems ha^{-1})				
Total	Fire-excluded	1244 (139)^{a}	834 (77)^{bA}	729 (70)^{A}
	Less Frequent	1355 (142)^{a}	104 (25)^{bA}	467 (125)^{B}
	Frequent	1621 (100)^{a}	47 (16)^{bA}	1624 (307)^{B}
Oak	Fire-excluded	13 (13)^{a}	5 (2)^{aA}	3 (2)^{A}
	Less Frequent	73 (30)^{a}	7 (4)^{bA}	4 (4)^{A}
	Frequent	79 (28)^{a}	0^{bA}	100 (48)^{B}
Red maple	Fire-excluded	681 (91)^{a}	369 (37)^{bA}	253 (29)^{B}
	Less Frequent	564 (142)^{a}	56 (28)^{bA}	298 (118)^{B}
	Frequent	900 (140)^{a}	23 (12)^{bA}	613 (123)^{B}
White pine	Fire-excluded	219 (96)^{a}	280 (70)^{bA}	306 (69)^{A}
	Less Frequent	100 (30)^{a}	2 (2)^{bA}	0^{A}
	Frequent	114 (51)^{a}	0^{bA}	10 (6)^{B}
Sourwood	Fire-excluded	50 (22)^{a}	24 (6)^{aA}	29 (7)^{A}
	Less Frequent	146 (78)^{a}	11 (5)^{bA}	131 (29)^{B}
	Frequent	138 (38)^{a}	1 (1)^{bA}	356 (40)^{B}
Blackgum	Fire-excluded	113 (31)^{a}	114 (27)^{aA}	78 (20)^{B}
	Less Frequent	255 (58)^{a}	22 (9)^{bA}	16 (7)^{A}
	Frequent	193 (47)^{a}	17 (10)^{bA}	90 (23)^{B}

Table 2.1. Total basal area ($m^2\ ha^{-1}$) for all stems ≥2cm DBH and absolute density (stems ha^{-1}) for overstory (≥20cm DBH), midstory (10-20cm DBH) and saplings (2-10cm DBH). Values are represented as mean (standard error). Because I tested differences between 1995 and 2007 and then 2007 and 2015 for each treatment, means within treatments that have the same lowercase letter *or* uppercase are not significantly different from each other ($\alpha = 0.05$). For example, in the frequent burn treatment, blackgum sapling stem density in 1995 was significantly different from blackgum sapling stem density in 2007 (different lowercase letters). Also, blackgum sapling stem density in 2015 was significantly different than blackgum sapling stem density in 2007 (two different uppercase letters).

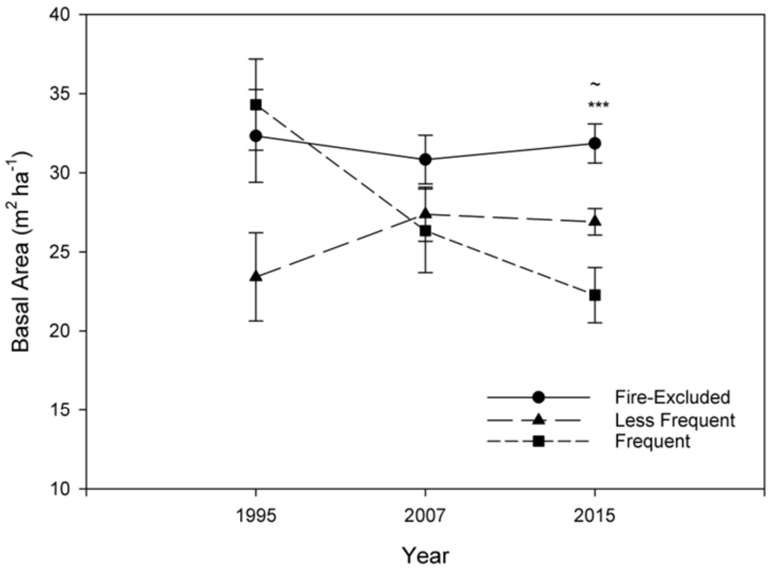

Figure 2.1. Mean total (stems ≥2 cm DBH) basal area (m^2 ha^{-1}) for the fire-excluded control and burn treatments before fire (1995), after fire (2007), and after a fire-free interval (2015). *** indicates both burn treatments were significantly different than the control, ~ indicates burn treatments were significantly different from each other (α = 0.05). Error bars are ± standard error of the mean.

Stem Density:

While fire had no significant effect on total overstory stem density, absence of fire led to significantly greater red maple growing into the overstory (Figure 2.2; Appendix A). Red maple overstory stem density increased significantly in the fire-excluded control from 1995 to 2007 whereas the changes in the burn treatments were not significant (Table 2.1). This resulted in both burn treatments having significantly lower red maple overstory stem density than the control in 2007 and also in 2015 (Figure 2.2). There was a significant treatment*year interaction for oak overstory stems. Although treatments did not differ significantly among years, the frequent burn treatment significantly decreased in oak overstory stems from 1995 to 2007 and from 2007 to 2015 (Table 2.1).

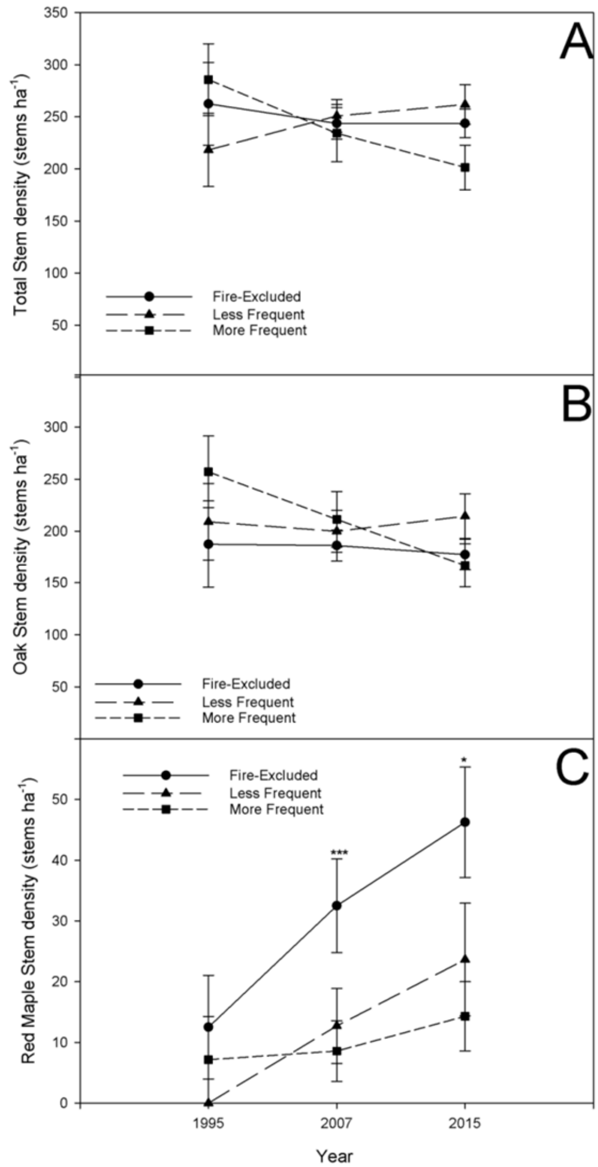

Figure 2.2. Mean overstory (≥20 cm DBH) stem density (stems ha^{-1}). (A) Total overstory stem density, (B) Oak overstory stem density, and (C) Red maple overstory stem density for the fire-excluded control and burn treatments before fire (1995), after fire (2007), and after a fire-free interval (2015). *** indicates both burn treatments were significantly different than the control, * indicates only the frequent burn treatment was significantly different than the control (α = 0.05). Error bars are ± standard error of the mean.

Burning significantly reduced midstory (10-20cm DBH) stem density (Figure 2.3; Table 2.1). Burn treatments had significantly lower midstory stems when compared to the control in 2007 and in 2015 (Figure 2.3). The frequent fire treatment produced significantly lower midstory stems than the less-frequent.

Despite decreases in oak midstory stems from 1995 to 2007 across treatments (Figure 3), none were significant (Table 2.1). However, all treatments had significant decreases in oak midstory stems from 2007 to 2015 resulting in oak midstory stem density being significantly greater on the less frequent burn treatment in 2015 when compared to the control (Table 2.1; Figure 2.3).

Red maple midstory stem density decreased in burn treatments but increased in the fire-excluded control, resulting in the control having significantly greater red maple midstory stems in 2007 and in 2015 (Table 2.1; Figure 2.3). Burning significantly reduced red maple midstory stems in the frequent burn treatment, but change in the less frequent treatment was not significant (Table 2.1). However, from 2007 to 2015, red maple midstory stems decreased significantly in the less frequent burn treatment, but not in the frequent treatment (Table 2.1).

White pine was the only other species or species group for which there were significant differences with treatment or time. There was a trend towards increasing white pine density in fire-excluded sites and a decline in the frequent burn treatment (Table 2.1; Figure 2.3). By 2015, the frequent burn treatment had significantly lower white pine midstory stem density than the control whereas the less frequent treatment did not differ. A significant site effect ($p = 0.0369$) revealed that, across all years and treatments, Klaber

Ridge had greater midstory stem density of white pine (22 stems/ha; S.E. ± 5 stems/ha) compared to Whittleton (11 stems/ha; S.E. ± 5 stems/ha).

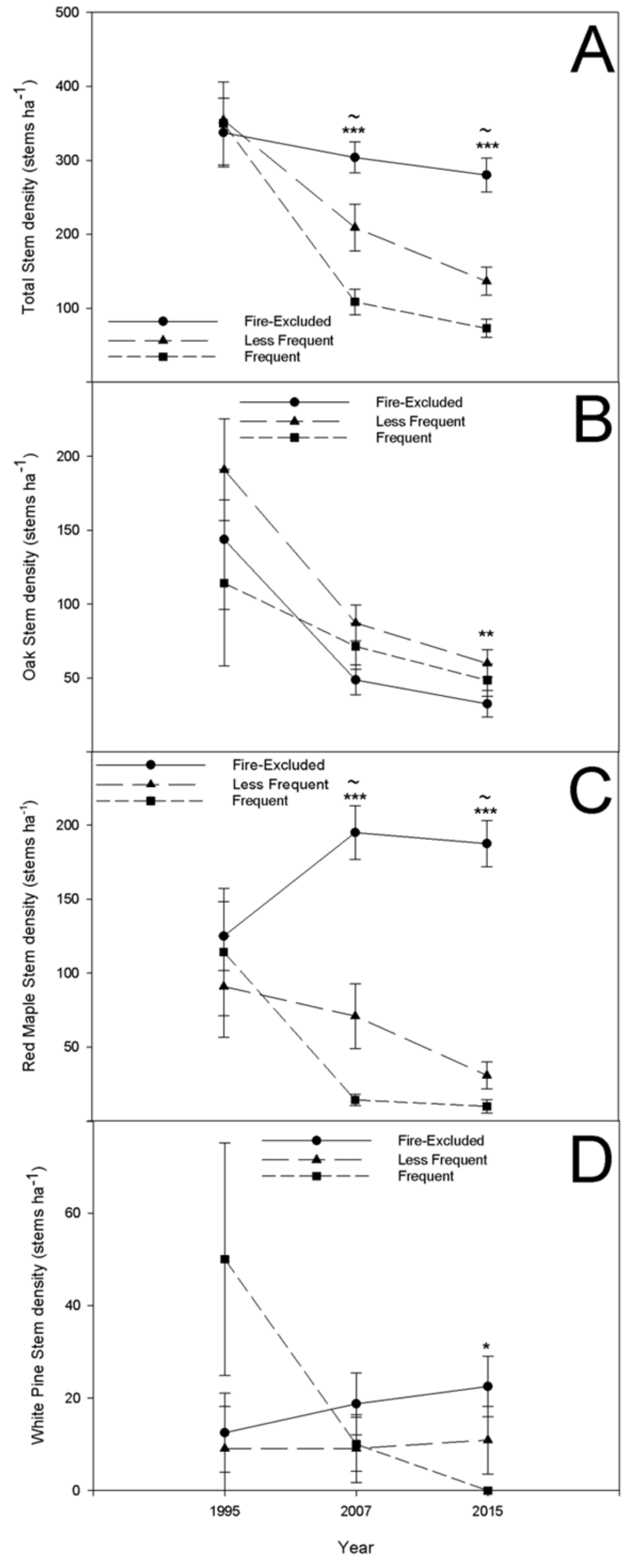
A
B
C
D
Fire-Excluded
Less Frequent
Frequent
Total Stem density (stems ha^{-1})
Oak Stem density (stems ha^{-1})
Red Maple Stem density (stems ha^{-1})
White Pine Stem density (stems ha^{-1})
~
***
**
*
1995
2007
2015
Year

Figure 2.3. Mean midstory (10-20 cm DBH) stem density (stems ha^{-1}) for fire-excluded control and burn treatments before fire (1995), after fire (2007), and after a fire-free interval (2015). A) Total midstory stem density B) Oak midstory stem density C) Red maple midstory stem density. D) White pine midstory stem density *** indicates both burn treatments were significantly different than the control, ** indicates the less frequent burn treatment was significantly different from the control, ~ Indicates that the burn treatments were significantly different from each other ($\alpha = 0.05$). Error bars are ± standard error of the mean.

Fire had a significant impact on sapling (2-10cm DBH) density with lower sapling density after repeated burning followed by an increase in sapling density during the fire-free period (Figure 2.4). Total sapling stem density decreased significantly across all treatments from 1995 to 2007 (Figure 2.4), but the decline was much greater in the burn treatments compared to the fire-excluded control (Table 2.1). During the fire-free interval, sapling density increased significantly in both burn treatments (Table 2.1; Figure 2.4), but remained flat in the control. As a result, the frequent burn treatment had significantly greater sapling stem density compared to the less frequent burn treatment and the fire-excluded treatment in 2015 whereas the less frequent and control were similar (Figure 2.4).

On burned sites, oak sapling stems decreased significantly compared to the fire-excluded control did not (Table 2.1; Figure 2.4). In response to the fire-free interval, oak sapling stems increased significantly in the frequent burn treatment but not in the less frequent (Table 2.1). As a result, oak sapling stem density was significantly greater in the frequent burn treatment compared to both the less frequent and control (Figure 2.4).

Red maple sapling density decreased significantly in all treatments between 1995 and 2007 but again, the decline was greater in burn treatments compared to the fire-excluded control (Table 2.1; Figure 2.4). During the fire-free interval, red maple sapling stem density increased significantly in the less frequent and frequent burn treatments from (Table 2.1), whereas red maple sapling stem density in the fire-excluded control decreased significantly (accompanied by a significant increase in red maple midstory stems during this same period). In 2015, the frequent burn treatment had significantly higher red maple sapling densities than both the less frequent and fire-excluded treatment (Figure 2.4).

White pine sapling density decreased significantly in the less frequent and frequent burn treatments, whereas white pine sapling density increased significantly in the fire-excluded control during this same period (Table 2.1; Figure 2.4). Burn treatments had significantly lower white pine sapling density in 2007 when compared to the fire-excluded control and this trend continued during the fire-free interval, resulting in significantly lower white pine sapling density in 2015 in burn treatments compared to the control (Figure 2.4).

Similar to red maple, sourwood sapling stems decreased significantly in burn treatments, and then significantly increased during the fire-free interval (Table 2.1; Figure 2.4). In the fire-excluded treatment, sourwood sapling stem density did not differ through time. Only the frequent burn treatment had significantly lower sourwood sapling stem density than the control in 2007, but both burn treatments were significantly higher than the control in 2015 (Figure 2.4).

Burning significantly reduced blackgum sapling stems in the less frequent and frequent burn treatments (Table 2.1; Figure 2.4). Only the frequent burn treatment saw significant increases in blackgum sapling stems during the fire free interval (Table 2.1). In 2015, the less frequent burn treatment had significantly lower blackgum sapling density than the frequent burn treatment and the control (Figure 2.4).

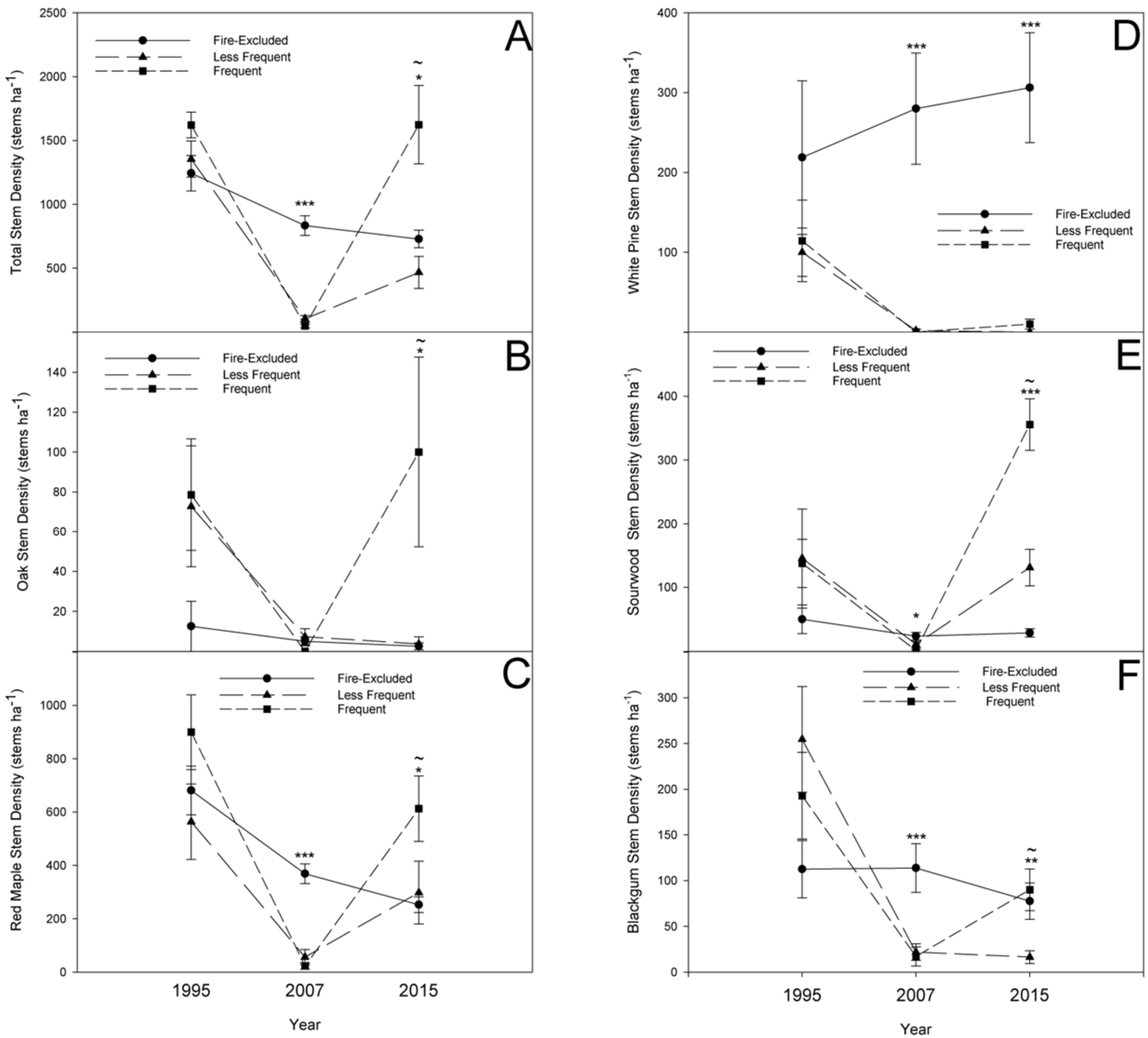
A
Fire-Excluded
Less Frequent
Frequent
Total Stem Density (stems ha-1)
B
Oak Stem Density (stems ha-1)
C
Red Maple Stem Density (stems ha-1)
D
White Pine Stem Density (stems ha-1)
E
Sourwood Stem Density (stems ha-1)
F
Blackgum Stem Density (stems ha-1)
1995
2007
2015
Year

Figure 2.4. Mean sapling (2-10 cm DBH) stem density (stems ha^{-1}) for fire-excluded control and burn treatments before fire (1995), after fire (2007), and after a fire-free interval (2015). Make note of differing y-axis scales depending on the response variable. (A) Total sapling stem density (B) Oak sapling stem density (C) Red maple sapling stem density (D) White pine sapling stem density (E) Sourwood sapling stem density (F) Blackgum sapling stem density. *** indicates both burn treatments were significantly different than the control, ** indicates the less frequent burn treatment was significantly different from the control, * indicates only the frequent burn treatment was significantly different than the control. ~ Indicates that the burn treatments were significantly different from each other ($\alpha = 0.05$). Error bars are ± standard error of the mean.

Prior to burning, oak regeneration density (stems <2cm DBH) was not significantly among treatments (Figure 2.5). However, oak regeneration stem density increased significantly from 2003 to 2015 in the frequent burn treatment ($p = 0.0032$), resulting in significantly greater oak regeneration density in the frequent burn treatment when compared to the less frequent treatment and the control (Figure 2.5). Hard pine regeneration increased significantly in the frequent burn treatment and control from 1995 to 2003 ($p = 0.0047$ and $p = 0.0095$). However, hard pine regeneration decreased significantly from 2003 to 2015 in the control ($p = 0.0454$). These changes resulted in both burn treatments having significantly higher hard pine regeneration when compared the control (Figure 2.5). Burning had no effect on total regeneration, red maple regeneration, white pine regeneration, sourwood regeneration, or blackgum regeneration.

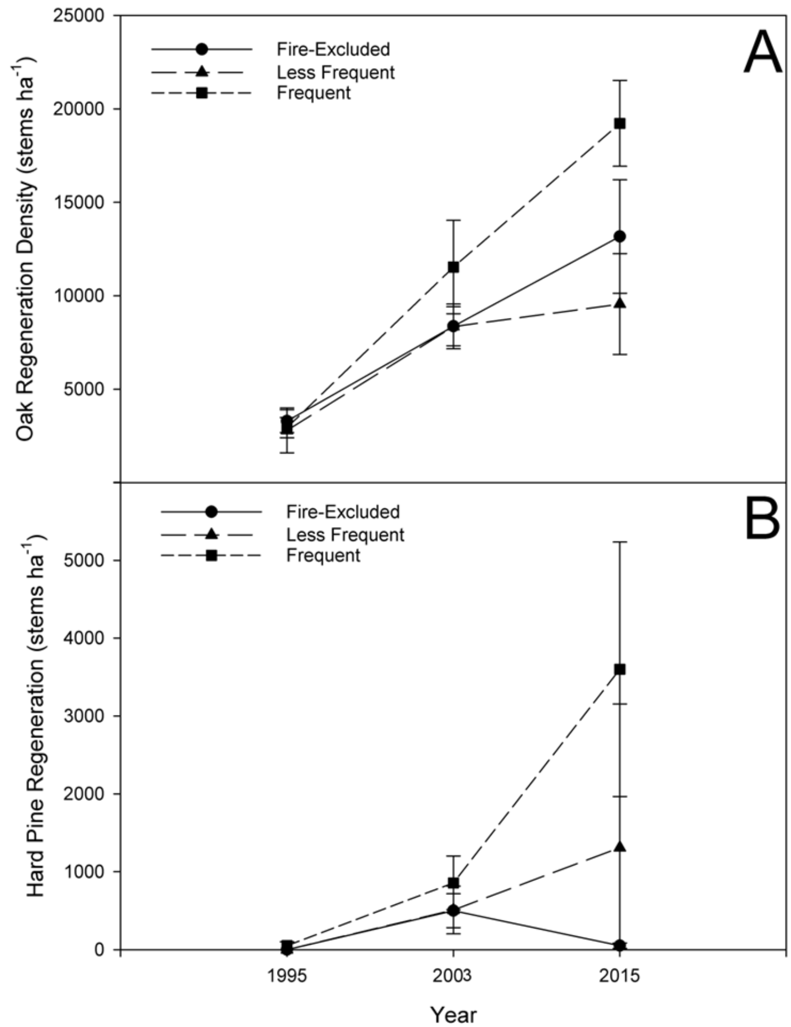

Figure 2.5. Mean regeneration (stems <2cm DBH) stem density (stems ha^{-1}) for fire-excluded control and burn treatments before fire (1995), before an additional burn (2003), and after a fire-free interval (2015). A) Oak regeneration density B) Hard pine regeneration density *** indicates both burn treatments were significantly different than the control, * indicates the frequent burn treatment was significantly different from the control, ~ indicates that the burn treatments were significantly different from each other ($\alpha = 0.05$). Error bars are ± standard error of the mean.

Discussion

Repeated burning reduced total stem density in the midstory and sapling layers but also reduced the density of red maple in these layers (Figure 2.6). The stem densities of overstory red maples in burn treatments were significantly lower when compared to the fire-excluded control (Figure 2.2). A fire-free period of approximately 10 years led to an increase in oak sapling density in the frequent burn treatment, but competitor saplings also took advantage of the fire-free period by increasing in density (Figure 2.4). This corroborates other studies in this region which found that competing species recover rapidly following prescribed fire (Albrecht and McCarthy, 2006; Arthur et al., 2015; Blankenship and Arthur, 2006; Hutchinson et al., 2012b). Most of these competitors which have reached sapling size in burn units were likely from sprout clumps due to top-kill. A previous study using the same plots found that burning produced more basal sprouts and red maple dominated the total number of basal sprouts (Blankenship and Arthur, 2006). During sampling in 2015, I noticed groups of red maple or sourwood stems where a larger tree was once present, but has since fallen over (pers. obs.). These clumps would sometimes have stems >2cm DBH which allowed them to be recorded for the sapling size class in 2015. Unfortunately, I did not distinguish between sprout saplings and individually rooted saplings in 2015 as it was not always apparent if a sapling was from a sprout clump or individual root system. Additional burning or herbicide treatment may be needed to control sprout clumps from competitor species. Other studies which include a period of no burning did not see dramatic increases in oak sapling stem densities but their fire free interval did not extend beyond 4 years (Hutchinson et al., 2012b; Keyser et al., 2017).

Although oak saplings did increase in density in the frequent burn treatment, there was no significant change for the less frequent burn treatment. A factor that may influence this different response to the fire-free interval could be light conditions. Although I did not measure canopy openness, total basal area for burn treatments were statistically different in 2015, with the frequent burn treatment having the lowest total basal area (Figure 2.1). This supports the notion that a significant reduction in basal area via a natural disturbance or mechanical manipulation may be necessary to facilitate any significant increase in oak regeneration (Brose et al., 2014). Another indication of higher light levels in the frequent burn treatment is that the stem density of hard pine regeneration was significantly higher in the frequent burn treatment compared to the less frequent. Also, hard pine saplings were only present in 4x treatment (93 stems/ha) in 2015 whereas they were absent in the less frequent and control treatments (data not shown). Although hard pine saplings were not included in this analysis because of a small sample size with many zero counts, the increase in hard pine sapling stems indicates light levels are high enough for these shade-intolerant species to establish and grow.

White pine saplings were virtually absent from burn plots after fire because of the extreme intolerance to fire of white pine stems <6cm DBH as demonstrated by prior research at these sites (Blankenship and Arthur, 1999). White pine develops thick bark when mature (Hengst and Dawson, 1994) so white pines that were large enough to resist fire damage likely persisted. Given that white pine lacks the ability to resprout after topkill and is extremely susceptible to fire damage as a small tree, repeated prescribed fire is effective at preventing white pine regeneration.

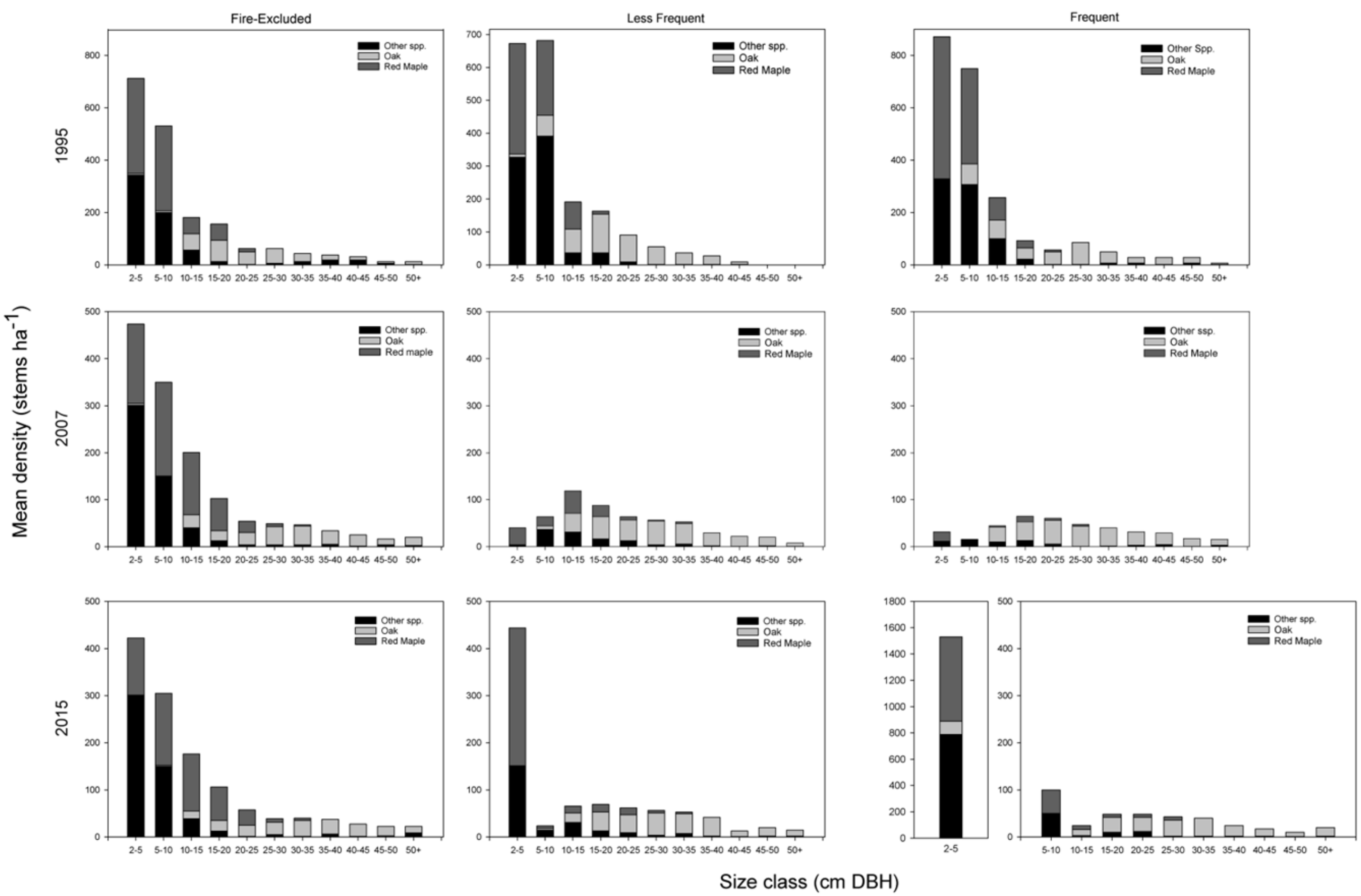
Fire-Excluded
Less Frequent
Frequent
1995
2007
2015
Other spp.
Oak
Red Maple
Mean density (stems ha-1)
Size class (cm DBH)
2-5
5-10
10-15
15-20
20-25
25-30
30-35
35-40
40-45
45-50
50+

Figure 2.6. Mean stem density (stems ha^{-1}) by size class beginning with 2-5cm DBH and then increasing by 5cm increments before burning (1995), after burning (2007) and after a fire-free interval (2015).

Burning resulted in stands with lower midstory stem density than the fire-excluded control and this remained significantly lower after the fire-free period. Within the midstory, burned stands lost red maple and oak stems whereas the fire-excluded control saw increases in midstory red maple, a trend that has been noted in another study in this region (Arthur et al., 2015).

Large red maple and other fire-intolerant species have survived repeated, low-intensity burning and have advanced into the overstory, further acting as a seed source for the regeneration pool in burned stands. However, fire-excluded sites have a higher density of overstory red maples than burned sites, indicating that the presence of fire on the landscape can reduce the number of mesophytic species reaching overstory status. This shows that even if oak regeneration is not improved due to burning, repeated burning can still help prevent the transition to a stand dominated by mesophytic species by thinning midstory red maple stems and limiting the recruitment of red maples into larger size-classes.

Prescribed fire and a fire-free interval may not be enough to put oaks at a competitive advantage if competitor species respond to the fire-free interval or survive to reach the overstory. Mechanical manipulation or more frequent or intense fires will likely be needed to further reduce overstory stem density and reduce competition from shade-tolerant species. Much effort and money would need to go into selectively killing red maple and other ecologically undesirable mesophytic species from the understory after burning due to sprouting responses. Red maple is known to sprout and be difficult to manage with herbicide (Hutchinson et al., 2016). It may be advantageous for land managers wanting to maximize the effect of prescribed fire to selectively kill targeted species by mechanical or chemical methods before applying fire in an effort to reduce the sprouting response of competitor species. Alternatively, repeated burning implemented for a longer period of time may be enough to reduce sprouts from competitor species. Prescribed fire did thin midstory red maples effectively, but these trees often created sprout clumps (pers. Obs.) which would be time consuming to kill with herbicide or mechanical means due to the high density of sprouts which is typical of red maple (Arthur et al., 2015; Blankenship and Arthur, 2006). Also, some trees of species which fire is applied to thin or control (e.g. red maple) have developed resistance to fire with size and were able to grow into the overstory size-class with the potential to be recruited in event of a canopy gap. Herbicide or selective thinning via harvest would benefit the restoration process by removing a seed source and simultaneously increasing light levels.

A fire-free period after repeated burning has potential to allow oak saplings and oak regeneration to increase in density. However, competitor species also responded positively to this period of no fire and can outnumber oak saplings. Our stands which burned repeatedly over 10 years still had species which are considered fire-intolerant advancing into the canopy layer (≥20cm DBH) indicating that much more effort and time may be needed to fully restore these fire-adapted oak ecosystems (See figure 6 in Nowacki and Abrams, 2008). However repeated burning was shown to reduce the densities of red maple reaching overstory status, highlighting the usefulness of using large-scale prescribed fire as a management tool to maintain oak dominance. Future studies should investigate the effects of differing fire return intervals and number of fires on the sprouting response of select species. Also, there is a need for applied studies which use selective thinning in conjunction with prescribed fire to restore oak ecosystems.

Chapter Three

Effects of wildfire severity on tree recruitment and stand structure in an oak-pine forest in the Red River Gorge, Kentucky

Introduction

Fire has been an important disturbance agent in the development of prehistoric, historic, and contemporary plant communities in the Eastern United States (Abrams, 1992; Delcourt and Delcourt, 1998), particularly in oak dominated systems (Nowacki and Abrams, 2015). Before European settlement, the number of fires and the fire return intervals were spatially and temporally variable and dependent on several factors including anthropogenic ignition, surface fuel production, fuel fragmentation, and cultural behavior (Guyette et al., 2002; Guyette et al., 2008). These fires ranged from low severity surface fires to stand replacing fires in years of drought (Wade et al., 2000). Today, fire regimes in this region are characterized by long fire return intervals due to fire suppression, typically occurring during the dormant season when weather conditions are dry (Lafon et al., 2005; Lafon and Quiring, 2012). Wildfires are heterogeneous in their severity, leaving a mosaic of unburned and burned patches. Despite the prevalence of wildfires in the eastern United States, there have not been many studies looking at the effects of varied wildfire severity on forest structure and tree recruitment. This is especially important when considering that wildfire in the Appalachian region has steadily increased over the past three decades (Lafon et al., 2005).

With climate warming, the probability of wildfire is predicted to increase in both frequency and extent (Guyette et al., 2014), yet there are few studies that have investigated vegetative recovery in the Appalachian region after wildfire. In addition, given the decline in regeneration success of fire-adapted species and the increase in fire-intolerant species throughout the eastern United States (Nowacki and Abrams, 2008), there's a need for increased understanding of the effects of varied severity wildfire on tree recruitment and overall forest structure. Specifically, it's important to understand how tree recruitment of fire-dependent tree species, such as oaks and pines, varies with wildfire severity. Also, it is useful to investigate tree recruitment patterns of competitor species in response to wildfire especially in a landscape which has experienced long-term fire suppression.

The Fish Trap Fire (FTF) was unintentionally ignited by campers in the Red River Gorge Geological Area in the Daniel Boone National Forest, Kentucky in Powell County on 24 October 2010 and was fully contained on 9 November 2010. The containment perimeter covers 674 ha of mostly public land. This presented a unique opportunity to study vegetation development after wildfire on public land. This research utilizes permanent vegetation plots established in 2011, one year after the FTF. Using prior assessments of fire severity based on Landsat data and the change in normalized burn ratio (Upadhaya, 2015), I used linear regression to examine the relationships between burn severity and forest vegetation response six years after the burn. I hypothesized that fire severity would be negatively related to stand density and stand basal area and a positively related with recruitment of species which depend on high light availability to regenerate (oaks and pines). Finally, I hypothesized that burn severity would have a negative relationship with the recruitment of fire-intolerant competitors including red maple, sourwood, and blackgum. Of the limited number of wildfire studies in the eastern U.S., there are no studies which relate fire severity and stand structure or tree recruitment at a wildfire site.

Methods

Site Description:

The landscape of the Red River Gorge is characterized by highly dissected uplands in the Cliff Section of the Cumberland Plateau (Braun, 1950). The fire was largely restricted to the xeric upland forest though it also spread down lower slopes (Upadhaya, 2015). Ridges in this region are dominated by oaks such as scarlet oak (*Quercus coccinea*) and chestnut oak (*Q. montana*), along with white oak (*Q. alba)* and black oak (*Q. velutina.*). Hard pines, shortleaf (*Pinus echinata*), pitch (*P. rigida*), and Virginia (*P. virginiana),* are also common overstory components. Hickories (*Carya* spp.) are also present in the overstory.

Elevations range from 177 m a.s.l. to 439 m a.s.l. The soils are primarily Alticrest-Ramsey on the ridges and Helechawa on midslopes whereas lower slopes are classified as Bledsoe-Berks. The underlying geology is composed of Pennsylvanian sandstones and conglomerates and shales of the lower Breathitt formation (Hayes, 1993). Mean annual precipitation is 130cm, with a mean growing season of 176 days. Mean annual temperature is 12° C, with mean daily temperatures ranging from 0° C in January to 31° C in July (Foster and Conner, 2001).

Data Collection:

Thirty 500 m^2 circular plots were installed in the burned area of FTF in August 2011 for post-fire data collection. Plot locations were selected using a stratified random approach to capture the ecological effects across burn severities throughout the entire burned area. Plot centers were permanently marked with rebar and GPS coordinates were taken for each point. For all stems ≥2 cm diameter at breast height (DBH), species, number of basal sprouts, and number of browsed sprouts were recorded. Within a 25 m^2 circular plot arranged around plot center, stems <2 cm DBH were counted, identified to species, and then recorded as either small seedlings (<50 cm height) or large seedlings (≥50 cm height). Only plots which were evaluated as upland sites dominated by upland tree species (e.g. *Quercus coccinea*, *Q. montana*, *Pinus echinata*, and *P. rigida*) were used for analysis which resulted in losing 3 plots. An additional plot was also not sampled in 2016 due to campers camping in the plot. A final plot was excluded from analysis because the sapling layer was not sampled in 2011 resulting in a total sample size of 25 plots.

Normalized burn ratio (NBR) was developed by Key and Benson (2003) using Landsat data and near infrared and mid-infrared bands. The change in NBR values is commonly used in the literature to characterize burn severity and forest structure (White et al., 1996). To calculate dNBR, normalized burn ratio (NBR) values were calculated for the study area the year before the fire and after the fire and then the change was calculated. This created a continuous variable as a surrogate for fire severity.

Data Analysis:

Burn severity, as defined by dNBR (Upadhaya, 2015), was used to examine the relationship between burn severity and forest vegetation response. Simple linear regression was used to model vegetation response variables estimated from vegetative data and dNBR values after assessing model assumptions using SAS (linearity, homogeneity, and normally distributed residuals). Recruitment between 2011 and 2016 for various species or species groups including oak species (*Quercus* spp.), pine species (*Pinus* spp.), red maple (*Acer rubrum*), sourwood (*Oxydendrum arboreum)*, and blackgum (*Nyssa sylvatica*) was calculated. Recruitment was calculated as the number of sapling layer (2-10cm DBH) stems counted in 2011 subtracted from the number of stems in 2016. Other response variables include 2016 total basal (all stems ≥2cm DBH) and 2016 stem density (stems ≥10cm DBH).

Results

Stem density (stems ≥10cm DBH) was significantly and negatively related to burn severity (Figure 3.1; p=0.0003, F=18.15, R^2=0.43). The total basal area (stems ≥2cm DBH) had a significantly negative relationship to burn severity (Figure 3.1; p= <0.0001, F=33.23, R^2=0.58).

Net recruitment of oak saplings (2-10cm DBH) from 2011 to 2016 was significantly and positively related to dNBR values (Figure 3.2; p<0.0009, F value: 14.34m R^2 = 0.37). Similarly, net recruitment of pine saplings was significantly and positively related to burn severity (Figure 3.2; p = 0.0002, F = 19.90, R^2 = 0.45). Conversely, red maple (p = 0.6015, F= 0.28, R^2 = 0.01), sourwood (p = 0.5946, F = 0.29,

$R^2 = 0.01$), and blackgum ($p = 0.2686$, $F = 1.29$, $R^2 = 0.05$) sapling recruitment did not vary significantly with burn severity (Figure 3.2).

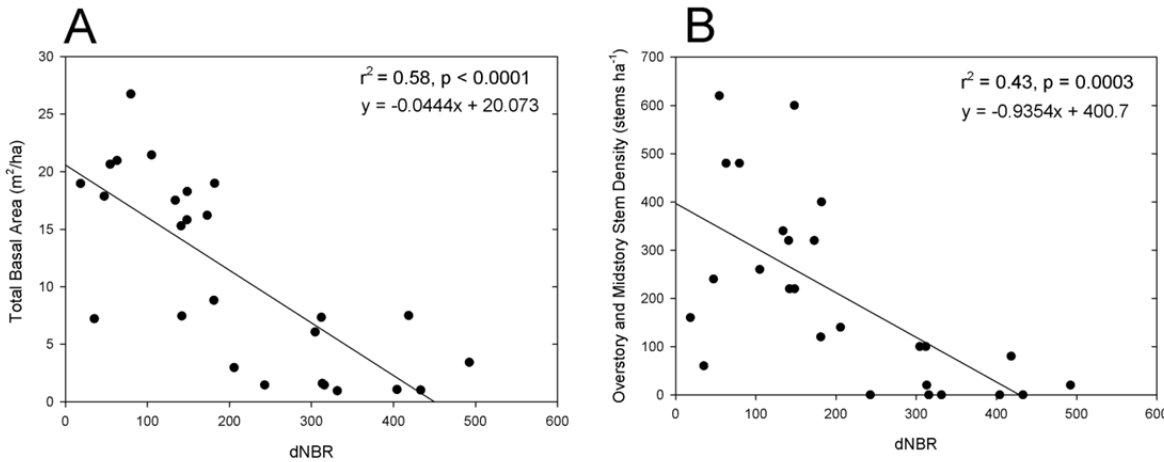

Figure 3.1. Linear regression analysis with 2016 basal area (m^2/ha) (A) and 2016 stem density (stems ha^{-1}) (B) as dependent variables and dNBR values as the independent variable (a proxy for fire severity). Stem density is represented by the stem density of all stems ≥10 cm DBH.

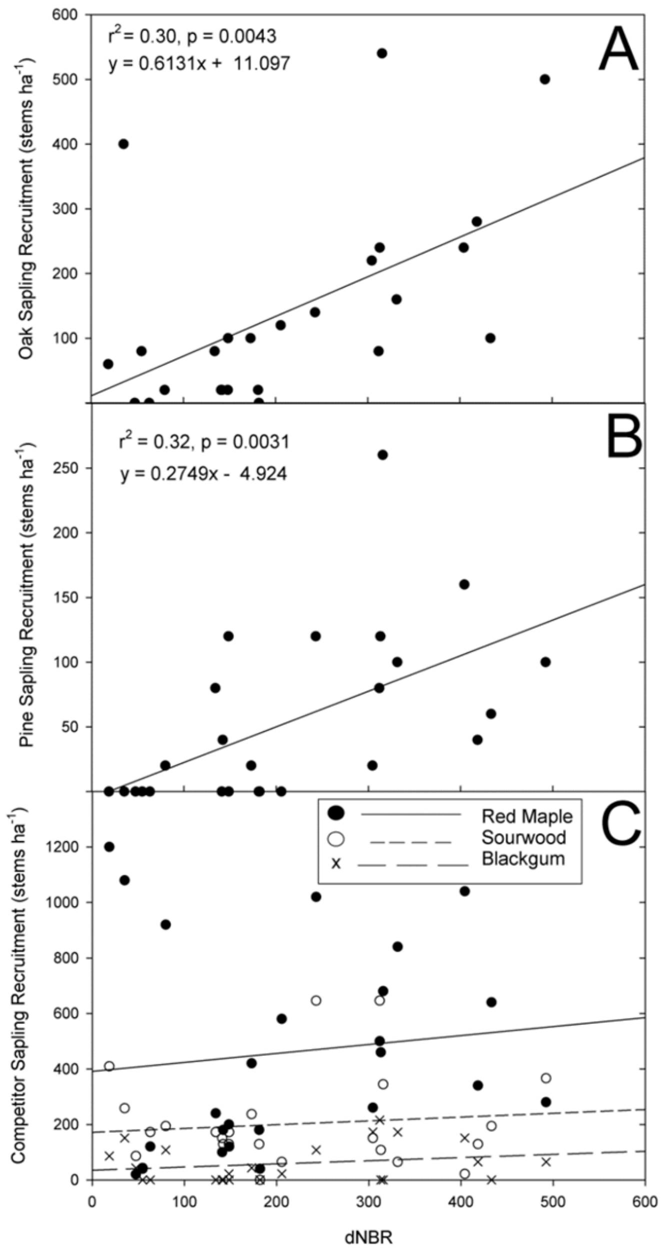

Figure 3.2. Linear regression analysis with oak sapling recruitment (A), pine sapling recruitment (B) and competitor sapling recruitment (C) from 2011 to 2016 as dependent variables and dNBR values (a proxy for fire severity) as the independent variable. Regression equations, p-values, and R^2 values are only shown if the regression analysis was significant ($\alpha = 0.05$). Oaks include all species in the genus *Quercus* L. and pines include all species in the genus *Pinus* L.

Discussion

The results of this study show that wildfire severity (dNBR values) can significantly predict the recruitment of two fire-dependent species groups as well as stand structure and total basal area. The positive relationship between oak sapling net recruitment and burn severity is likely due to increasing severity wildfire creating the open environment where oak species can be competitive and grow to this size class. Pine sapling net recruitment also shows this positive relationship with fire severity. Areas which burned with high severity can create conditions which are favorable for pine seedling germination, establishment, and growth such as mineral soil exposure and high light levels (Burns and Honkala, 1990).

In contrast to oak and pine recruitment, red maple, sourwood, and blackgum sapling recruitment showed no significant relationship with fire severity which did not support my hypothesis of decreased recruitment with increasing fire severity. Red maple is a competitor to oak and other fire adapted species in this region (Fei et al., 2011). Even with its relative fire-intolerance, it is a good competitor with the ability to sprout prolifically after fire, take advantage of canopy disturbance, reach sexual maturity quickly, and produce a tremendous number of propagules each year (Burns and Honkala, 1990; Blankenship and Arthur, 2006). Sourwood and blackgum can also sprout prolifically, are tolerant of shade, and can respond positively to increased light levels (Burns and Honkala, 1990). Although there is literature which suggests repeated prescribed fire can initially reduce shade-tolerant competitors (Blankenship and Arthur, 2006), this single, varied intensity wildfire was not enough to eliminate the group from this upland environment. Some larger competitor trees may have survived fire where severity was low thus serving as a seed source for the surrounding burned landscape. Future fire in this area could further impact surviving competitors and act as an environmental filter for future tree recruitment. However, at this stage of recovery, red maple is still a key competitor to oaks even in high light environments which should favor oaks and other fire-adapted species. If fire-adapted ecosystems reach a state in which oak and pine regeneration is rare or absent, varied severity wildfire may hasten the transition to forests dominated by shade-tolerant species in the understory. Indeed, in 2016, mean red maple sapling stem density was 491 stems/ha (S.E. ± 73 stems/ha), sourwood was 203 stems/ha (S.E. ± 34 stems/ha), and blackgum was 73 stems/ha (S.E. ± 12 stems/ha) compared to 142 stems/ha (S.E. ± 30 stems/ha) for oaks and 155 stems/ha (S.E. ±74 stems/ha) for pines. Because I did not measure the competitive status of saplings in this study, it is unclear whether oaks and pines will out-compete established competitors and be recruited to overstory positions. However, when plots were classified based upon burn severity classes (Updahaya, 2015), the relative density of red maple is much lower in high severity plots (38%) and moderate severity plots (37%) when compared to lower severity plots (63%) as shown in Figure 3.3. High and moderate severity plots have higher relative density of oaks (16% and 11% respectively) when compared to low severity plots (2%). Likewise, the relative density of pines is higher in high and moderate severity plots (8% for both) when compared to the low severity plots (0.5%). Low severity burn plots are similar in relative density to that of prescribed fire plots (see Chapter Two) whereas both moderate and high severity burn plots have much more oak and pine saplings. Still, it should be noted that moderate and high severity burn plots also have a larger relative density of “other” species which are mainly composed of early-successional species such as sassafras, black locust, and big toothed aspen (Figure 3.3).

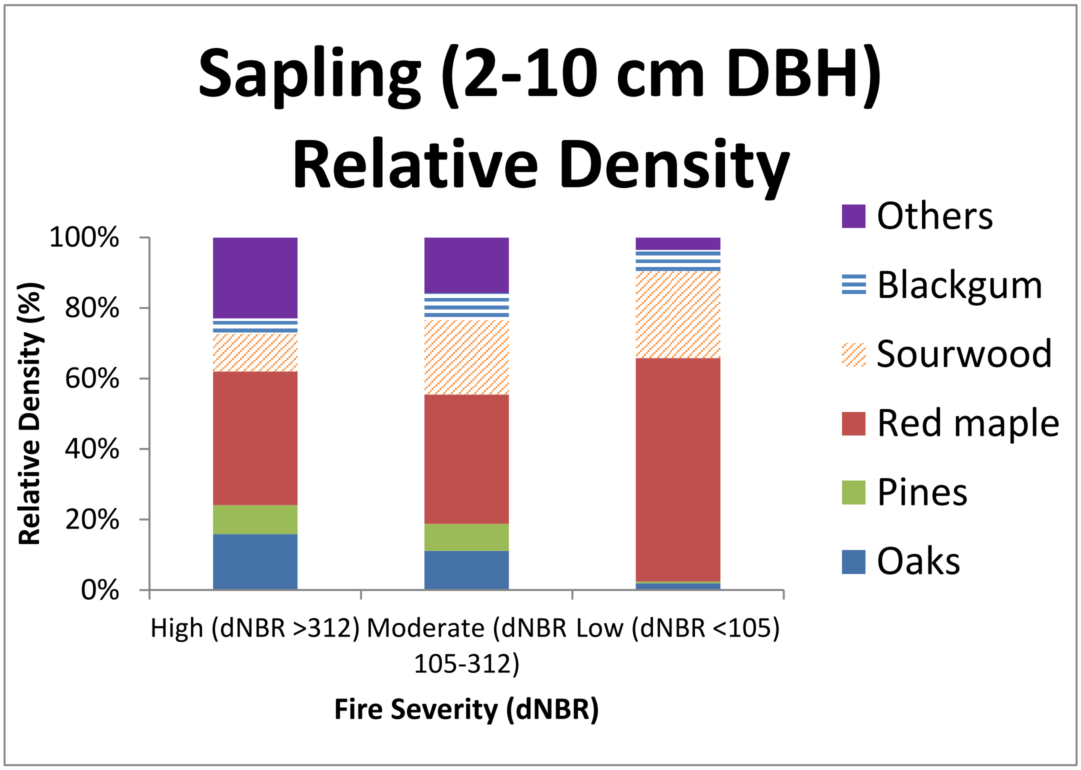

Figure 3.3. Relative density of sapling layer (2-10cm DBH) stems across several burn severity categories based on Updahaya (2015). Relative density was calculated as the sum of all stems for that species or species group divided by the total number of stems for each severity class. Other species include *Sassfras albidum*, *Robinia pseudoacacia*, *Rhus copallina*, *Populus grandidentata*, *Juglans nigra*, and *Liriodendron tulipifera*.

Although research involving wildfire in the Appalachian region is sparse, there are some studies which have gleaned useful information regarding effects of wildfire on vegetation. A study in West Virginia found that areas with high frequency and recent wildfire had higher densities of oak saplings (Thomas-Van Gundy et al., 2015). However, competitor saplings, such as red maple, benefited from infrequent and recent fire whereas their seedlings benefited when there was no fire or infrequent fire, indicating that frequent fire is needed for oak-pine dominated systems to persist. My data show that this single, varied intensity wildfire (an infrequent event in this area) has promoted the recruitment of oak saplings. However, without recurring fire, it is unlikely that key competitors to oak will be out competed being that they may have been established in the understory before the fire and have since recruited across plots.

A study on wildfire in the Appalachians of southern Pennsylvania showed that red maple saplings were absent from burned areas whereas oak saplings dominated (Signell et al., 2005). One key finding of that study was that overstory stem density greater than 400 stems/ha and understory stem density greater than 200 stems/ha can limit oak regeneration. In this research, 66% of plots with 0 stems/ha net recruitment of oak saplings all meet or exceed overstory stem density threshold (n=3). The only plot that did not meet this threshold and still had no recruitment was a plot in an unburned area of the landscape.

Significant relationships were also found for stem density and total basal area (Figure 3.1). In the Southern Appalachians, Hagan et al. (2015) found that wildfires resulted in the mortality of large overstory trees leading to increased sapling stem density which the authors attributed to the mortality of large overstory trees, vigorous resprouting, and the establishment of new individuals. Oaks increased in density and this increase was larger where plots burned twice. Conversely, oak densities decreased in plots which haven't burned over the same time period. The negative relationship of fire severity and stand density and total basal area that I present here likely lead to the recruitment of fire-dependent species such as oaks and pines in our study area. This is useful for managers who may be tasked with restoring or managing oak-pine forests. Resources could be focused on areas which burned at low severity to further reduce stand density by mechanical manipulation or herbicide treatment. Conversely, areas which burned at high severity after years of no fire may be targeted to control competing vegetation and favor oak and pine regeneration. Areas which have been severely burned may also harbor pioneer species which are non-native, invasive plants such as royal paulownia (*Paulownia tomentosa)* (Upadhaya, 2015). It would be useful to allocate resources to these areas to manage invasive species before they become established and reproduce.

This study shows that there is a positive relationship between fire severity and oak and pine recruitment, but there is no relationship between fire severity and a key competitor, red maple. Six years after the occurrence of a wildfire, red maple saplings outnumber oaks and pines, but it is unclear whether this pattern will persist as time goes on. Without additional burning at the Fish Trap Fire site to control fire-intolerant vegetation, red maple, and other competitors, may take advantage of increased light levels and out-compete oaks and pines. When fire was once much more prevalent in the landscape, recurring fire likely kept mesophytic species at low densities in oak-pine forests except at protected sites (Lafon et al., 2017). However, once fire suppression took place, these species became established in areas which used to burn, thus the effects of high severity wildfire on forest composition may have changed in response to the changing forest composition. I also found a significant negative relationship between total basal area and stem density (Figure 3.1). This relationship could prove to be useful for managers in this region which intend to use remotely sensed data (as a metric for fire severity) to evaluate forest structure and then plan restoration or silvicultural treatments.

Appendix A

Table of p-values from repeated measures ANOVA for the overstory, midstory, and sapling layers. P-values are missing when that species/species group was not tested due to excessive zeros in the data set. P-values in bold are significant at α = 0.05.

Factor	df	Total	Oak	Red Maple	White Pine	Sourwood	Blackgum
		Overstory stem density (stems ha^{-1})					
Site	1	**0.0167**	**0.0002**	0.9098			
Treatment	2	0.8404	0.517	**0.0146**			
Year	2	0.7879	0.1466	**< 0.0001**			
Trt*Year	4	0.1096	**0.0159**	0.3026			
		Midstory stem density (stems ha$^{-1)}$					
Site	1	0.2303	0.5642	0.2812	0.0369		
Treatment	2	**<0.0001**	**0.0481**	**<0.0001**	0.6388		
Year	2	**<0.0001**	**<0.0001**	**0.0183**	0.9396		
Trt*Year	4	**0.0054**	0.8916	**0.0034**	**0.0188**		
		Sapling stem density (stems ha^{-1})					
Site	1	0.0212	0.0535	0.2036	0.874	0.612	0.8326
Treatment	2	0.0025	**0.0073**	0.0247	**<0.0001**	**0.0025**	0.6119
Year	2	**<0.0001**	**0.002**	**<0.0001**	0.0654	**<0.0001**	**<0.0001**
Trt*Year	4	**<0.0001**	**0.001**	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**

Appendix B

Table of p-values from repeated measures ANOVA for regeneration density (stems <2cm DBH) and total basal area (all stems ≥2 cm DBH). P-values in bold are significant at $\alpha = 0.05$.

Factor	df	Total	Oak	Hard Pines	Red Maple	White Pine
		Regeneration stem density (stems ha^{-1})				
Site	1	0.8844	0.5615	0.2356	**<0.0001**	0.0601
Treatment	2	0.4358	**0.0331**	**0.0084**	0.4889	0.3169
Year	2	**<0.0001**	**<0.0001**	**0.0003**	**<0.0001**	**<0.0001**
Trt*Year	4	0.6305	0.221	**0.0008**	0.1996	0.3739
		Basal Area (m^2/ha)				
Site	1	0.2303				
Treatment	2	**0.0088**				
Year	2	0.1282				
Trt*Year	4	**0.0077**				

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Vita

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