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Fire and Vegetation Dynamics in the Southern Appalachian Mountains

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FIRE AND VEGETATION DYNAMICS IN THE SOUTHERN APPALACHIAN MOUNTAINS

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by
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ABSTRACT

Prescribed burn regimes in the forests of the southern Appalachian Mountains have been a topic of research since the revival of fire management in the United states. The presented two studies address two important topics: the viability of long term dormant season burn regimes to reaching management goals, and how the seasonality of prescribed burning may influence sprouting dynamics of target species.

The Fire and Fire Surrogate Site at Green River Game Land, Polk County, NC, provided forest structure and compositional data both before and after a 15-year, periodic dormant season burn regime. We found a significant decrease in smaller size class trees after the regime. Secondly, basal area of non-desirable mesophytic hardwoods, such as red maple (*Acer rubrum*), mountain laurel (*Kalmia latifolia*), rhododendron (*Rhododendron spp.*), and blackgums (*Nyssa sylvatica*), showed a significant decrease as a result of the burn regime. These results imply that if the current regime is to continue, it may result in less mesic, more fire-conducive forests. However, overstory oak abundance also declined significantly during the study period. The abundance of oaks in the midstory did increase in the burn treatment, but the difference was not statistically different from the control. Although periodic dormant season burning may help eliminate mesophytic tree species, future overstory composition may not be comparable to historical conditions. With American chestnut trees absent from the overstory, and hemlocks in decline, it may be unreasonable to think that restoration to historical composition is possible.

The results of the seasonality experiment show differences in the sprouting of desirable and undesirable species by month of burn. An undesirable species, red maples, and a desirable species, shortleaf pines (*Pinus echinata*), show nearly inverse sprouting trends throughout the year. This could mean the timing of prescribed fires may favor management objectives that eliminate red maples from a site, while simultaneously promoting the growth and regeneration of shortleaf pines. The mean sprout biomass of mountain laurel, a common species targeted for control in the southern Appalachian Mountains, was significantly less when burned in September compared to all other months, but was significantly higher than all other months in October and December. This result suggests there may be an optimum time for prescribed fire if an objective is control of mountain laurel.

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CHAPTER ONE

OVERSTORY AND MIDSTORY RESPONSES TO 15 YEARS OF PERIODIC DORMANT SEASON BURNING IN THE SOUTHERN APPALACHIAN MOUNTAINS

Introduction

Disturbances are known to influence vegetation composition and assembly on large and small scales (He and Mladenoff 1999). In many regions of the United States, including the southeast, fire has been one of the most influential disturbances in shaping many plant communities (Delcourt and Delcourt 1997). The current composition and structure of southeastern forests can be attributed, at least in part, to the role of fire during four significant periods: the pre-Columbian era (earlier than 1700's), European settlement era (1700's- early 1900's), fire exclusion era (1930's – late 1900's), and the fire management era (late 1900's- present day) (Stanturf et. al 2002, Lafon et al. 2017).

Fire has long occurred in the southern Appalachian Mountains, whether anthropogenically or naturally ignited, but fires during these periods had different uses and values (Stanturf et al. 2002, Lafon et al. 2017). During the pre-Columbian period, Native Americans started fires for many purposes such as clearing understory, improving habitat for hunting, and pest control (Delcourt and Delcourt 1997). Early Europeans adopted many of the land management techniques of the native Americans, including fire. The settlers used fire mainly for clearing agriculture such as crop land clearings in the flat lands and clearing for open grazing in the hilltops and other sites not suited for crops (Williams 1992, Stanturf et al. 2002).

Fires in the southern Appalachian landscape burned frequently and sometimes across very large areas (Lafon et al. 2017). Because of landscape heterogeneity, some sites are more prone to fire than others. As fire passes through a region, the unequal fire behavior between sites creates a mosaic of early and late successional plant communities (Lafon et al. 2017). Fire continued its role across the southeastern montane landscape throughout the extensive industrial logging episode of the late 19th and early 20th century (Stanturf et al. 2002). Forest fires abruptly ended during the fire suppression era, consequently changing the successional trajectory of southern Appalachian forests (Lafon et al. 2017). After a series of devastating fires in the late 1800's to the early 1900's, the United States Forest Service began to adopt new policies, such as the 10 o'clock policy (1936), which required the immediate suppression of all fire, both human-ignited and natural, before 10:00 a.m. the next morning (Brose et al. 2001, Lafon et al. 2017). This, coupled with earlier widespread exploitative logging (Lafon et al. 2017) and biotic factors such as the chestnut blight (Ellison et al. 2005), had significant ramifications on the successional trajectory that has shaped plant communities of the southern Appalachian Mountains (Nowacki and Abrams 2008, Lafon et al. 2017).

Without fire to control the recruitment of thin-barked and smaller diameter trees, many of the previously fire-maintained forests and woodlands were converted to closed-canopy mesic forests, while allowing new species to take over the midstory (Nowacki and Abrams 2008, Lafon et al. 2017). Shade-tolerant, fire-sensitive species such as birches (*Betula* spp.), blackgums (*Nyssa* spp.), yellow-poplars (*Lireodenderon tulipifera*), and maples (*Acer* spp.) (Nowacki and Abrams 2008, Lafon et al. 2017) established in the midstory, resulting in a stratified forest in which the overstory was recruited by one fire regime and the midstory by another fire regime (Lafon et al. 2017). The heliophytic species, such as the yellow pines, and the fire tolerant species, such as oaks, are predicted to slowly diminish and be replaced by the new mesophytic species from the midstory unless restoration efforts are enacted (Nowacki and Abrams 2008, Lafon et al. 2017). Additionally, the closing of the canopy altered the abiotic environment on the forest floor from a bright, dry environment to a more shaded and humid one that was more conducive to the germination and establishment of those same mesophytic hardwoods and less conducive to fire. Nowacki and Abrams (2008) referred to this phenomenon as "mesophication".

The fire management era is so named due to the use of prescribed fires to meet forest management objectives (Stanturf et al. 2002). In the southern Appalachian Mountains, some of the main objectives of prescribed fire include restoration to historic open woodland structure, altered community composition in favor of target species, and hazardous fuel reduction as a means to prevent widespread devastating wildfires (Waldrop et al. 2016). However, since the push to integrate fire back into the ecosystem in the 1980's (Lafon et al. 2017), very few long-term studies have been conducted to assess if fire can be used to successfully break the mesophication feedback process.

To fill the gap in information on the long-term effects of prescribed fire on mesophication, we analyzed a 15-year dataset from the National Fire and Fire Surrogate Study site in western North Carolina (Waldrop et al. 2016). This study provides insight on the effects of a periodic (3-6-year return interval) dormant season burn regime in the southern Appalachian Mountains. In it, we examine changes in diameter distributions and overstory composition over time. We also examine the midstory dynamics in burned and unburned treatments by assessing changes in the composition, abundance, and recruitment of midstory trees into the overstory. Overall, it will help provide evidence as to whether these fire management strategies are effective at counteracting mesophication and if restoration to open woodland conditions in the region is an obtainable objective.

We hypothesize that fire intolerant mesophytic hardwoods will have decreased in stem density and basal area in the overstory throughout the burn treatment period, relative to an unburned control, with oaks and fire tolerant pines being least affected by the regime. A second hypothesis is that trees in the smaller size classes will decrease in number and basal area coverage due to burning because they are more susceptible to damage.

Methods

The study site is located in the Green River Game Land in Polk County, North Carolina (Figure 1.1). The Game Land is managed by the North Carolina Wildlife Resources Commission for wildlife habitat, timber, and other resources. Elevations range from 350 m to 750 m. Forest composition was mixed-oak, with yellow pines (Virginia pine [*Pinus virginiana*], pitch pine [*Pinus rigida*], shortleaf pine [*Pinus echinata*, and Table Mountain pine [*Pinus pungens*]) on xeric ridges and eastern white pine (*Pinus strobus*), blackgums, red maples, and ericaceous shrubs predominant in moist coves. Soils are primarily of the Evard series, which are moderately deep, well-drained mountain upland soils (Keenan 1998, Waldrop et al. 2016).

As a part of the National Fire and Fire Surrogate Study (NFFSS) established by the Joint Fire Science Program in 2000, study areas were chosen to represent different regions across the United States. For the southern Appalachian region, portions of the Green River Game Land were designated to be subjected to the NFFSS protocols. The NFFSS site in Green River included four treatments: burn (B), mechanical (M), mechanical and burn (MB) combination, and a control (C). Three replicates with each of the four treatment areas (Figure 1.1) were each 10-12 ha in size. Buffers for each treatment were 20 m wide. The treatments were prescribed with the intention of including the variety of topography that would be sufficiently representative of the region. Each treatment received a 50m x 50m grid overlay with 36 - 40 marked grid points for data sampling, ten of which (for each treatment) were randomly selected and used as points of origins for 0.1-ha (20 x 50 m) grid point plots for vegetation data collection.

Vegetation sampling was performed on the 0.1-ha grid point plots that were each 50 x 20 m in size and divided into 10, 10 x 10 m, subplots. All trees on one side of the 50 x 20 m plots (five plots) 10 cm at DBH or larger were measured. The data collected for each tree included diameter at breast height (DBH: diameter at 1.4 meters above the ground), species, and status (live or dead), along with a unique tree number. Percent shrub cover data (all mountain laurel [*Kalmia latifolia*], rhododendron [*Rhododendron* spp.] and herbaceous plants taller than 1.4 m) were excluded from this analysis. All tree/sapling species greater than 1.4 m tall and less than 10 cm in diameter were categorized into

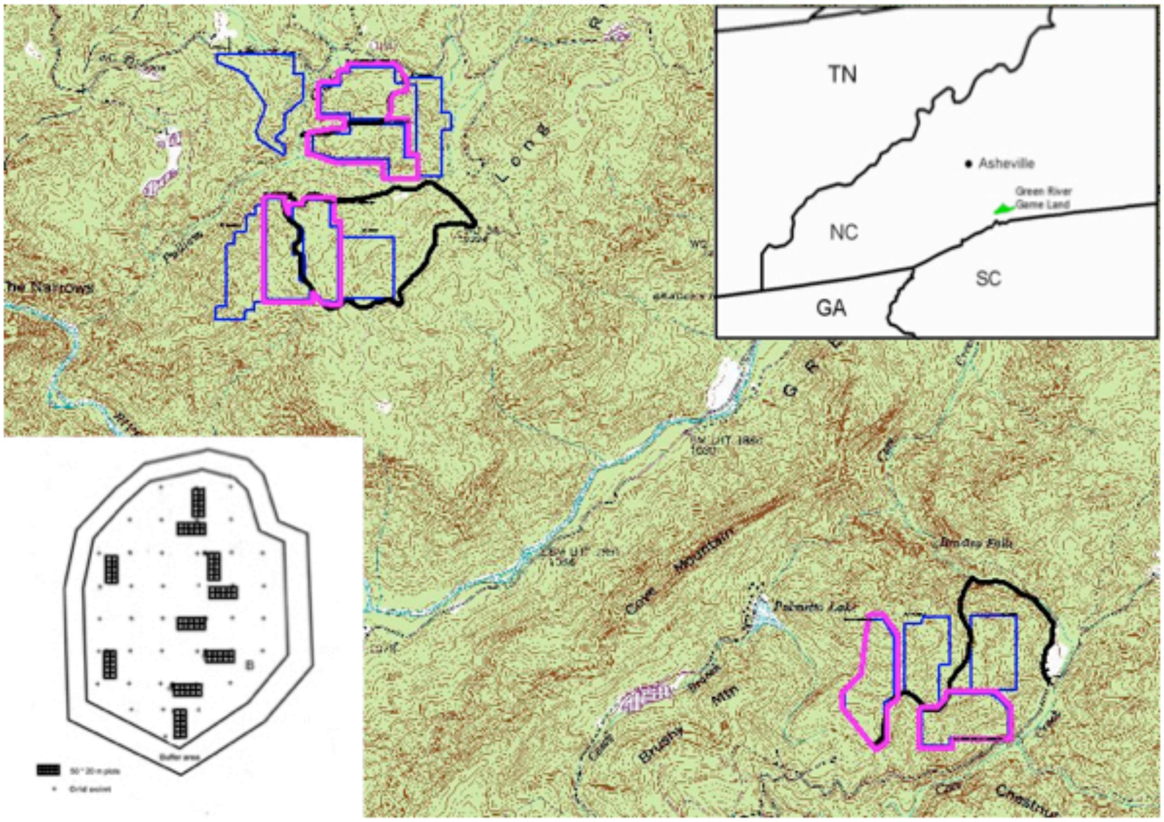

Figure 1.1. B treatments are outlined by black lines representing fire breaks. C treatments are outlined in blue. M treatments are outlined in purple and MB treatments are overlapping areas bordered by both black and purple lines. The M and MB data were not included in this report. Inset in the top right shows the location of the Green River Game Land in Polk County, NC. Inset in bottom left shows an example of one treatment area with ten grid-points.

DBH size classes: 0-3 cm, 3-6 cm, and 6-10 cm. The stem count of each species in each size class was recorded in the five subplots.

Burns were conducted during the dormant season in years 2 (2003), 5 (2006), 11 (2012), and 15 (2016). Pre-treatment data (year 0) were collected during the summer of 2001, and the last round of post-treatment data was collected in summer of 2016 (year 15). This pre- and post-treatment dataset was used to examine the burn regime as a whole to build on the previously reported data which studied annual trends of the four treatments (Waldrop et al. 2016). For this examination, the data from 2001 and 2016 in treatments B and C were isolated, examined, and compared to assess the impact of the dormant season burn regime.

Survival analysis was performed to determine which species had an average overall increase (winners) and decrease (losers) in the overstory between treatments. For analysis purposes, the species were placed into one of five groups of management interest: mesophytic hardwoods (MSHW), hardwoods (HW), oaks (OAK), yellow pines (YELLPINE), and white pines (WHITEPINE). All hemlock species (*Tsuga spp.*) were omitted from the analysis as mortality and/or decline of that species is related to the invasion of the hemlock wooly adelgid (Alman et al. 1965) and would potentially confound our treatment results. The HW species group included hickory (*Carya spp.*), flowering dogwood (*Cornus florida*), sourwood (*Oxydendrum arboreum*), basswood (*Tilia americana*), Carolina silverbell (*Halesia tetraptera*), sassafras (*Sassafras albidum*), fraser magnolia (*Magnolia frasieri*), black locust (*Robinia psuedoacacia*) and unspecified species listed simply as hardwoods. The MSHW group is based off of the species listed in Nowacki and Abrams (2008). These species included all birches (*Betula* spp.), blackgums (*Nyssa sylvatica*), yellow-poplar (*Lireodendron tulipifera*), American beech (*Fagus grandifolia*), black cherry (*Prunus* spp.), and maples (*Acer* spp.). The OAK group encompassed all *Quercus* spp. The YELLPINE group consisted of shortleaf pine, Table Mountain pine, Virginia pine (*Pinus virginiana*), and pitch pine. The same analysis was used to determine which species are the drivers of compositional and structural changes. In the midstory survival analysis by species, all pines (WHITEPINE and YELLPINE) were grouped together because of the lack of yellow pines in the midstory.

The change (pre-treatment: year 0, to post treatment: year 15) in diameter distribution (trees/ha in each size class) of the overstory was analyzed to determine which size classes had significantly increased or decreased throughout the treatment period compared to the other treatment (B or C). Overstory trees were categorized into size classes 1, 2, and 3 (10 cm < DBH < 25 cm, 25 cm < DBH < 45 cm, and DBH greater than 45 cm respectively).

Forest structure effects were analyzed by examining basal area at the 0.1 ha-plot level. To further the analysis, basal area for each size class was analyzed individually to determine what size classes exhibited the most significant changes in basal area. The basal area per hectare represented by each size class was calculated for sample plot in the B and C treatments. Additionally, basal area for each species or group of interest was examined to determine long-term effects on the plant communities.

All data were managed and analyzed using JMP Pro 13.2 (SAS institute, Cary NC). The fit model function was used to analyze the change of basal area (delta) in treatment B compared to the deltas of treatment C (alpha = 0.05). Treatment was given a fixed effect and replicates as well as grid points nested in replicates were given random effects. This analysis assessed the effects of the treatment on the changes in structure and composition of the forests at the 0.1 ha-plot level.

Results

The survival analysis shown in tables 1.1 and 1.2 provides insight to the recruitment of midstory trees to the overstory and species turnover in the overstory. MSHWs in the overstory increased by 15.4 trees/ha on average in the C treatment but decreased by roughly 43 trees/ha on average as a result of the B treatment (Table 1.1; p = 0.0189). The C treatment showed a net decrease in MSHW in the midstory by 504 trees/ha compared to an increase of 548 trees/ha in the B treatment (Table 1.2; p = 0.0035). The B treatment shows a net increase of MSHWs in the midstory (Table 1.2), but an overall decrease in the overstory (Table 1.1). Overstory trees in the OAK group responded to the regime, decreasing significantly more (~ 61 trees/ha) in the B treatment than in the C treatment (decreased by roughly 30 trees/ha; p = 0.0310). OAKs showed no net increase in the midstory C treatment and an increase by 202 trees/ha in the B treatment (Table 1.2; p = 0.1643). In the B treatment, oaks showed a net increase in the midstory by 202 stems/ha but an overall net decrease in the overstory. HWDs in the overstory decreased by roughly 15 trees/ha in the C treatment and decreased significantly more (~ 30 trees/ha) in the B treatment (Table 1.1; p = 0.0111). The HWD group in the midstory responded dramatically to the B treatment where the mean abundance decreased by nearly 85 trees/ha in the C treatment, but increased by roughly 274 trees/ha in the B treatment (Table 1.2; p = 0.0006). YELLPINEs did not respond to the B treatment in the overstory where they decreased by 38.9 and 37.4 in the B and C treatments, respectively (Table 1.1: $p = 0.9392$). The WHITEPINE group however, did respond to the B treatment, where their mean abundance increased by 2.1 trees/ha compared to almost 18 trees/ha in the C treatment (Table 1.1; $p = 0.0008$). The YELLPINE and WHITEPINE groups were lumped into a single PINE group in the midstory because of the lack of YELLPINEs in the midstory. This is likely due to the advanced seral stages in the majority of the sample plots. In the midstory, the PINE group had a very high response to the B treatment where stem counts decreased by nearly 153 trees/ha compared to the small increase around 8 trees/ha in the C treatment (Table 1.2; $p = <0.0001$). Most of this change was likely driven by eastern white pine's disproportional presence in the midstory.

Table 1.1: Overstory survival table. The average difference of trees/ha in the overstory (10 cm < DBH) between year 0 and 15 between B and C treatments and the associated p-value of the difference between treatments (alpha = 0.05) for the overstory trees by group in the Green River Game land, Polk County, NC.

Mean Overstory Survival			
Species Group	ΔC	ΔB	P-value
MSPHW	15.4	-42.67	0.0189*
OAK	-29.5	-61.33	0.0310*
HWD	-15.17	-29.87	0.0111*
YELLPINE	-38.9	-37.4	0.9392
WHITEPINE	17.5	2.1	0.0008*

ΔC= Change in mean stem counts of overstory trees from year 0 to year 15 in the control treatments represented as trees per hectare
ΔB= Change in mean stem counts of overstory trees from year 0 to year 15 in the burn treatments represented as trees per hectare
* = Indicates that change in mean stem counts for associated species group is significantly different between B and C treatment

Table 1.2: Midstory survival table. The average difference of trees/ha in the midstory (0 cm <DBH < 9.9 cm) between year 0 and 15 between the B and C treatments and the associated p-value of the difference between treatments (alpha = 0.05) for the midstory trees by group in the Green River Game land, Polk County, NC.

Mean Midstory Survival			
Species Group	ΔC	ΔB	P-value
MSPHW	-504	548	0.0035*
OAK	0	202	0.1643
HWD	-84.67	274.1	0.0006*
PINE	7.7	-152.67	<0.0001*

ΔC= Change in mean stem counts of midstory trees from year 0 to year 15 in the control treatments represented as trees per hectare
ΔB= Change in mean stem counts of midstory trees from year 0 to year 15 in the burn treatments represented as trees per hectare
* = Indicates that change in mean stem counts for associated species group is significantly different between burn and control treatments

Figure 1.2 shows the diameter distribution by size class for the overstory of the burn and control treatments in year 0 and year 15 (pre- and post-treatment) represented as trees/ha by size class. Both pre- and post-treatment diameter distributions show a 'reverse-J' shape, but the post-treatment datum appears to be flattening out due to the decrease in the number of trees/ha in size class 1 (DBH < 10 cm). Effects of the burn regime on size class 1 were the greatest with decrease treatment B by 5,420 stems/ha and the decrease by 1,860 stems/ha in treatment C (Table 1.2: $p = 0.0218$). The number of stems in size class 2 (10 cm > DBH < 25 cm) decreased in the B treatment by 580 stems/ha, although this was not statistically different ($p = 0.0504$) from the increase of 40 trees/ha in treatment C. Stems in size class 3 (DBH > 25 cm) in treatment B increased by 400 trees/ha, not statistically different ($p = 0.5093$) from the increase of 300 trees/ha in treatment C.

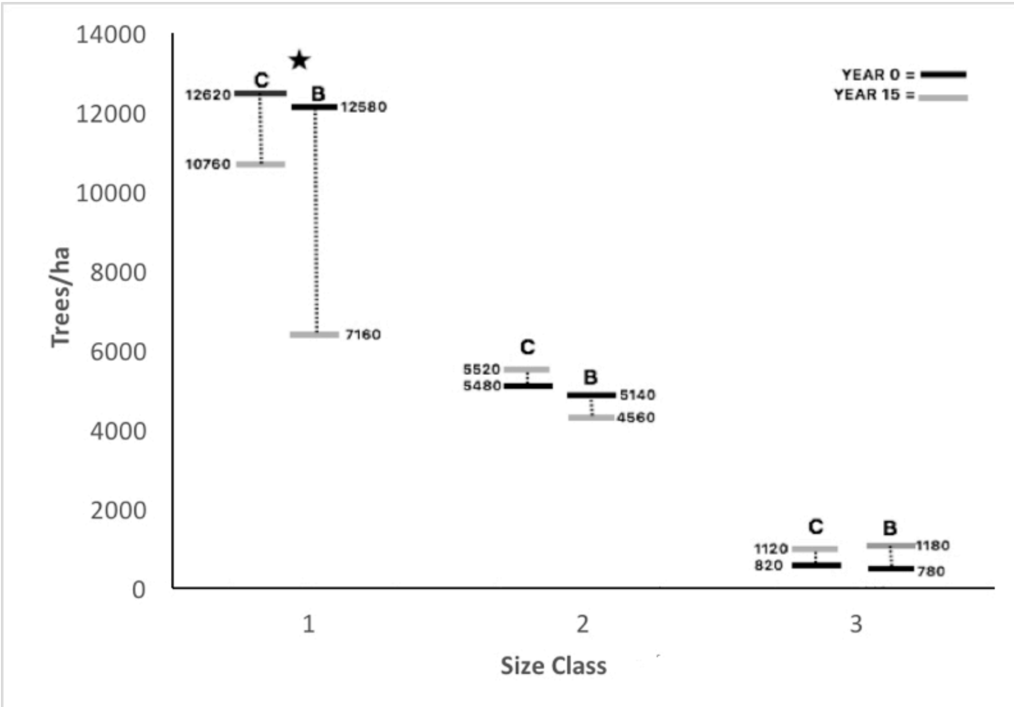

Figure 1.2: Diameter distribution by size class (size class 1: 10 cm < DBH < 25 cm, size class 2: 25 cm < DBH < 45 cm, and size class 3: DBH>45cm) of trees in the control treatment > 1.4 m in height and >10cm in diameter in Green River Game Land, Polk county, NC. The star symbol represents a statistical difference between treatments.

Figure 1.3 is a visual representation of basal area by size class. Analysis of the reduction of basal area in size class 1 trees, decreasing from 2.65 $m^{2/}$/ha to 1.94 $m^{2/}$/ha was not significant compared the C treatment (p = 0.1476). Basal area of size class 2 trees did not vary between treatments (p = 0.4651). Basal area of trees in size class 3 was 5.89 $m^{2/}$/ha in year 0 and 7.63 $m^{2/}$/ha in year 15 in the B treatment. This was not statistically significant (p = 0.4401) from the results observed in the C treatment, 5.72 $m^{2/}$/ha in year 0 and 8.43 $m^{2/}$/ha in year 15. None of these changes are significant, therefore there was no effect on basal area of trees in each size class from the burn regime.

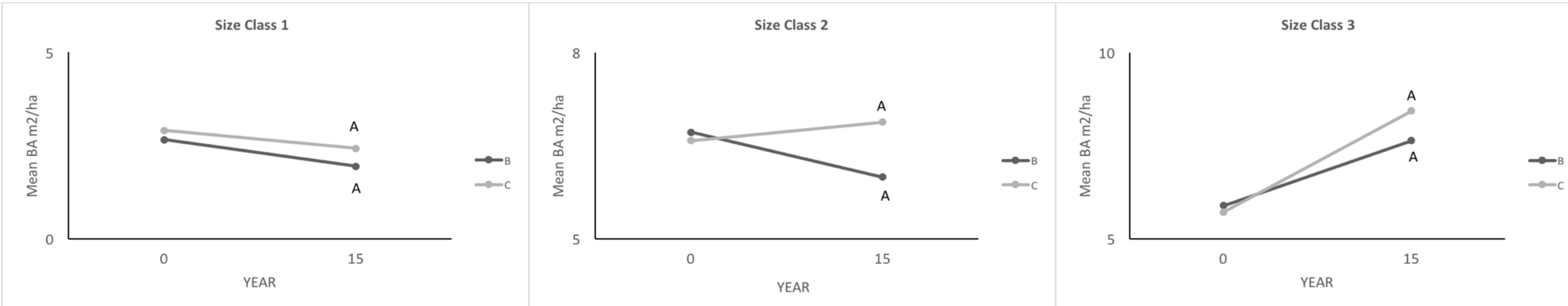

Figure 1.3: Basal area (m^2/ha) by size class (size class 1: 10 cm < DBH < 25 cm, size class 2: 25 cm < DBH < 45 cm, and size class 3: DBH>45cm) analysis on the grid point level where the slope of the line indicates the change in mean basal area based on the average grid point in the Green River Game Land, Polk County, NC. Different letters within each size class indicate significant difference in the change of mean basal area between treatments B and C.

Mean basal area (m^2/ha) by species group was analyzed to determine the effect of treatment on the species level. There was no significant change in YELLPINE (p = 0.1407), WHITEPINE (p = 0.8135), OAK (p = 0.7801), or HWD (p = 0.0723). The MSPHW group was the only group to show a statistically significant change from year 0 to year 15 (p = 0.0299) when compared to the change in the C treatment. The mean basal area in this group in the B treatment increased from 5.37 m^2/ha in year 0 to 6.34 m^2/ha in year 15 compared to an increase from 4.14 m^2/ha to 4.91 m^2/ha in the C treatment (Figure 1.4).

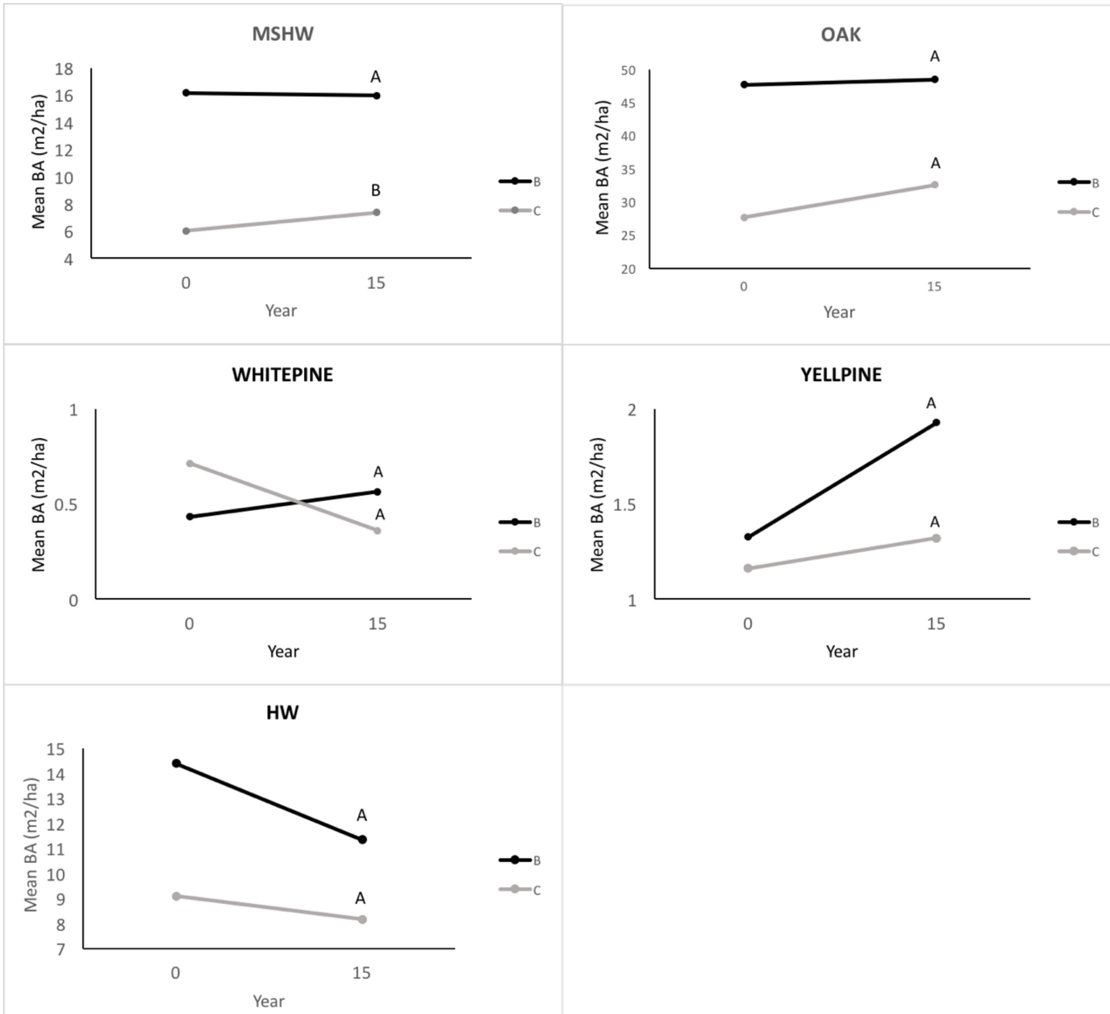

Figure 1.4: Basal area (m^2/ha) by size species group on the grid point level for trees > 10 cm at DBH in the Green River Game Land, Polk County, NC. The slope of the line indicates the change in mean basal area for that group at the grid point-level. Different letters within each species group indicate significant difference in the change of mean basal area between treatments B and C.

Discussion

Studies of fire and fire effects in the southern Appalachian Mountains have shown a mosaic of structural and compositional changes, at multiple scales, in response to fire (Lafon et al. 2017). This is largely a result of the variation in topography, soil types, regional weather patterns, fuel conditions, and previous vegetation (Hagan et al. 2015, Lafon et al. 2017). At smaller scales, variation in site conditions across the burn and control units of this study was accounted for by random sampling but could still have contributed to the different effects of burning in some plots. Due to this heterogeneity, fire severity and intensity likely varied significantly between the random array of plots (Waldrop et al. 2009, Waldrop et. al 2016).

Waldrop et al. (2016) tested the effects of fire and other fuel reduction treatments on three main objectives; restoration to an open woodland (structural changes), overstory and shrub regeneration (compositional changes) and fuel reduction in the same study site. Their analysis did not include the fourth burn in 2015 and they examined the effects year by year, as opposed to the entire regime. They found that burning only slightly reduced the overstory density (represented by basal area) but suggest that the burn regime could eventually achieve the structural changes desired for restoration. Additionally, their analysis suggested that burning did promote oak reproduction when compared to the control treatments. They acknowledge site variation as a potential confounding variable. The burn regime did significantly reduce fuel loading.

Just as Waldrop et al. (2016) reported, total overstory density decreased after the B treatment (Table 1.1). The benefit of this analysis is the incorporation of changes in mean midstory stem count (Table 1.2). With this comparison, we can infer which species groups may be growing into the overstory from the midstory, giving us a glimpse at the potential overstory composition if the burn regime is to remain constant. If, for instance, one sampling group had a decrease in midstory abundance after the treatment, but that species group had an increase in the overstory after the 15-year treatment, it could be inferred that ingrowth into the overstory size category occurred. Our results suggest that Nowacki and Abrams' (2008) prediction that the mesophication process will continue on forested landscapes where fire is suppressed may be accurate. The overstory showed a net decrease in mean stem count in the overstory after the burn regime, but an average increase in mean stem counts in the midstory, suggesting that the burn regime is restricting the MSHW group to smaller size classes, or alternatively, is promoting the regeneration of MSHWs. If the latter is true, this could provide insight into the future overstory composition. If more MSHWs in the midstory are able to outcompete surrounding tree species and recruit into the overstory, the overstory may become dominated by the MSHW group. Hardwoods displayed similar trends in the burn regime, but many of the species that make up the HW group are midstory trees, such as flowering dogwood, sassafras, black locust, and sourwood, which have forms that rarely reach the largest size classes.

Mesophication is arguably one of the biggest challenges that land managers face when trying to reintroduce fire to these systems. Nowacki and Abrams (2008) describe the mesophication process as a "positive feedback cycle" where micro-environmental conditions such as increased humidity, lower temperatures, and more shaded conditions with less flammable fuel beds continually facilitate the regeneration of shade-tolerant mesophytic species to the detriment of fire-adapted species (Nowacki and Abrams 2008).

The changes associated with mesophication on the forest floor also can restrict certain species, like pines, from germination by reducing pine seeds' chances of exposure to favorable conditions for growth. Most yellow pine seeds fare best on bare mineral soil or a thin litter and duff layer, which mesic forests rarely offer because of their continuous flat laying litter bed (Kreye et al. 2013). Yellow pine seed survival is also hindered by the reduction of high light levels reaching the forest floor in a mesophytic forest, due to the dense midstory and overstory (Waldrop and Brose 1999). This could contribute to our PINE species group results. All species groups, except for PINE, showed an increase in average stem count per hectare in the midstory. A potential confounding factor for the PINE group is the clustering of yellow pines and white pines. The yellow pines (Table Mountain pine, shortleaf pine, pitch pine, loblolly pine) have thick bark when mature, are less shade tolerant, and thrive in dry exposed sites and are relatively fire-tolerant. Eastern white pine is shade tolerant, thin-barked when juvenile, and thrives in moist environments but expand into xeric sites when fire is suppressed (Blankenship and Arthur 1999).

Increasing overall stem counts will alter the abiotic environment on the forest floor. Oaks can persist in the understory for very long periods of time relative to other species groups (Van Lear 2004). The oak seedlings require some sort of disturbance, large or small, to tip the competitive scale in their favor (Brose et al. 1999). Oak seedlings have proven to be relatively fire-tolerant, leading scientists to believe fire may be a key to oak regeneration (Brose and Van Lear 1998). After the burn regime, Oaks decrease significantly in the overstory, which may be a result of delayed mortality of large trees or even natural death of old oaks (Waldrop et al. 2016). The increase of oak stems in the midstory analysis of the burn treatment suggests oaks may be benefiting, although slightly, from the burn regime, but may still be outpaced by the increase of MSHW in the midstory, and thus may continue to be at a competitive disadvantage (Fei and Steiner 2009).

Changes in stem counts and composition do not necessarily relate to changes in basal area. If the large trees remain in the overstory, the abiotic environment underneath may be nearly as dark, shaded, and moist as it would be if no fire had occurred. Certain species such as yellow-poplar are very fast growing, but saplings rely on canopy disturbance shoot into the overstory and outcompete other trees for light (Beck and Della-Bianca 1981). Oaks can persist as small trees for a very long time, especially if they have a pre-established root system (Kuenzel and McGuire 1942, Clark and Watt 1971). They are slow growing but persistent, which suggests they may require much shorter disturbance return intervals to kill off competing vegetation until they can reach a size where their vast root systems can outcompete neighboring trees (Clark and Watt 1971).

Although stem counts for mesophytic hardwoods decreased in the overstory, we wanted to further examine the overstory dynamics. The increase in range and mean basal area of the MSHW group, and its decrease in total overstory stem counts after the B treatment suggests that the large MSHW trees have survived and continued to grow. Although conclusive, this trend does not seem to be limited to MSHWs alone. The mean and range for all species in size class 3 increased as well. This could be a result of the larger trees being more resistant to top-kill than smaller trees. The significant decrease in trees in size class 1 and the increase in trees/acre in size class 3 from pre- to post-treatment suggests this as well. The decrease of PINEs in the overstory and midstory over the course of 15 years is likely attributed to the natural path of succession in which shade intolerant trees that reached the overstory naturally are no longer able to regenerate on the forest floor below them because of the changes on the forest floor such as shading and competition.

Conclusion

A fifteen-year dormant season burn regime did not fully combat the mesophication process on our study site that has shifted the structure and composition of forests in the southern Appalachian Mountains. Nonetheless, there is some evidence that this regime is aiding to reverse the trend. The dormant season burn regime had modest effects on forest structure by reducing the number of smaller trees in the overstory, but it also promoted the recruitment (likely from sprouts) of MSHWs into the midstory. This suggests that the composition of the overstory may eventually be dominated by the less desirable MSHWs, unless adaptive management actions are taken to achieve a different goal.

Management implications of these results lean towards changing characteristics of the burn regime such as return interval or season. Periodically introducing a growing season burn within the regime may have significant effects on reducing competition of undesirable mesophytic species and potentially tip the competitive advantage to more fire tolerant species and potentially aiding in restoration efforts. Burning in the growing season when the plants have a smaller proportion of their carbohydrates stored root systems may target tolerant trees when they are most vulnerable. Changing fire return intervals may also be an effective method in changing the outcomes of management plans. Lengthening the time between burning could drastically change fire behavior, severity, and intensity. Perhaps only fifteen years of management has not been long enough to fully reverse the effects of more than 80 years of fire suppression. More long-term studies, perhaps using different fire regimes, are needed to further scientific understanding of fire management techniques in the southern Appalachian Mountains.

CHAPTER TWO

INFLUENCE OF FIRE SEASONALITY ON THE RESPROUT RESPONSE OF FOUR SOUTHERN APPALACHIAN TREE SPECIES

Introduction

Fire has occurred in the southern Appalachian Mountains since pre-Columbian times but has only gained research interest within the last 20-30 years (Lafon et al. 2017). The assembly of plant communities in this region was largely influenced by historical fire regimes (Delcourt and Delcourt 1997; Waldrop et al. 2016; Lafon et al. 2017). Frequent fire, for example, restricted thin-barked species such as red maple (*Acer rubrum*), eastern white pine (*Pinus strobus*), and mountain laurel (*Kalmia latifolia*) from highly exposed xeric sites like those found on high elevation ridgetops and southwestern slopes (Turrill 1998; Waldrop and Brose 1999). After public land acquisition increased in the early 1900's and fire suppression policies became more prominent in the 1930's, these species began to spread, changing the structure and composition of the forests across all scales (Abrams 1992; Ducey et al. 1996; Waldrop and Brose 1999).

In pre-settlement southern Appalachia (late 15th century and earlier), surface fires maintained oak-pine ecosystems in a relatively open state allowing higher light levels to reach the forest floor to facilitate the growth of grasses, forbs, and shrubs (Abrams and Downs 1990; Nowacki and Abrams 2008; Lafon et al. 2017). The historic low tree density of the montane oak-pine ecosystems was largely a result of long term fire regimes of the region. The term 'fire regime' refers to the behavior of fire, the fire return intervals, typical fire season, and any other aspect of a site's average fire history (Pyne 1982; Agee 1993; Whelan 1995; Pyne et al. 1996; Guyette et al. 2002). Historic fire regime in the southern Appalachian Mountains was characterized by approximately 7 – 10 year return intervals, moderate to severe fire behavior, and mostly dormant season fires, inhibiting the establishment and persistence of smaller diameter stems and fire sensitive species (Nowacki and Abrams 2008; Knapp et al. 2009, Lafon et al. 2017). Grasses and forbs dominated the understory, large thick-barked species were predominant in the overstory, and the midstory was largely absent with the exception of few resilient individuals that grew into fire-tolerant sized classes and would eventually reach the canopy (Delcourt and Delcourt 1997). With the absence of frequent surface fires, midstory tree density in southeastern oak-pine open woodlands increased. Xerophytic species became displaced by fire sensitive, mesophytic species such as birches (*Betula* spp.), blackgums (*Nyssa sylvatica*), yellow-poplars (*Liriodendron tulipifera.*), and maples (*Acer* spp.) (Nowacki and Abrams 2008; Lafon et al. 2017). Overstory tree richness also increased as fire-restricted species recruited into tree size classes (Nowacki and Abrams 2008). The process of this shift from open woodlands to dense, vertically stratified, closed canopy forests is referred to as "mesophication" (Nowacki and Abrams 2008). Increased shading caused by the dense midstory and overstory further limits the potential for successful spread of fires (Nowacki and Abrams 2008).

Many tree species are known to sprout after disturbances as an adaptation for survival (Fei and Stiener 2009). Prescribed burning at different times of the year may influence the vigor of resprouting after top kill by fire, but this may also vary by species. Sprout vigor is referred to here as the plants' investment towards sprouting following a disturbance which, presumably, increases the likelihood of future survival. Although some evergreen species can take advantage of unseasonably warm days, during the coldest parts of the year neither evergreen nor deciduous plants have the ability to photosynthesize sunlight into carbohydrate energy (Rohde and Bhalerao 2007). Belowground storage of carbohydrate reserves during the dormant season provides perennial plants with a measure of protection against fire (Rohde and Bhalerao 2007). When temperatures rise again in the spring, plants break dormancy using their carbohydrate reserves from the previous year to promote new growth in leaves, primary growth, and secondary growth. After breaking dormancy, the plants begin the photosynthetic process over again and build up energy reserves for primary growth, secondary growth, reproduction, and storage for future years (Rohde and Bhalerao 2007). The movement and storage of carbohydrates to the root system creates a soil barrier between nutrient reserves and aboveground disturbances like fire (Rohde and Bhalerao 2007). Due to seasonal differences in carbohydrate storage, disturbance during different times of year may affect a plants ability to resprout and compete favorably with other species (Little and Somes 1956).

For the purposes of this study, the time of year in which a fire occurs is referred to as fire seasonality. In the southeastern United States, most prescribed burning has been conducted during the dormant season (January – March), when leaf off has increased the amount of fine fuel on the ground (leaf litter) and increased light penetration to the forest floor creates a more fire conducive environment (Knapp et al. 2009). Fire managers in the southeast commonly refer to middle to late dormant season as “burn season,” although there is little scientific evidence to support whether this is the best time to burn when trying to meet certain management objectives such as the restoration historical community composition and structure, or reducing hardwood encroachment on industrial pine plantations.

In fact, managers and scientists across the region have increasingly noted that dormant season burning often does not adequately control undesirable fire-sensitive species like red maple and yellow-poplar (Huntley and McGee 1981, Barnes and Van Lear 1998). Huntley and McGee (1981) found, when burning 3-year-old hardwood clear cuts, that dormant season burning had virtually no impact on red maple regeneration. Barnes and Van Lear (1998) found that dormant season burning was not as effective as growing season burns in controlling yellow-poplar. Growing season burning has been successful in management objectives such as increasing advanced oak regeneration, reducing hardwood competition on oak regeneration and controlling hardwood encroachment in pine forests (Waldrop et. al 1992, Van Lear 1998, Brose and Waldrop 2014). Hooper (1969) studied the effects of prescribed burning on mountain laurel and rhododendron southern Appalachian Mountains. His study utilized prescribed burning in the fall season, measured mortality and monitored resprouting up to eighteen months after burning. He found a substantial increase in total number of mountain laurel and rhododendron stems, even after over 83 percent of the original stems had been completely top killed or severely burned during the prescription.

This study aims to address the question of fire seasonality in regards to four species of management interest. Our objectives were (1) identify the best time to burn if the goal is to eliminate or reduce the presence of undesirable species (i.e. red maple or mountain laurel); (2) identify the best time to burn in order to tip the competitive edge towards desirable species (i.e. black oak [*Quercus velutina*] and shortleaf pine [*Pinus echinata*]). Three variables (maximum sprout length, number of total sprouts, and sprout biomass) were used as proxies for sprouting vigor. We hypothesize that sprouting of the two desirable species, shortleaf pine and black oak, will be less affected by burn season than the two undesirable species, red maple and mountain laurel, because they are presumably more fire-adapted species. Results may also indicate that burning in the early growing season months (April-May) will have the greatest effect on decreasing resprout vigor of the two deciduous species, red maple and black oak, and less so for the shortleaf pine and mountain laurel. This is attributed to the energy deficit just after the trees have broken dormancy and used stored energy for new leaf growth. Additionally, we hypothesize that the least vigorous sprouting will occur on individuals burned during the early dormant season months (October and December) and late dormant season (March). We hypothesize a more gradual change in resprouting for the two evergreen species, presumably because of their ability to take advantage of warm days and photosynthesize year-round.

Materials and Methods

Site Description

The Clemson Experimental Forest (CEF) is roughly 7,082 hectares immediately surrounding Clemson University in Pickens County, South Carolina, part of the Piedmont region of the United States. The CEF is primarily comprised of an oak/hickory complex.

The forest was replanted by the Civilian Conservation Corps in the 1930's with the goal of stabilizing eroded agricultural lands (Hartman and Rentz 1938; Hagan et al. 2014). The soil orders of this region include Ultisols, Entisols, and Inceptisols. Soil series included Pacolet (fine, kaolinitic, thermic Typic Kanhapludults), Cataula (fine, kaolinitic, thermic Oxyaquatic Kanhapludults), and Cecil (fine, kaolinitic, thermic Typic Kahnapludults) (U. S. Department of Agriculture–Natural Resources Conservation Society [USDA/NRCS] 2008; Hagan et al. 2014). The CEF is in the udic moisture regime with a mean annual precipitation of 46.15 in. (117.22 cm), and the thermic temperature regime with a mean annual temperature of 63.63 F (15.4 C) (U.S. Historical Climatology network 2018). Late-successional overstory-species include white oak (*Quercus alba L.*), water oak (*Quercus nigra*), black oak, yellow-poplar, pignut hickory (*Carya glabra*) and American beech (*Fagus grandifolia*). Understory species include Japanese honeysuckle (*Lonicera japonica*), Christmas fern (*Polystichum acrostichoides*), juvenile overstory species, and multiple species of maple (Hagan et al. 2014).

All individuals of the same species are in the same area of the CEF. The CEF was chosen as the study site for this project because of its close proximity to the University. Although the site is located in the Piedmont, the study uses species that are common in the southern Appalachian Mountains. Trees from three sites were sampled: an exclusive site for shortleaf pine and another for the mountain laurel, and one site included both red maple and black oak (Figure 2.1). Sample individuals were chosen based off of size. Trees were chosen to be between 1.5 m and 5 m tall with root collar diameters no larger than 7 cm.

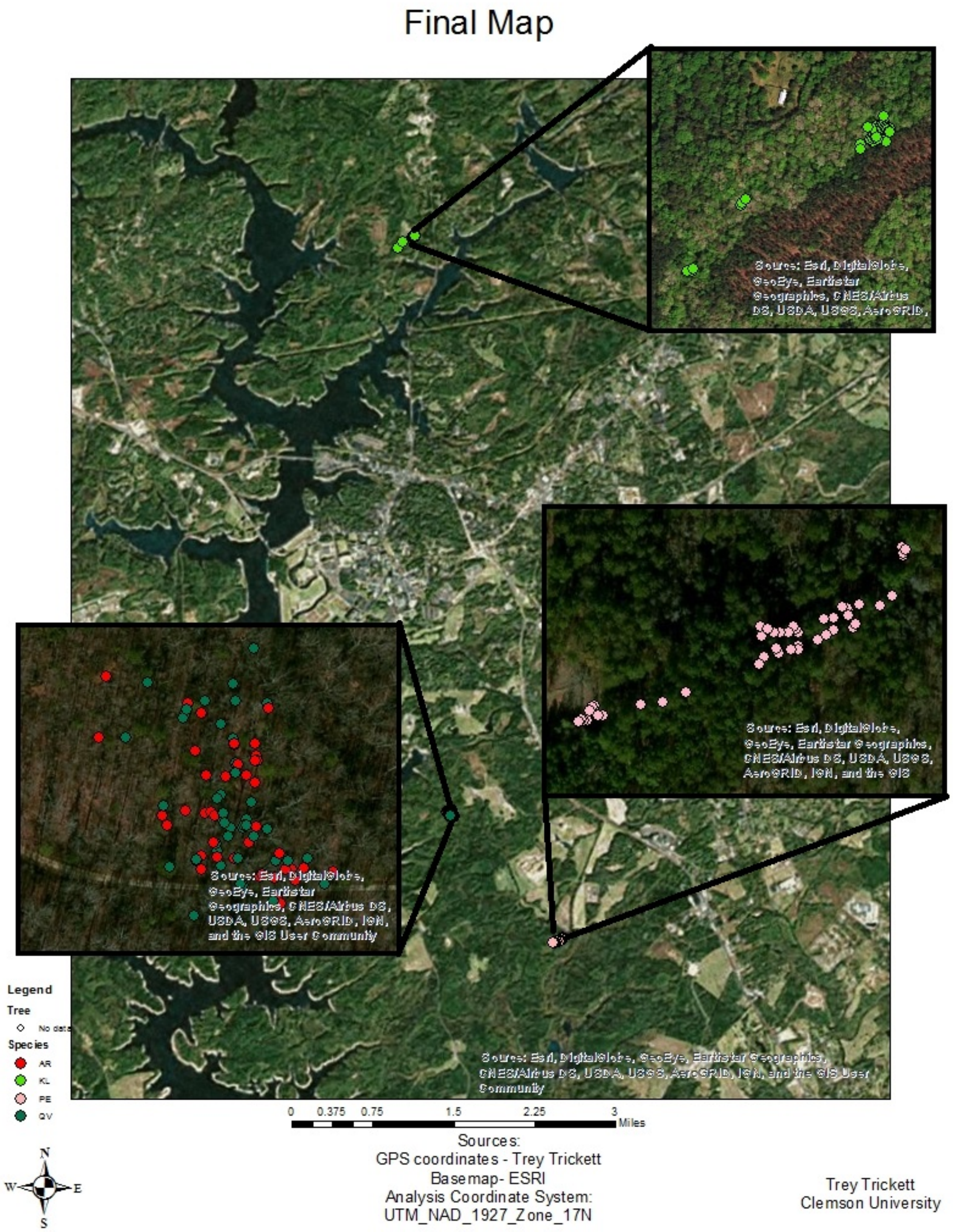

Figure 2.1. Map including all 3 study areas in Clemson South Carolina. Torched trees (some missing points) are represented as points on the map.

Study Design

For this study, we selected two species considered to be fire tolerant (shortleaf pine and black oak), and two fire sensitive species (mountain laurel and red maple). Stems were burned 30 cm above the root collar for 45 seconds on each side using a hand-held, propane-fueled weeding torch with a valve adjusted output (Figure 2.2). The temperature goal corresponding to flame intensity was between 700 and 800 degrees Celsius, which is comparable to a typical prescribed surface fire of moderate to high intensity in the southern Appalachian Mountains (Waldrop et al., 2009). Six individuals of each of the 4 species were located and top-killed via propane torch (Red Dragon VT 2-23C) (Figure 2) at the beginning of nearly every month from March to December, 2016. During November 2016 there was a region-wide burn ban, preventing the use of the torch during that month. This resulted in a collective sample size of 54 individuals for each species.

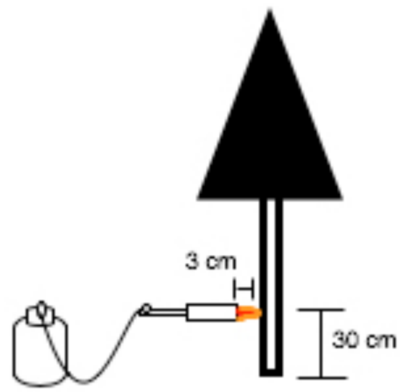

Figure 2.2. Diagram of the torching method for each tree. Torch held at 30 cm above the ground and 3 cm away from the stem for 45 seconds on each side of the stem.

Data Collection

Trees were tagged using aluminum tags marked with a tree number and blue paint around the stem so that it would be visible when revisiting for one-year post-fire data collection. One year after burning, all trees from that month were revisited and sprouts from the 24 sample trees were harvested, bagged, and labeled. The harvested sprouts were frozen until they could be counted, then placed into size categories (< 3 cm: 3 – 15 cm: 15 – 25 cm: 25 – 50 cm: > 50 cm) and the longest sprout per sample was measured. Total number of sprouts, length of the longest sprout (cm), and the biomass (g) of sprouts were then measured in the lab and samples were placed in separate paper bags. The deciduous sprouts (red maple and black oak) were stripped of their leaves to avoid bias in favor of the growing season samples, due to the variability of the biomass added in the leaves. Biomass was measured by oven-drying the bagged sprouts at 75 degrees Celsius for 48 hours. Samples were then weighed and stored.

The study variables (total sprouts, maximum sprout length, and biomass) were chosen to address the different sprouting strategies the different species may have. For instance, one species may invest its energy in large quantities of smaller sprouts, versus smaller numbers of large sprouts, but the overall sprout biomass measurements may be similar. Maximum sprout length, for some species, may be a reliable indicator of future competitive ability, since longer sprouts may be more likely to survive than shorter ones (Clabo and Clatterbuck 2015).

Data Analysis

Data for the 3 dependent variables (total number of sprouts, maximum sprout length, and total sprout biomass) were analyzed using JMP Pro 13.2 (SAS Institute, Cary, NC). Standard one-way ANOVA testing was performed on each species for each variable for all months. If the variances were unequal for a particular species/variable combination, Welch's one-way ANOVA for unequal variances testing was performed. Normal distributions were determined by visually assessing normal quantile plots. Biomass analysis for mountain laurel was transformed logarithmically to meet distribution assumption for the analysis. All other assumptions were met for data used. If the appropriate ANOVA produced statistically significant results (alpha = 0.05), post-hoc analysis using Student's t-tests was performed for each pair comparing variable means by month within species.

Results

Total Sprouts

The total number of sprouts of red maple did not vary significantly between months ($p = 0.2839$). Total sprouts ranged from 0-21 sprouts with a total average of 10.3 sprouts (Figure 2.3a).

Analysis of mountain laurel total sprouts does not show significant difference between months ($p = 0.2769$). Total sprouts ranged from 0 – 112 sprouts with a mean of 38.5 sprouts. Figure 2.3b shows the mountain laurel sample mean total sprouts over months.

For shortleaf pine, the total number of sprouts was significantly influenced by the month ($p = 0.0426$). Total sprouts of shortleaf pine ranged from 0 – 157 sprouts with a mean of 31.8 sprouts. Figure 2.3c shows shortleaf pine sprouting rising from March (25.0 sprouts) to May where it reached its maximum mean of 55.3 sprouts. There is a sharp drop in mean sprouts from May to June where total mean sprouting was lowest (6.8 sprouts). Sprouting of shortleaf pine rose from June to August (49.5 sprouts) before declining again through December (13.8 sprouts). Post-hoc comparisons revealed significant differences in mean sprouts between the months of May and June (p = 0.0263). Sprouting in April (49.5 sprouts) and August also differed significantly when compared to June (p = 0.0493).

Total number of sprouts for black oak across all months ranged from 0 – 14, with a mean of 3.8 sprouts/month. Although there was no significant relationship between total sprouts and month (p = 0.1156), sprout numbers generally declined during the study period (Figure 2.3d).

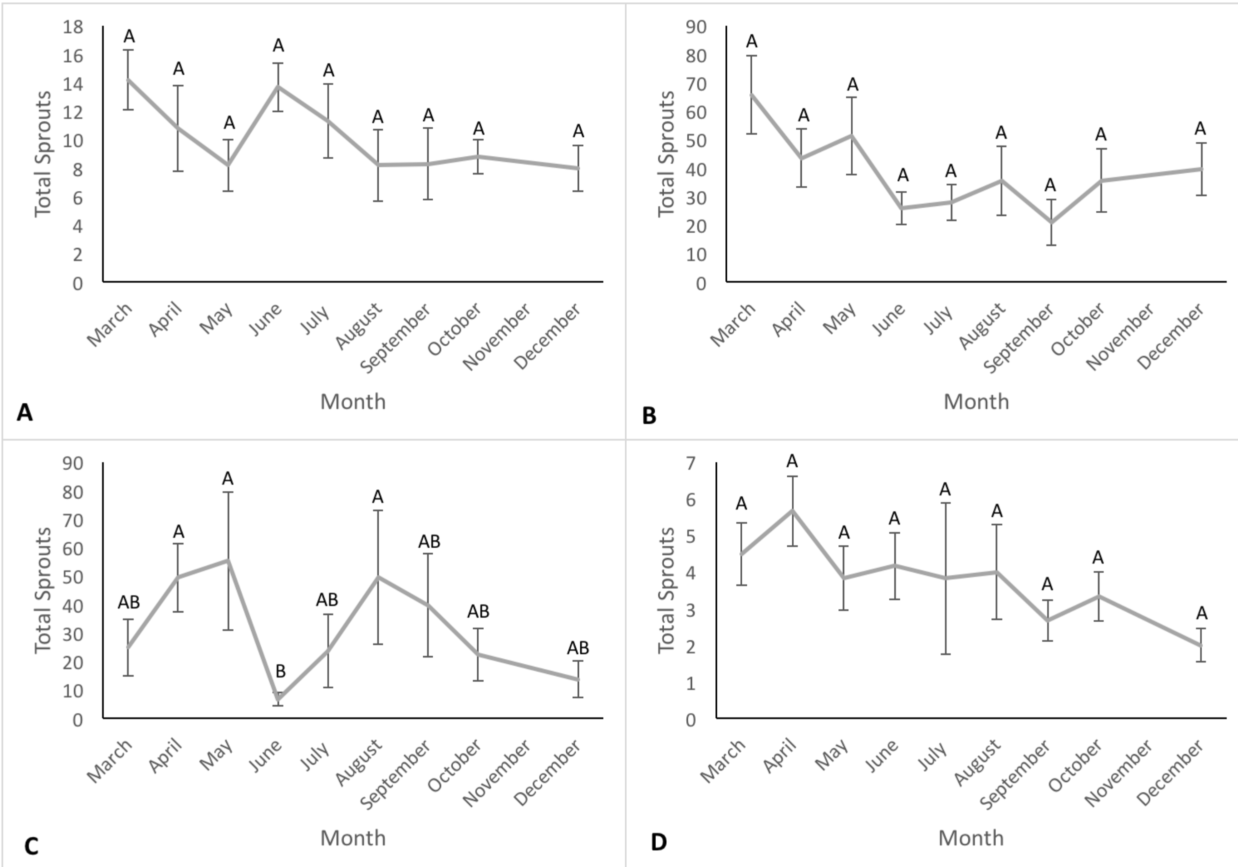

Figure 2.3: (A- red maple) (B- mountain laurel) (C- shortleaf pine) (D- black oak) Line charts connecting mean total sprouts for species experimentally burned with a torch (n = 6, per species, per month), from March to December 2016 in the Clemson Experimental Forest, Clemson, SC, USA. Sprouts were collected 1 year after burning. Error bars conveying standard error. Monthly means with different letters are statistically different at alpha = 0.05. Data are not available for November due to a region-wide burn ban that prevented the use of the torch during that month.

Maximum Sprout Length

There was a significant relationship between maximum sprout length and months for red maple ($p = 0.0151$). Maximum sprout lengths ranged from less than 3.0 cm to 88.5 cm with a mean of 48.1 cm. Post-hoc analyses identified 7 months with significant differences in means. Mean maximum sprout length decreased from March (51.3 cm) to its lowest mean in April (27.0 cm), then began to steadily increase from April to July (57.1 cm) (Figure 2.4a). The difference of maximum sprout length in August (36.1 cm) varied significantly from maximum sprout length in September (65.0 cm) and October (60.9 cm).

Maximum sprout length did not vary significantly by month for mountain laurel ($p = 0.3575$). Sprout lengths ranged from < 3.0 cm to 34.0 cm with a mean of 17.3 cm. Mean maximum length was highest in March at 20.0 cm and lowest in September with a mean of 12.3 cm (Figure 2.4b).

There was a significant relationship between maximum sprout length of shortleaf pine and month ($p = 0.0323$). Maximum sprout length ranged from 0.0 – 59.0 cm with a mean of 19.6 cm. Post-hoc analyses identified a significant difference in mean maximum sprout length between March (29.2 cm) and June (5.9 cm) ($p = 0.0167$). Figure 2.4c shows a modest downward trend from March to May (24.3 cm), then an abrupt decrease from May to June. Mean maximum sprout length rose from June to September (23.5 cm), then began to decline incrementally through December (12.4 cm).

Analyses did not reveal a significant difference between black oak maximum sprout length and month ($p = 0.7256$). Maximum sprout lengths ranged from 0.0 – 100.0

cm with a mean of 41.8 cm. Figure 2.4d shows highest mean maximum sprout length in September (51.4 cm).

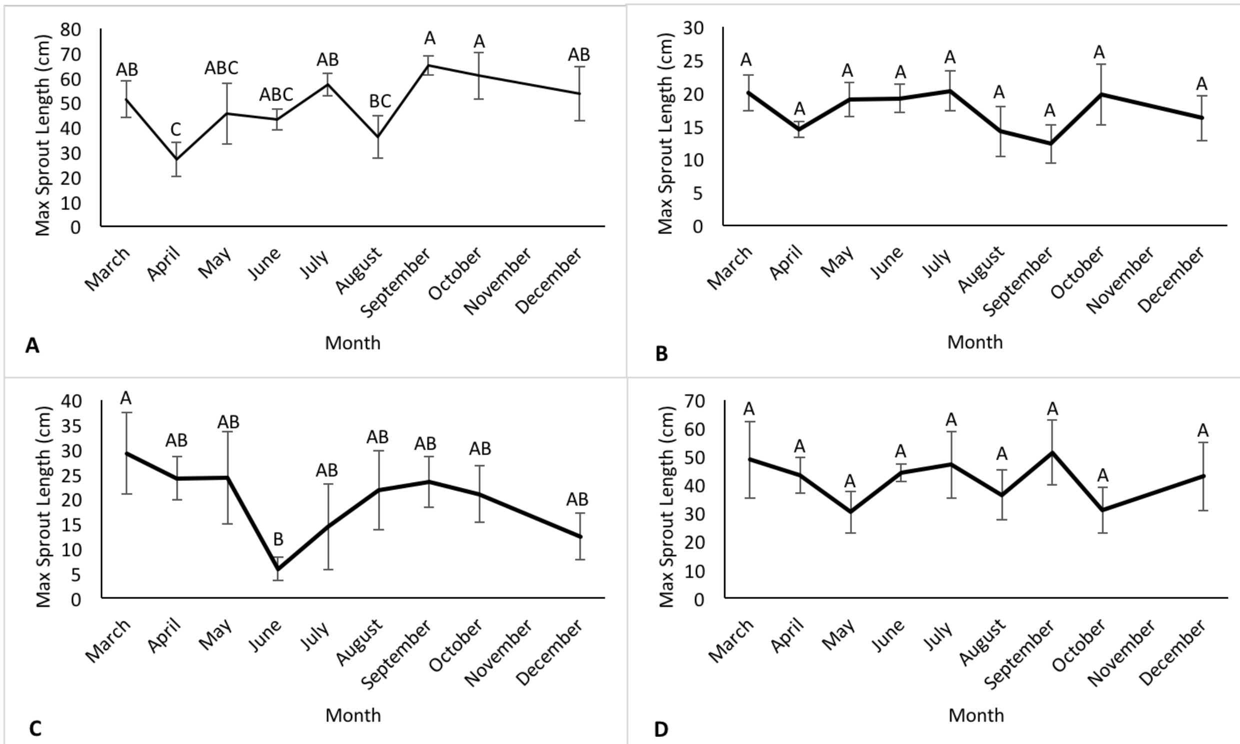

Figure 2.4: (A- red maple) (B- mountain laurel) (C- shortleaf pine) (D- black oak) Line charts connecting mean maximum sprout length (cm) for species experimentally burned with a torch (n = 6, per species, per month), from March to December 2016 in the Clemson Experimental Forest, Clemson, SC, USA. Sprouts were collected 1 year after burning. Error bars conveying standard error. Monthly means with different letters are statistically different at alpha = 0.05. Data are not available for November due to a region-wide burn ban that prevented the use of the torch during that month.

Sprout Biomass

Analysis of total sprout biomass by month did not prove statistically significant for red maple (p = 0.1329). Biomass ranged from 0.0 – 18.2 g with a mean of 5.4 g (Figure 2.5a).

Total sprout biomass for mountain laurel varied significantly by month (p = 0.0010). Biomass ranged from 0.0 – 75.1 g with a mean of 10.1 g. Post-hoc analyses of means identified significant difference of mean biomass in September (2.1 g), which was significantly lower than all other months, and in October (28.8 g) and December (29.1 g), which were significantly higher than all other months. The largest differences in mean biomass was from October to September (p = < 0.0001).

Shortleaf pine did not show a significant difference in sprout biomass by month (p = 0.5251). Biomass for shortleaf pine sprouts ranged from 0.0 – 86.5 g with a mean of 14.2 g (Figure 2.5c).

Sprout biomass of black oak did not vary significantly by month (p = 0.4671). Sprout biomass of individuals ranged from 0.0 – 27.8 g with a mean of 6.1 g (Figure 2.5d).

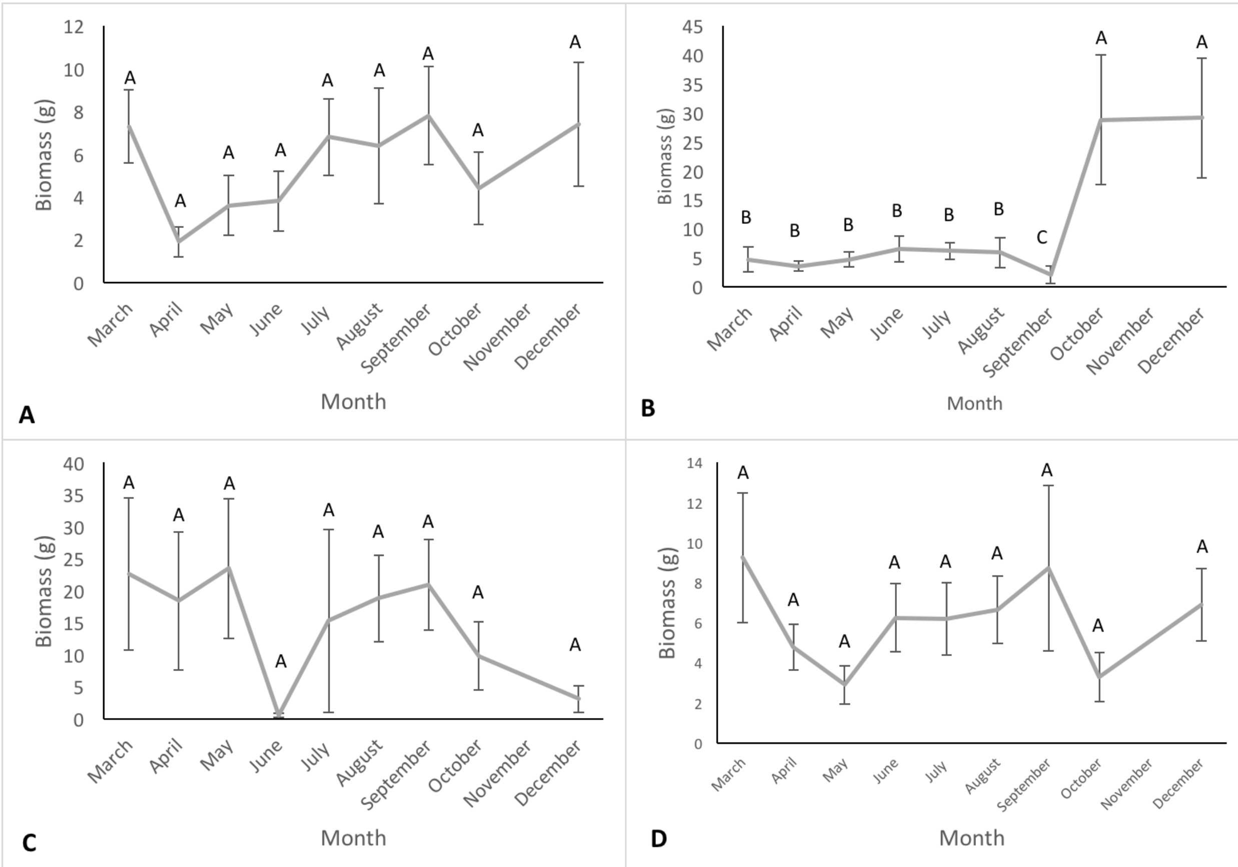

Figure 2.5: (A- red maple) (B- mountain laurel) (C- shortleaf pine) (D- black oak) Line charts connecting mean sprout biomass (g) for species experimentally burned with a torch (n = 6, per species, per month), from March to December 2016 in the Clemson Experimental Forest, Clemson, SC, USA. Sprouts were collected 1 year after burning. Error bars conveying standard error. Monthly means with different letters are statistically different at alpha = 0.05. Data are not available for November due to a region-wide burn ban that prevented the use of the torch during that month.

Discussion

Red maple

Red maples are widespread across the eastern half of North America and are considered to many foresters an inferior and undesirable timber resource because they often appear poorly formed, especially on low-quality sites (Walters and Yawney 1990). In recent years, an increase in overstory red maples has been observed, often co-occurring with a decline of more desirable species, such as oaks (Lorimer 1984; Abrams 1998; Fei and Steiner 2009). The increasing dominance of red maples throughout the eastern United States has been largely attributed to their ability to resprout and recapture growing space after a disturbance, in addition to their proliferation on sites undisturbed by fire (Brose and Van Lear 1998; Fei and Steiner 2009).

In many montane oak-pine communities in the southern Appalachian Mountains, red maples are one of the main species that managers aim to control to curtail the mesophication process (Fei and Stiener 2007, Nowacki and Abrams 2008, Brose and Waldrop 2014, Dey 2014). This study shows that after top-kill by fire, red maples have a tendency to allocate resources towards creating moderate amounts of medium to large sprouts. Because of this, the best sprout variables to consider for the red maple would be maximum sprout length as well as sprout biomass.

Based on the results, if managers are looking to control red maple in the understory and midstory, the best times for burning would be right after the leaves have sprouted from the buds but are not fully formed yet resulting in the plant's largest energy deficit of the year. In this case (Piedmont region in 2016) that time was in May. This is particularly interesting considering most prescribed fires in the region are conducted between January and March. Burning between August and December may also be a viable time to burn if other factors or objectives prevent the burning during the May energy deficit. Shortleaf pine shows a nearly inverse trend when compared to red maple, suggesting that burning during the aforementioned months would be best for controlling red maple while simultaneously promoting vigorous resprouting of shortleaf pine.

Mountain laurel

Controlling mountain laurel is an important management goal in the southern Appalachian forests (Vose et al. 1993). Once established on a site, it can overtake the midstory and potentially affect the regeneration of overstory trees (Vose et al. 1993). Managers have termed the vast groves of mountain laurel that are now common in previously open communities “laurel hells” (Ducey et al. 1996). Literature in recent years has suggested the overabundance of mountain laurel has affected the recruitment of desirable oak species (Moser et al. 1996) and pine species such as Table Mountain pine (Waldrop and Brose 1999). Mountain laurel can also create more extreme fire behavior such as higher flame height, intensity, and severity, by acting as a ladder fuel in which surface fires engulf the shrubs’ crowns and potentially lead to crown fire conditions (Waldrop and Brose 1999).

The results of this study show that after top-kill by fire, mountain laurel has a tendency to allocate resources towards large amounts of very small sprouts. In only a few cases were large sprouts found. Because of this tendency, it seems that total sprout biomass is likely the best variable to consider when deciding the optimum season for controlling mountain laurel with fire. Based off of the biomass results, the best fire season for controlling mountain laurel is during June, July or between September and March. These results likely reflect the physiology and phenology of mountain laurel. Because mountain laurel growth responds to day/night temperature, it can take advantage of the increased amount of light that reaches the midstory on warm winter days before deciduous overstory has developed leaves (Asiah et. al 1992, Öquist and Huner 2003). Managers should make decisions on burn season based off of the physiological state of the plant communities around them. During the treatment period, mountain laurel was likely just breaking dormancy in March. Resprouting of mountain laurel was pretty well curtailed during all of the growing season. Burning between October and January would be the least effective for controlling mountain laurel.

Shortleaf pine

Shortleaf pines are one of the few pine species that have the capability to resprout after disturbances (Guldin 1986; Clabo and Clatterbuck 2015). Shortleaf pine numbers have been reduced since the fire suppression era in the middle to second half of the 20th century (Birch et al. 1986; Coffey 2012; Clabo and Clatterbuck 2015). Other factors contributed to this decline, such as southern pine beetle outbreaks and an industrial preference towards loblolly pine (*Pinus taeda*) (Birch et al. 1986).

A study by Clabo and Clatterbuck (2015) examined the sprouting capability of shortleaf pines one year following a top-kill treatment. The sample seedlings were subjected to either clipping or top-kill by fire in April. They found that one-year-old planted seedlings did not have high survival rates or large numbers of sprouts in response to these treatments during the early growing season. However, our results display some of the highest means for both total number of sprouts and maximum sprout length during the early growing season (April and May). Again, managers should consider the physiological state of the plant communities they are treating. Early growing season was a good time to burn during our study period because at that time, they had only barely begun to put on new growth and still had plenty of energy reserves belowground.

The results of this study showed shortleaf pines had a tendency to produce large amounts of sprouts of all different sizes. Maximum sprout length and total number of sprouts are likely the best variables to consider when assessing the plant's ability to survive after top-kill by fire. Based off of total sprouting, the best time to burn while promoting survival of established shortleaf pines would be between April and May, or between August and September, especially if there is competition from red maples.

Black oak

The regeneration and promotion of oaks in southern Appalachian communities has been a hot topic for research since the reintroduction of fire research since the middle to late 20th century (Moser et al. 1996). Oaks' unique life history, tolerance to disturbance, and their wildlife and timber value makes them a prime genus of interest in fire research (Van Lear 2004). Their ability to resprout repeatedly from root collar buds (Waldrop et al. 1987; Van Lear 2004), their thick bark which insulates the cambium from heat (Hare 1965; Van Lear 2004), and the fact that their acorns are often buried by local fauna and thereby insulated from heat by the soil are all adaptations that previously aided in the perpetuation of upland oak communities (Van Lear 2004).

Literature over the years has suggested that the absence of fire has limited oak regeneration (Lorimer 1984; Host et al. 1987; Abrams and Nowacki 1992). Other factors have been linked to the limiting of oak regeneration to shrub competition and the accumulation of shad tolerant trees in the understory (Hannah 1987; Crow 1988; Lorimer et al. 1994; Moser et al. 1996). Mountain laurel in particular has been of high management concern throughout the eastern United States because of its high shade tolerance and aggressive vegetative growth habit (Chapman 1950; Moser et al. 1996). Moser et al. (1996) investigated the effects of fire intensity on competitive dynamics between red and black oaks on mountain laurel. Their results suggested that light understory fires alone are not sufficient for oak regeneration in Northeastern Connecticut. Nyland et al. (1982) considered single burns are not likely to enhance oak recruitment significantly. Van Lear and Watt (1993) advocated the use of prescribed fire before harvesting a site to favor oak regeneration. Other studies have shown that the incorporation of multiple management prescriptions such as selective harvesting in combination with burning, shelterwood-burn technique, may be a viable technique for establishing advanced oak regeneration in the Piedmont region of South Carolina (Brose et al. 1999).

Like Nyland et al. (1982), our results show that the resprout response of oaks to a single burn is not influenced by season. This is likely because of their ability to persist in the understory until conditions are optimum for establishment in the midstory and eventual growth into the overstory (Van Lear 2004). Oaks' unique growth investments towards an expansive root system over above ground growth results in a high root to shoot ratio (Reich et al. 1980). This could be why we observed small amounts of large sprouts. Although leaf data were not reported, these small sprouts low to the ground did appear to develop very large and healthy leaves. Single burns may not have had much of an effect on the black oaks.

Conclusion

This study shows that all plant responses to fire are not equal. Likely due to their morphological, phenological, and physiological differences, sprouting response of the target species differed with season of burning. This stresses the importance of selecting a burn season that corresponds with specific management goals that aim to promote one group of species while controlling another. The key to achieving management goals that have been elusive throughout the recent fire management era could be found with providing fire managers with scientific insight on what to community effects can be expected from burning in different times of the year.

It is important to mention this potential confounding factor as a result of the artificial nature of our torching treatment. Unlike natural or prescribed surface fires in which the shallow fine roots may be damaged by subsequent soil heating, the stem torch treatment did not simulate this phenomenon. This could have resulted in more resprouting and/or less mortality in sample trees. It is equally as important to mention the months correlate to the conditions of each tree species during that month. This will vary between ecoregions, namely between the Piedmont and montane regions.

Our results suggest that promotion of shortleaf pine and the control of red maple could be accomplished with prescribed burning in April and May. Control of mountain laurel may be best accomplished with prescribed fires in June and July. However, burning earlier in the growing season could accomplish moderate control of mountain laurel and significant control of other red maple. Oaks did not seem to have a response to burn season after a single fire. This supports the idea that sprouting enables them to persist in the understory until the optimum time comes to establish and grow into a more fire-tolerant size class.

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