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15 Years of Fire and Fire Surrogate Treatment Effects on Understory Vegetation in the Southern Appalachian Mountains, USA

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15 YEARS OF FIRE AND FIRE SURROGATE TREATMENT EFFECTS ON UNDERSTORY VEGETATION IN THE SOUTHERN APPALACHIAN MOUNTAINS, USA

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ABSTRACT

Decades of fire exclusion in the Southern Appalachian region have caused the forests to convert from open woodlands to closed canopy mesic forests with sparse understories. The main objectives of this study were 1) to assess the effects of four fuel reduction methods (burned [B], mechanical fuel treatment [M], mechanical treatment + burned [MB], and control [C]) on understory vegetative functional groups from 2001-2016; and 2) to investigate understory community-level responses after 15 years of treatment effects. In response to the first objective, oak species had significant increases in MB and B, relative to other treatments. However, mesic hardwood species had comparably significant increases in B, driven by red maple. Similarly, shrub species had significant increases in M, driven by mountain laurel and great rhododendron. Conversely, forb and graminoid species had non-significant increases in cover among all treatments. In response to the second objective, vegetation patterns seemed to overlap with respect to treatment type, suggesting little separation in understory community. However, some clusters from the hierarchical cluster analysis showed divergent communities from C treatments, particularly for shrubs and herbaceous species. In response to the third objective, select herbaceous species indicate changes in understory abiotic conditions, suggesting reversal from mesic conditions. Additionally, these findings suggest the M may not serve as a surrogate for B treatments over 15 years. MB treatments, however, are providing sufficient abiotic conditions conducive to understory oak, pine, and herbaceous species regeneration. Overall, these fire and fire surrogates (FFS) (B, M, MB and C) suggest a slow response in understory vegetation.

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CHAPTER ONE

UNDERSTORY RESPONSES TO 15 YEARS OF REPEATED FUEL REDUCTION TREATMENTS IN THE SOUTHERN APPALACHIAN MOUNTAINS, USA

Introduction

Historical records suggest that much of the southern Appalachian region was characterized as an open woodland with open canopies, little mid-story vegetation and a lush herbaceous understory (Ayers & Ashe 1905, Cronon 1983, Van Lear & Waldrop 1989, Denevan 1992). Frequent wildfire helped to create these conditions, which support the regeneration of fire-tolerant species, like oaks (*Quercus* spp.), yellow pines (*Pinus* spp.), and many herbaceous species (grasses, forbs, etc.) (Lorimer 1992). However, forest composition in the southern Appalachians has shifted from open woodlands to closed-canopy mesic forests due to decades of fire exclusion that began in the early 1900s (Lafon et al. 2017; Waldrop et al. 2008). This ecological shift, often called "mesophication", is the result of excess shade created by dense mesic thickets in areas that were previously open and dry, causing a cool, damp microclimatic effect in the understory (Nowacki & Abrams 2008). These conditions support the encroachment of fire-sensitive species such as red maple (*Acer rubrum*), yellow-poplar (*Liriodendron tulipifera*), American beech (*Fagus grandifolia*), black birch (*Betula lenta*), blackgum (*Nyssa sylvatica*), and black cherry (*Prunus serotina*) (Nowaki & Abrams 2008, Waldrop et al. 2008, Brose 2010; Brose & Waldrop 2014). Additionally, there has been an increase in flammable mid-story mesic species, like mountain laurel (*Kalmia latifolia*) and rhododendron (*Rhododendron* spp.), that create competition for more fire-adapted species, negatively impacting their regeneration success (Monk et al. 1985, Nowacki & Abrams 2008).

Fire exclusion has also led to increased midstory basal area, increased fuel loading, and forest homogenization (Brose et al. 2001, Brose & Van Lear 1998, Elliot et al. 1999). However, because of this region's extreme topography, varying soil types, and variable precipitation levels, it is one of the most complex landscapes to manage (Stanturf et al. 2002). Additionally, this region is a hotspot for exurban development, which makes wildfire risk one of the top concerns among managers (Olsen et al. 2017). The traditional method of forest management in the southern Appalachians has been dormant season (January - March) burning since fire was re-introduced to the landscape in the late 1980s (Van Lear & Waldrop 1991). Ideal burning conditions (wind, relative humidity, fuel moisture, etc.) exist in predictable cycles during the dormant season making planning easier and prescribed burning safer. Most dormant season prescribed fires tend to be low-intensity and do not fully prevent the growth of mesic hardwoods in the understory due to resprouting (Van Lear & Waldrop 1989). This is mainly because carbohydrate storage occurs in the roots during the dormant season which allows hardwood root stocks to remain viable following winter burns, resulting in prolific resprouting (Van Lear & Waldrop 1989, Elliott et al. 1999, Tift & Fajvan 1999, Gilbert et al. 2003).

Because many mesophytic species like red maple and mountain laurel are known to rapidly resprout after top-kill, using dormant season fire in this region can be problematic (Barnes & Van Lear 1998, Clinton et al. 1998, Waldrop et al. 2008, Brose et al. 2013). However, studies have reported that oaks and other hardwoods were better maintained with repeated burns or with treatments that resemble the effects of repeated burning, often called fire surrogates (Brose et al. 2013, Elliot et al. 1999, Brose et al. 1999). Additionally, fire surrogate regimes, like mechanical fuel reduction or mechanical fuel reduction and burning, may also be necssary to meet management objectives in heavily populated areas where prescribed burning may not be possible (Brose & Van Lear 1998; Waldrop et al. 2016). Recently established management objectives for the restoration of this region focus on fuel reduction for wildfire prevention, wildlife habitat promotion, and forest structure restoration (Waldrop et al. 2008).

Many studies have reported varied short and long-term effects of fuel reduction treatments on forest composition, structure and function (Arthur et al. 1998; Elliott et al. 2004; Mohr et al. 2004). For example, Dolan and Parker (2004) reported consistent vegetative patterns among three different site types (dry slopes, mesic slopes, and mesic uplands) following a single dormant season prescribed burn. Their results suggest that burning alone does not produce a sufficient amount of light for shade-intolerant seedlings or herbaceous species survival (Dolan & Parker 2004). Similarly, Clinton and others (1993) found a single site preparation fell and burn treatment to be ineffective at reducing *K. latifolia* regeneration in the mid and understory. However, felling and burning produced site conditions that were adequate for pine seed germination, and reduced *K. latifolia* vigor enough for various hardwood species to establish and grow (Clinton et al. 1993). Additionally, Iverson and others (2008) reported reductions in competitive hardwood species (*A. rubrum.* and *L. tulipifera*) and successful increases of oak and hickory densities after a combination of mechanical thinning and two late dormant season prescribed fires. However, these competitive species developed different strategies in dealing with new conditions created by mechanical thinning and burning, and thus suggest difficulty for long-term management strategies (Iverson et al. 2008). Due to the relatively new use of prescribed fire and alternative fuel reduction techniques throughout the southern Appalachian region, many knowledge gaps still exist for obtaining these management goals. These include: management impacts on wildlife species, managing oak regeneration, season of burn effects on forest composition, the effectiveness of surrogates as alternatives to fire treatments, and effects of long-term repeated fire and fire surrogate treatments on forest composition (Waldrop et al. 2016).

To address these knowledge gaps, a group of scientists and land managers from federal and state agencies, universities, and private entities started a program called the Fire and Fire Surrogate (FFS) study that has been implemented at various locations throughout the US (Washington to Florida) since the early 2000s (FFS Study Plan 2001; Youngblood et al. 2005; Schwilk et al. 2009). These research sites address the effects of alternative treatments, namely mechanical fuel reduction (M), prescribed burning (B), and combined treatments (MB), on various environmental factors in areas previously understood to be fire-dependent (FFS Study Plan 2001; Youngblood et al. 2005). Each site involves the measurement of environmental factors, including vegetation dynamics, fuel loading and fire behavior, soils, wildlife, entomology, pathology, and economics, to quantify the ecological tradeoffs of each treatment (FSS Study Plan 2001). At the southern Appalachian FFS study site, each treatment was designed to restore forest ecosystems by reestablishing natural ecosystem processes (B), stand structure (M), or both (MB) (Waldrop et al. 2016). This site is one of the longest-remaining FFS sites in the southeastern US, with a total of 4 dormant-season prescribed burns and 2 mechanical treatments conducted by 2016 (Greenberg et al. 2017). A body of literature has been produced from the southern Appalachian site including research on soil resources, woody species regeneration, fire behavior, fuel dynamics, and multiple wildlife populations in response to B, M, MB, and C treatments (Vose et al. 2004; Greenberg et al. 2007; Coates et al. 2010; Waldrop et al. 2010; Waldrop et al. 2016). However, there are very few published reports that focus on understory vegetation, more specifically long-term tree, shrub, and herbaceous species responses (Phillips & Waldrop 2008; Waldrop et al. 2016). Thus, the purpose of this study is to further investigate the long-term effects of 15 years of repeated B, M, and MB treatments on understory vegetation functional groups (i.e. tree, shrub, and herbaceous species) in the southern Appalachian Mountains.

I determined overall changes in understory response to repeated B, M, MB, and C treatments from the pre-treatment year (2001) to the post-treatment year (2016). Functional groups (i.e. pines, oaks, mesic hardwoods, shrubs, graminoids, and forbs) were used to assess trends of change in understory stem density and cover percent for all fuel reduction treatments. To explain significant changes occurring within the functional groups, I also identified various secondary functional group trends from 2001 to 2016. I focused on trends associated with white pines, yellow pines, red oaks, white oaks, *A. rubrum*, *B. lenta*, *F. grandifolia*, *L. tulipifera*, *N. sylvatica*, *P. serotina*, *K.latifolia*, *Rhododendron* spp., other ericaceous shrubs, non-ericaceous shrubs, annual grasses, perennial grasses, sedges, nitrogen-fixing forbs, non-nitrogen-fixing forbs, and ferns.

Methods

Location

The study area is located in Polk County, North Carolina on the Green River Game Land, which is managed for wildlife, public recreation, timber and other resources by the North Carolina Wildlife Resources Commission (NCWRC) (Appendix A-1). The Game Land covers 5,841 hectares and is classified as a mountainous region, where elevations range from about 300 m to 800 m. When the study was initiated in 2001, forests in the study area were about 80 years old to 120 years old and consisted of mixed- xeric or mesic *Quercus* and *Pinus* species depending on the topographic position. Shortleaf pine (*Pinus echinata*), pitch pine (*P. rigida*) and *P. virginiana* can be found on dry ridge tops while *P. strobus* can be found in moist coves. Ericaceous shrubs, like *Kalmia latifolia* and *Rhododendron maximum*, made up a dense mid-story layer throughout the study area (Waldrop et al. 2016). Most of the soils are of the Evard series (fine-loamy, oxidic, mesic, Typic Hapludults) in areas that can be described as moderately deep, well-drained, mountain uplands.

Study Design

The study design is a randomized complete block, which includes 4 treatments located within 3 replicate blocks, for a total of 12 treatment sites. The treatment sites cover an average of about 12 hectares (ha) each, which include a 4-ha buffer zone. Both the treatment site and the buffer zone received the same experimental treatment. Each treatment site incorporated many combinations of elevation, aspect, and slope in the landscape. Because landscape conditions were highly variable, these data were not separated during the analysis. Within each of the replicate blocks, 4 separate sites were randomly assigned to one of the 4 treatments: control (C), prescribed burning only (B), mechanical fuel reduction (M), and prescribed burning plus mechanical fuel reduction (MB) (Appendix A-2). The B treatment was applied 4 times by 2016 during the month of February or March (2003, 2006, 2012, and 2015); the M treatment was applied 2 times by 2016 during winter months (2001-2002 and 2011-2012); and the MB treatment application coincided with all of the M and B treatments (2001-2002, 2003, 2006, 2011-

2012, 2012, and 2015). All treatment areas were sampled in the pre-treatment year (2001), and all post-treatment measurements were taken during the following growing seasons, with the last measurements taken in 2016.

A 50 x 50 m grid was established in the treatment site, with grid points permanently marked and georeferenced. Ten 0.1 ha sample plots were established at randomly selected grid points within each treatment site to measure vegetation. Each plot is 50 m x 20 m and divided into 10 subplots, each 10 m x 10 m. Within each subplot, 2 1m^2 quadrats were established in the northwest and southeast corners to measure ground layer vegetation, using modified Whittaker plots (Keeley & Fotheringham 2005). The data for analyses includes all understory vegetation (<1.4 m tall) as recorded by species and abundance (in stems/ha or cover); this includes data for tree, shrub, and herbaceous species. Cover classes used in this sampling method were recorded as follows: 1 (<1%), 2 (1% - 10%), 3 (>10% - 25%), 4 (>25% - 50%), 5 (>50% - 75%), and 6 + (>75%). To generate workable cover class values for analysis, we used a percentage value that represents the median of the percent ranges for each cover class; for example, 5.5 % would be used for the cover class 2, etc.

Analysis

Each recorded species was grouped into a primary functional group (pines, oaks, mesic hardwoods, shrubs, graminoids, or forbs) to assess overall trends of understory vegetation change in response to treatments (B, M, MB, and C). The species within these primary groups were then grouped into a secondary functional group (yellow pines, white pines, red oaks, white oaks, *Kalmia latifolia* & *Rhododendron* sp., other ericaceous shrubs, non-ericaceous shrubs, annual grasses, perennial grasses, sedges, nitrogen-fixing forbs, non-nitrogen-fixing forbs, and ferns) to better explain any trends or drivers of vegetation change in response to treatments. The only primary functional groups that were broken down and analyzed at the species level was the mesic hardwoods and a portion of the shrubs, as these species are of management concern in this region. The mesic hardwoods and shrubs chosen were defined by Nowacki and Abrams (2008). An initial mixed-effects analysis of variance (ANOVA) was conducted for each of the primary functional groups to test the overall treatment effects on understory stem density (stems/ha) for woody species, and cover % for herbaceous species. These analyses were conducted on the change in stem density or cover % over time, which was found by taking the delta of the abundance data (2016-2001). A second ANOVA was conducted for all associated secondary functional groups to test the treatment effects within understory vegetation groups more specifically. The ANOVA model consists of treatment as the fixed variable, and grid point as the random variable nested within in each replicate block. Least squared means (LSM) comparisons were calculated from each model to determine significant treatment effects. Tukey's post hoc comparisons were then conducted for each model to determine differences among treatment effects. For all analyses, treatment effects were considered significant with an α of 0.05. All analyses were conducted using the lmerTest package in the program R (Kuznetsova et al. 2016).

Results

For oak, pine, mesic hardwood, and graminoid functional groups there was an increase in stems/ha and cover % from 2001 to 2016, in all treatments. Most of the woody functional groups experienced the largest increases of stems/ha in the MB treatment. However, mesic hardwood species experienced the largest increases of stems/ha in the B treatments, and shrub species experienced the largest increases of cover % in the M treatments. Graminoid and forb species experienced little to no change in cover % in all treatments.

Oaks

For the oak primary functional group, the ANOVA results showed a significant effect of treatment on understory stem change per ha from 2001 to 2016 ($F_{3,114}$; P_O < 0.01; Figure 1.1). The largest increases of oak regeneration (<1.4 m tall) were observed in MB (23,400.0 stems/ha) and B (21,700.0 stems/ha) treatments, and the smallest increases observed in C (13,835.0 stems/ha) and M (11,150.0 stems/ha) treatments. Tukey's post-hoc comparisons showed that only increases in MB treatments were different from the increases in M treatments.

At the secondary functional group level, white oaks showed larger overall increases in stems/ha than red oaks, however these increases were not statistically different between treatments ($F_{3,\ 1717.8}$; Table 1.1). Despite this, the largest increases in white oak stems/ha were observed in B (13,350.0 stems/ha), MB (12,616.7 stems/ha), and C (12,050.0 stems/ha) treatments, and the smallest increase was observed in M (6,583.3 stems/ha). Conversely, the effects of treatment on understory stem change per ha for red oaks was significant ($F_{3,\ 91.5}$; $P < 0.01$). Red oaks had the largest increases in the MB (10,785.0 stems/ha), B (8,350.0 stems/ha), and M treatments (4,565.0 stems/ha), and the smallest increase in C (1,835.0 stems/ha). Post-hoc comparisons of treatments showed that increases of stems/ha in MB and B were different from increases in C and M treatments.

Pines

For the pine primary functional group, the ANOVA results showed a significant effect of treatment on understory stem change per ha from 2001 to 2016 ($F_{3,130.79}$; P_{P} = 0.03; Figure 1.2). The largest increases of pine regeneration were observed in MB (850.4 stems/ha) and B (650.4 stems/ha) treatments, and the smallest increases observed in M (133.8 stems/ha) and C (53.8 stems/ha) treatments. Though there was a significant effect of treatment on pine stem density, Tukey's post-hoc comparisons showed that changes in stem density from 2001 to 2016 were similar for all treatments.

At the secondary functional group level, the ANOVA for yellow pines showed that there was a significant effect of treatment on understory stem change per ha ($F_{3, 117.8}$; P_{YP} = 0.01) (Table 1.1). The largest increases in yellow pine stems/ha were observed in MB (900.0 stems/ha) and B (716.7 stems/ha), and the smallest increases were observed in M (50.0 stems/ha) and C (16.7 stems/ha). Tukey's post hoc comparisons showed that only increases in MB were different from the increases in M and C. Conversely, the white pine ANOVA showed no significant effect of treatment on stem change per ha ($F_{3, 115.7}$). White pines had the largest increases in M (83.3 stems/ha) and C (33.3 stems/ha) and decreases in B (-66.7 stems/ha) and MB (-50.0 stems/ha).

Mesic hardwoods

For the mesic hardwood primary functional group, the ANOVA results showed a significant effect of treatment on understory stem change per ha from 2001 to 2016 ($F_{3,114.0}$; $P_{MHW} < 0.01$) (Figure 1.3). The largest increases of mesic hardwood regeneration (<1.4 m tall) were observed in B (24,066.7 stems/ha), followed by increases in M (6,716.7 stems/ha) and MB (6,700.0 stems/ha), with the smallest increase observed in C (500.0 stems/ha). Tukey's post-hoc comparisons showed that increases in B treatments were different from the increases in all other treatments.

At the secondary functional group level, the ANOVA for *B. lenta*, *F. grandifolia*, or *P. serotina* showed no effect of treatment on understory stem change per ha ($F_{3,\ 118.1}$; $F_{3,\ 104.0}$; $F_{3,\ 82.7}$), however, the ANOVA for *A. rubrum, N. sylvatica* and *L. tulipifera* significant effects of treatment on understory stem change per ha ($F_{3,\ 2136.1}$, $P_{AR} < 0.01$; $F_{3,\ 58.7}$, $P_{NS} < 0.01$; $F_{3,\ 329.5}$, $P_{LT} < 0.01$) (Table 1.1). *A. rubrum* showed larger increases in stems/ha than any other species, with the largest increases observed in B (20,650.0 stems/ha), followed by increases in M (6,300.0 stems/ha), MB (1,016.7 stems/ha), and C (150.0 stems/ha). Tukey's post-hoc comparison showed that increases in B treatments were different from increases in all other treatments. Similarly, *N. sylvatica* showed large increases in stems/ha in MB (3,505.2 stems/ha) and B (1,472.1 stems/ha) treatments, and smaller increases in C (448.50 stems/ha) and M (288.8 stems/ha) treatments. Tukey's post-hoc comparison showed that increases in MB were different from increases in all other treatments. *L. tulipifera* also showed large increases in stems/ha in MB (2,100.0 stems/ha) and B (1,566.7 stems/ha), followed by modest increases in M (200.0 stems/ha), and decreases in C (-83.3 stems/ha). Tukey's post-hoc comparison showed that only increases in MB were different from increases in C.

Shrubs

For the shrub primary functional group, the ANOVA results showed a significant effect of treatment on understory cover % change ($F_{3,\ 84.3}$; $P_S = 0.01$) (Figure 1.4). The largest increases in shrub cover % were observed in M (4.5 %), while decreases were observed in C (- 1.1 %), B (- 0.4 %), and MB (- 0.2 %) treatments. Tukey's post-hoc comparison showed that increases in M were different from increases in all other treatments.

At the secondary functional group level, only the ANOVA for *K. latifolia* & *Rhododendron* sp. showed a significant effect of treatment on understory cover % change ($F_{3, 114.0}$, P_{KL}< 0.01) (Table 1.1). The largest increase of cover % for *K. latifolia* & *Rhododendron* sp. was observed in M (10.0 %), followed by increases in C (2.3 %) and MB (0.5 %), and decreases in B (- 3.2 %). Tukey's post-hoc comparisons showed that increases of cover % in M were different from all other treatments. Although the ANOVA showed no significant effect of treatment on change in cover % of other ericaceous shrubs, decreases in cover % were observed in C treatments (-3.3 %), and little to no increases in cover % were observed in MB (1.0 %), M (0.6 %), and B (0.0 %) treatments ($F_{3, 114}$). Similarly, the ANOVA for non-ericaceous shrubs showed no significant effect of treatment on change in cover %, however, increases were observed in MB (5.4 %), M (3.4 %), and B (3.2 %) treatments, and decreases were observed in C (-1.2 %) ($F_{3, 82.2}$).

Graminoids

For the graminoid primary functional group, the ANOVA results showed no significant effect of treatment on understory cover % change from 2001 to 2016 ($F_{3, 92.1}$) (Figure 1.5).

At the secondary group level, only the ANOVA for sedges showed a significant effect of treatment on cover % change ($F_{3. 91.0}$; P_{sedge}< 0.01) (Table 1.1). Increases of sedge cover % were observed in MB (2.4 %) and B (1.3 %), followed by little change observed in M (0.3 %) and C (-0.2 %). Tukey's post-hoc comparison showed that only increases in MB were different from increases in M and decreases in C treatments. Although the ANOVA for perennial grasses showed no significant effect of treatment on cover % change, increases of cover % were observed in MB (2.7 %), B treatments (1.7 %), and C (1.5 %) treatments, while little to no change was observed in M (0.0 %) treatments. Similarly, the ANOVA for annual grasses showed no significant effect of treatment on cover % change ($F_{3, 114}$), as there was little to no change was observed for all treatments: M (0.6 %), MB (0.2 %), C (0.0 %), and B (0.0 %).

Forbs

For the forb primary functional group, the ANOVA results showed no significant effect of treatment on understory cover % change from 2001 to 2016 ($F_{3,116}$; Figure 1.6).

At the secondary functional group level, only the ANOVA for ferns showed a significant effect of treatment on cover % change ($F_{3, 114}$; $P_{fern} = 0.02$) (Table 1.1). Modest increases in fern cover % change were observed in MB (1.4 %), M (0.5 %), and B (0.1 %), while decreases in cover % change were observed in C (-3.2 %). Tukey's post-hoc comparison showed that only increases in MB were different from decreases in C treatments. While the ANOVA for nitrogen-fixing forbs and non-nitrogen-fixing forbs did not show a significant effect of treatment on cover % change, increases in nitrogen-fixing forbs were observed in all treatments: MB (1.5 %), M (1.4 %), B (0.7 %), and C (0.1 %). Additionally, increases in non-nitrogen-fixing forbs were observed in MB (1.4 %), B (1.2 %), and M (0.2 %) treatments, while decreases were observed in C (-0.6 %) treatments.

Discussion

Oaks

Overall increases in understory oak stems for all treatments indicates a high potential for oak growth to the mid and over-story canopy layers. However, this potential growth is negatively affected by the presence of competitive species, low light availability, and heavy amounts of litter and duff cover preventing germinant growth (Lorimer 1992). The highest increases in understory oak stems was observed in MB, followed closely by B treatments, with the least amount of regeneration observed in M, suggesting that burning is necessary for increased oak regeneration. This is likely because it improves seedbed conditions by decreasing litter and duff cover and by also decreasing competitive hardwood species (Tift & Fajvan 1999). However, our findings suggest that burning alone seems to only temporarily reduce the amount of shade-tolerant hardwood species, which may not be sufficient for oak seedlings to grow competitively (Gilbert et al. 2003, Iverson et al. 2008, Hutchinson et al. 2005, Brose 2010, Waldrop et al. 2016). Brose (2010) reports that burning improved seedbed conditions by decreasing litter cover but did not reduce competitive hardwood species enough to sustain oak growth. Additionally, Phillips et al. (2010) reported comparable results to ours with more increases in oak regeneration in MB than B due to large canopy openings from overstory mortality, increasing light availability to the forest floor. MB treatments often result in higher-intensity fires than B treatments, which can cause a greater reduction in litter cover and midstory basal area (BA). Previous reports from this study site showed the lowest litter and duff loading and largest reductions in basal area in MB compared to B, C, and M treatments (Waldrop et al. 2008, Waldrop et al. 2016). White oak species were observed to have more stem increases over red oak species, with the most prevalent increases in MB followed closely by C. This may be the result of a substantial white oak masting event that occurred prior to the second prescribed burn, as reported by Jones (2005) (Phillips et al. 2010).

Pines

Regeneration of yellow pine species increased the most after MB treatments, which closely follows results from related studies also conducted in the southern Appalachian region (Waldrop et al. 2008, Pile & Waldrop 2016, Waldrop et al. 2016). These studies suggest that B treatments create favorable microsite conditions for yellow pines that include decreased duff and litter depths that allow germination in mineral soil and increased light availability by reducing competitive seedling density. However, the slight increases in M and significant increases in MB suggest that additional reductions in competitive hardwood species are also necessary for successful yellow pine regeneration (Vose et al. 1995, Vose et al. 1997, Waldrop & Brose 1999, Elliott & Vose 2005, Reilly et al. 2017). Jenkins et al. (2011) reported increased yellow pine seedling density after total fuel reductions exceeded 60% in B treatment areas. However, greater increases in yellow pine seedling density were observed in M treatments that led to understory reductions exceeding 80%. These results – in concert with our findings – suggests that drastic structural changes are needed to reduce competition enough to enhance yellow pine regeneration (Jenkins et al. 2011). Our findings suggest that white pine reduction was achieved by both B and MB treatments, but more stem reduction occurred in B treatments. Elliott et al. (2002) found that competition does not strongly affect white pine growth, however, a strong negative relationship is present between white pine regeneration and litter consumption during a fire. When young, pine regeneration is strongly inhibited by fire due to the lack of bark and root development. However, white pines are especially inhibited by fire during the germination process, as most germinants originate in moist fuel masses rather than in mineral soil (Elliott & Vose 2005). This growth strategy makes this species more susceptible to germinant mortality following fire treatments, which was observed in our results.

Mesic Hardwoods

Most studies conducted in mixed-oak forests report dominance or co-dominance of mesic hardwood species in the understory and midstory resulting from fire suppression (Brose & Van Lear 1998, Gilbert et al. 2003, Elliott & Vose 2005, Hutchinson et al. 2008). These species are often characterized by their fire-sensitivity, shade tolerance, and ability to grow quickly in mesic microsites, making them some of the largest competitors of oak regeneration (Brose & Van Lear 1998, Elliott et al. 1999, Tift & Fajvan 1999). Our data coincides with several studies conducted in this region, which report the most dominant species observed among study sites are *A. rubrum, L. tulipifera,* and *N. sylvatica* (Nowacki & Abrams 2008, Waldrop et al. 2006, Waldrop et al. 2008, Phillips et al. 2010, Waldrop et al. 2016). The overall increase in mesic hardwood stems for all treatments suggests these species respond positively to disturbance, with the most drastic response led predominantly by *A. rubrum* in B treatments. However, our results contradict much existing research that have reported decreases in *A. rubrum* stem density following one or two B treatments (Elliott et al. 1999, Hutchinson et al. 2005, Iverson et al. 2008, Jenkins et al. 2011, Reilly et al. 2017). This may be because most *A. rubrum* stems originated as sprouts, which are more capable of rapid height growth than new seedlings, causing more advanced prolific sprouting from existing roots (Tift & Fajvan 1999, Beck & Hooper 1986, Palik & Pregitzer 1992). Our results suggest that frequent dormant season burning may also be creating more favorable microsite conditions for mesic hardwood species by decreasing litter and duff and allowing root growth into mineral soil (Hammond et al. 2015, Gilbert et al. 2003). The smallest stem increase was observed in C and MB treatments, indicating either transfer of understory stems into the midstory, slow growth of stems due to competition in C, or mortality of stems from mechanical removal in MB. *L. tulipifera* and *N. sylvatica* were much less abundant before any of the treatments but increased following MB treatments. Phillips et al. (2010) reports scarcity of *L. tulipifera* prior to treatments due to their light-demanding nature and the lack of a large disturbance. They report that B allowed some germination in mineral soil but did not provide enough light for sustained growth of these species (Phillips et al. 2010). This suggests that these species will only experience sustained growth with a canopy-opening disturbance, as seen in MB treatments and reflected in our results.

Shrubs

We observed increases in shrub cover following repeated M treatments, and little to no change in the other treatments. This is primarily due to characteristic resprouting in *K. latifolia* and *Rhododendron* sp. (McGinty 1972, Phillips & Waldrop 2008). The most abundant ericaceous shrub species in all treatments, *Vaccinium* spp. and *K. latifolia* (in both 2001 and 2016), suggests vigorous resprouting as reported in Waldrop et al. (2010 & 2016), and Phillips and Waldrop (2008). Some short-term studies report immediate decreases in hardwood shrubs and saplings after thinning (Waldrop et al. 2008, Phillips et al. 2010), however, these results are not consistent with ours and were likely temporary at these sites. Our long-term findings coincide with many shorter-term studies that also report decreases in ericaceous shrub cover following one or two burn treatments (Elliott et al. 1999, Hutchinson et al. 2005, Phillips & Waldrop 2008, Iverson et al. 2008, Jenkins et al. 2011). Other non-ericaceous shrub species were not highly abundant in our plots prior to treatments and only showed minor increases in cover % among all treatments, therefore they were not a large component of the understory stem cover from 2001 to 2016.

Graminoids

Overall, increases in graminoid cover observed for all treatments is consistent with other studies conducted in this region, which report moderate but non-significant increases in cover following a disturbance (Elliott et al. 1999, Thomas et al. 1999, Zenner et al. 2006, Brockway et al. 1998). However, Phillips & Waldrop (2008) observed significant increases in graminoid cover following MB and B treatments, specifically observed in grass species. Similarly, Elliott et al. (1999) concluded that burning significantly stimulates both growth and fruiting of herbaceous understory vegetation. Our increases, though moderate, were observed mostly in MB treatments, indicating possible limitations associated with B and M treatments and life history characteristics. For example, lack of regeneration in vegetation can be due to a lack of seed source, as seeds may be absent in the seedbank, seed production may be low, or source populations may be scarce (Hille Ris Lambers et al. 2005). Our findings also suggest that conditions required for germination are not optimal under B, M, and C treatments. For example, Shiffman & Johnson (1992) reported finding an extremely sparse dispersal of herbaceous seeds in the seedbank, which was due to significant seed accumulation in the humus layer, rather than the mineral soil, causing seeds to decay from excess moisture. In our study site, fire exclusion could have had a similar effect on graminoid seed decay.

Additionally, studies in similar areas report that fire did not completely consume the humus layer, creating more evidence for seed decay at our study site (Boerner et al. 2007, Waldrop et al. 2007).

Forbs

Like graminoids, our findings show that forbs did not significantly change from 2001 to 2016. We observed increases in B, M, and MB for all three forb subgroups (nitrogen-fixing forbs, non-nitrogen-fixing forbs, and ferns), suggesting that successful forb regeneration requires either a large disturbance, or repeated small disturbances, that improve growth conditions incrementally. Other studies report that the most significant increases occur immediately following more intensive treatments (B and MB) that often result in canopy openings and large midstory stem reductions (Arthur et al. 1998, Ducey et al. 1996, Phillips & Waldrop 2008, Elliott et al. 1999). Our findings suggest that drastic structural changes create more available space and sunlight, which promotes the growth of forb species. However, both our findings and results from Waldrop et al. (2016) suggest that rapid regeneration of competitive woody species decreases light availability on the forest floor, and subsequently slows forb growth until the next treatment. This suggests that larger structural changes, resulting in sustained light availability and other micro-site modifications, will be required for herbaceous vegetation to become more permanent residents of the community.

Conclusions

This study provides results on the effects of long-term repeated fire and fire surrogate treatments on understory vegetation, as needed by southern Appalachian forest managers. We found M to be successful in reducing mesic hardwood stems, but unsuccessful in reducing other undesirable species, like ericaceous shrubs and white pine, and it did not significantly change herbaceous cover. B produced significant increases in oaks, pines, and graminoid cover, while also reducing ericaceous shrub cover and white pine stems. However, B also promoted significant increases of red maple stems, and little to no herbaceous species regeneration. MB treatments increased oak, yellow pine, non-ericaceous shrub, graminoid, and forb regeneration, and reduced ericaceous shrub cover, and red maple and white pine stems.

With the exception of C, each of the treatments satisfy a subset of the management objectives by reducing mesic hardwood stem density (M and MB) and ericaceous shrub cover (B and MB), and increasing oaks, pines, and a lush herbaceous understory (B and MB). However, MB treatments produced effects most consistent with our restoration goals over all other treatments. The treatments creating the largest structural changes are likely to produce the most favorable results for oak, pine, and herbaceous species regeneration, as observed in B and MB treatments. Furthermore, management implications should be catered to specific objectives set by the landowners and managers involved. Continuing this and other FFS studies will improve our understanding of long-term forest restoration. However, additional research will be required to explore alternative management strategies, and better explain the changes that have occurred in the southern Appalachian landscape. This may include comparing growing season burns with dormant season burns, and seedbank inventories.

TABLES

Table 1.1: Summary statistics of treatment effects (Control = C, Burned = B, Mechanical removal of stems <10 cm dbh = M, and combination of M + B = MB) used within the secondary functional group analyses of variance (ANOVAs). These secondary functional groups included: red and white oak stems/ha; white and yellow pine stems/ha; 6 mesic hardwood species stems/ha; ericaceous and non-ericaceous shrub % cover; annual grass, perennial grass, and sedge % cover; and nitrogen-fixing forb, non-nitrogen-fixing forb, and fern % cover in 2001 and 2016 in the Green River Game Land, NC. Treatments with different lowercase letters were statistically different from one another, in terms of stem count change, during the study period.

Primary Functional Group	**Secondary Functional Group**	**Treat-ment**	**Stems/ha in 2001**	**Stems/ha in 2016**	**Change in stems (Avg/ha)**
Oaks	Red Oaks	C	2400.0	4183.3	1835.0 (SD ± 2,100.0) a
		B	3466.7	11816.7	8359.0 (SD ± 7,050.0) b
		M	3033.3	7600.0	4565.0 (SD ± 3,450.0) a
		MB	3783.3	14566.7	10785.0 (SD ± 6,450.0) b
	White Oaks	C	2500.0	14550.0	12050.0 (SD ± 16,518.0)
		B	1633.3	14983.3	13350.0 (SD ± 11,827.9)
		M	1750.0	8333.3	6583.3 (SD ± 8,381.1)
		MB	2466.7	15083.3	12616.7 (SD ± 12,792.6)
Pines	White Pines	C	0.0	33.3	33.3 (SD ± 126.9)
		B	116.7	50.0	-66.7 (SD ± 340.7)
		M	66.7	150.0	83.3 (SD ± 437.1)
		MB	50.0	0.0	-50.0 (SD ± 152.6)
	Yellow Pines	C	0.0	16.7	16.7 (SD ± 91.3) a
		B	50.0	766.7	716.7 (SD ± 1,243.5) ab
		M	16.7	66.7	50.0 (SD ± 201.3) a
		MB	0.0	900.0	900.0 (SD ± 2,127.0) b
Mesic Hardwoods	*Acer rubrum*	C	9866.7	10016.7	150.0 (SD ± 8,273.5) a
		B	13766.7	34416.7	20650.0 (SD ± 25,490.4) b
		M	4550.0	10850.0	6300.0 (SD ± 9,201.4) a
		MB	8616.7	9633.3	1016.7 (SD ± 7,082.6) a
	Betula lenta	C	0.0	33.3	33.3 (SD ± 126.9)
		B	0.0	150.0	133.3 (SD ± 730.3)
		M	0.0	0.0	0.0 (SD ± 0.0)
		MB	0.0	16.7	16.7 (SD ± 91.3)

	Fagus grandifolia	C	0.0	0.0	0.0 (SD ± 0.0)
		B	16.7	50.0	0.0 (SD ± 0.0)
		M	16.7	16.7	0.0 (SD ± 0.0)
		MB	0.0	33.3	33.3 (SD ± 182.6)
	Liriodendron tulipifera	C	650.0	583.3	-83.33 (SD ± 558.4) a
		B	1050.0	5033.3	1566.7 (SD ± 2,718.8) bc
		M	333.3	433.3	200.0 (SD ± 783.5) ab
		MB	400.0	2550.0	2100.0 (SD ± 3,521.9) c
	Nyssa sylvatica	C	233.3	750.0	448.5 (SD ± 720.2) a
		B	1183.3	4666.7	1472.1 (SD ± 1,984.8) a
		M	466.7	750.0	288.8 (SD ± 805.2) a
		MB	350.0	3633.3	3505.5 (SD ± 3,531.8) b
	Prunus serotina	C	266.7	250.0	-17.2 (SD ± 608.6)
		B	433.3	850.0	230.3 (SD ± 817.2)
		M	200.0	83.3	-86.37 (SD ± 373.3)
		MB	166.7	266.7	63.6 (SD ± 468.6)
Shrubs	*Kalmia latifolia* & *Rhododendron* spp.	C	16.0	18.3	2.3 (SD ± 12.0) a
		B	15.0	11.8	-3.1 (SD ± 9.7) a
		M	15.8	25.8	10.0 (SD ± 12.5) b
		MB	15.0	15.4	0.4 (SD ± 12.0) a
	Other Ericaceous Shrubs	C	11.9	8.6	-3.3 (SD ± 7.9)
		B	10.1	10.1	0.0 (SD ± 7.1)
		M	11.7	12.3	0.6 (SD ± 6.4)
		MB	14.1	15.1	1.0 (SD ± 8.5)
	Non-Ericaceous Shrubs	C	5.6	4.4	-1.2 (SD ± 8.2)
		B	5.9	9.1	3.2 (SD ± 8.3)
		M	2.8	6.2	3.4 (SD ± 12.8)
		MB	7.6	13.0	5.4 (SD ± 12.1)
Graminoids	Annual Grasses	C	0.0	0.0	0.0 (SD ± 0.1)
		B	0.0	0.0	0.0 (SD ± 0.0)
		M	0.0	0.6	0.6 (SD ± 3.3)
		MB	0.0	0.2	0.2 (SD ± 1.0)
	Perennial Grasses	C	1.7	3.2	1.5 (SD ± 7.4)
		B	0.7	2.4	1.7 (SD ± 2.5)
		M	0.7	0.7	0.0 (SD ± 1.9)
		MB	0.9	3.6	2.7 (SD ± 2.7)
	Sedges	C	0.4	0.2	-0.2 (SD ± 1.0) a

		B	0.6	1.9	1.3 (SD ± 2.5) ab
		M	0.7	1.0	0.3 (SD ± 1.7) a
		MB	0.5	3.0	2.5 (SD ± 3.8) b
Forbs	Nitrogen-fixing Forbs	C	0.8	1.0	0.1 (SD ± 2.5)
		B	0.0	0.7	0.7 (SD ± 1.7)
		M	0.0	1.4	1.4 (SD ± 3.6)
		MB	0.0	1.5	1.5 (SD ± 2.4)
	Non-Nitrogen-fixing Forbs	C	3.9	3.4	-0.6 (SD ± 2.8)
		B	3.1	4.3	1.2 (SD ± 3.1)
		M	3.6	3.7	0.2 (SD ± 3.7)
		MB	4.3	5.7	1.3 (SD ± 4.2)
	Ferns	C	4.0	0.8	-3.2 (SD ± 8.7) a
		B	1.6	1.7	0.1 (SD ± 4.1) ab
		M	1.0	1.7	0.5 (SD ± 3.2) ab
		MB	2.2	3.4	1.4 (SD ± 6.7) b

FIGURES

1.1

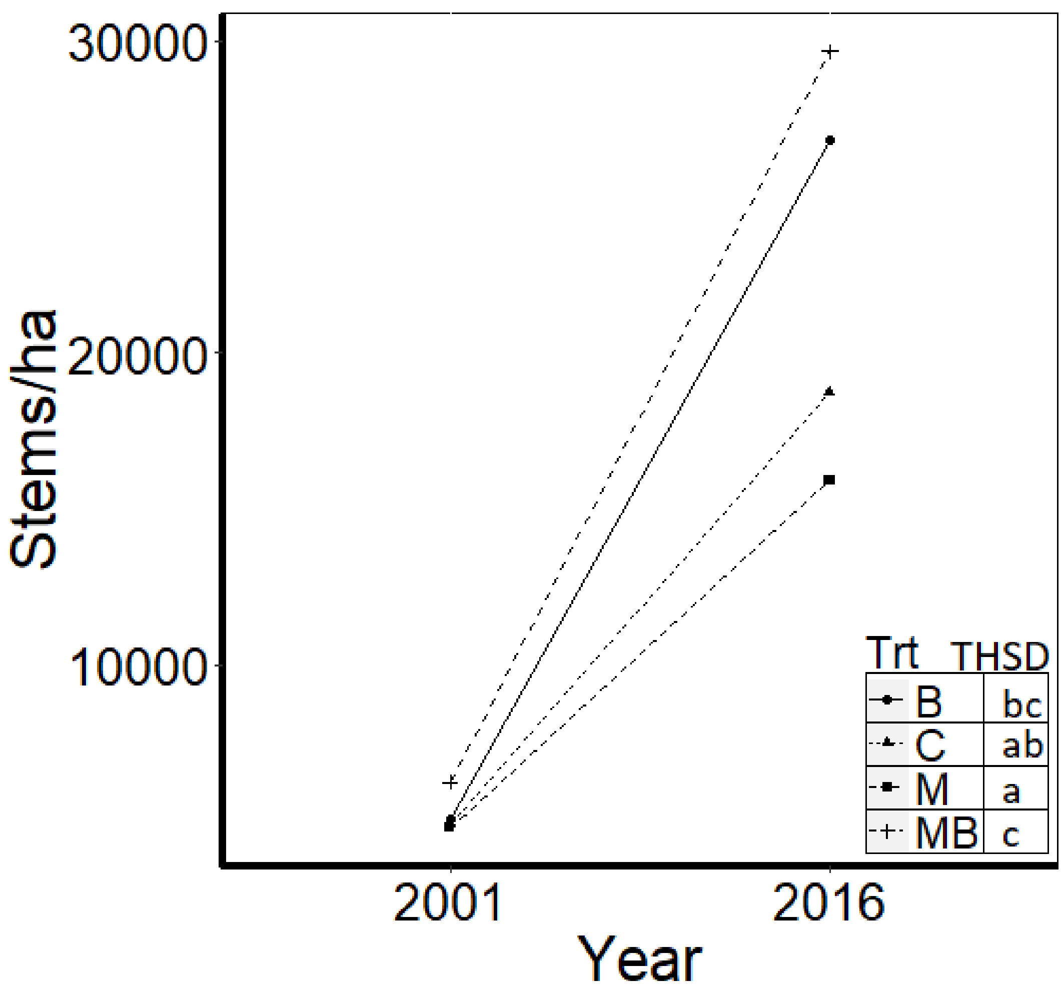


Figure 1.1: Results from the oak primary functional group analysis of variance (ANOVA) showing overall increases of stems/ha from the pre-treatment year (2001) to post-treatment year (2016). Post-hoc comparison (Tukey's Honest Significant Difference Test; THSD) letters show differences in stem increases only in mechanical treatment (M).

Figures (Continued)

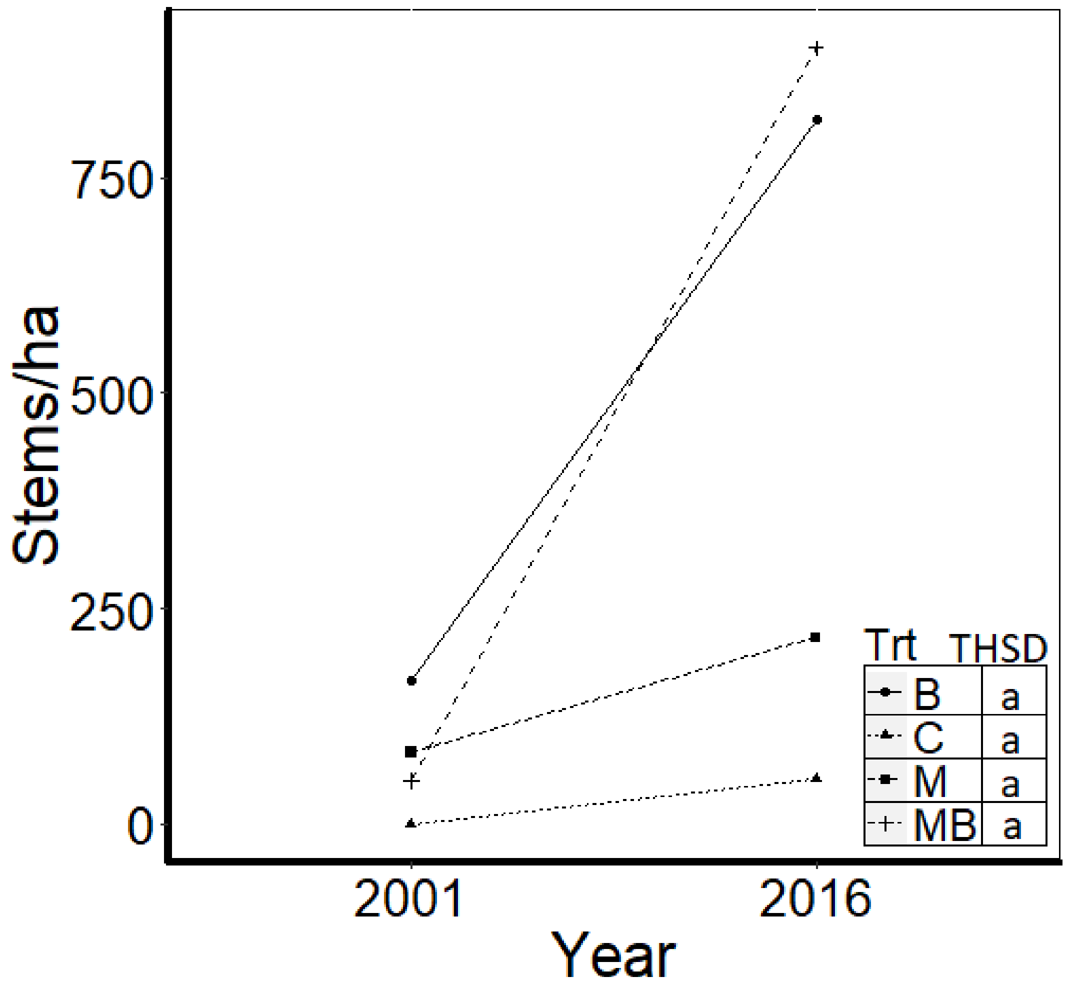


Figure 1.2: Results from the pine primary functional group analysis of variance (ANOVA) showing overall increases of stems/ha from the pre-treatment year (2001) to post-treatment year (2016). Post-hoc comparison (Tukey's Honest Significant Difference Test; THSD) letters show no differences of increases in pine stems/ha among all treatments (B, C, M, and MB).

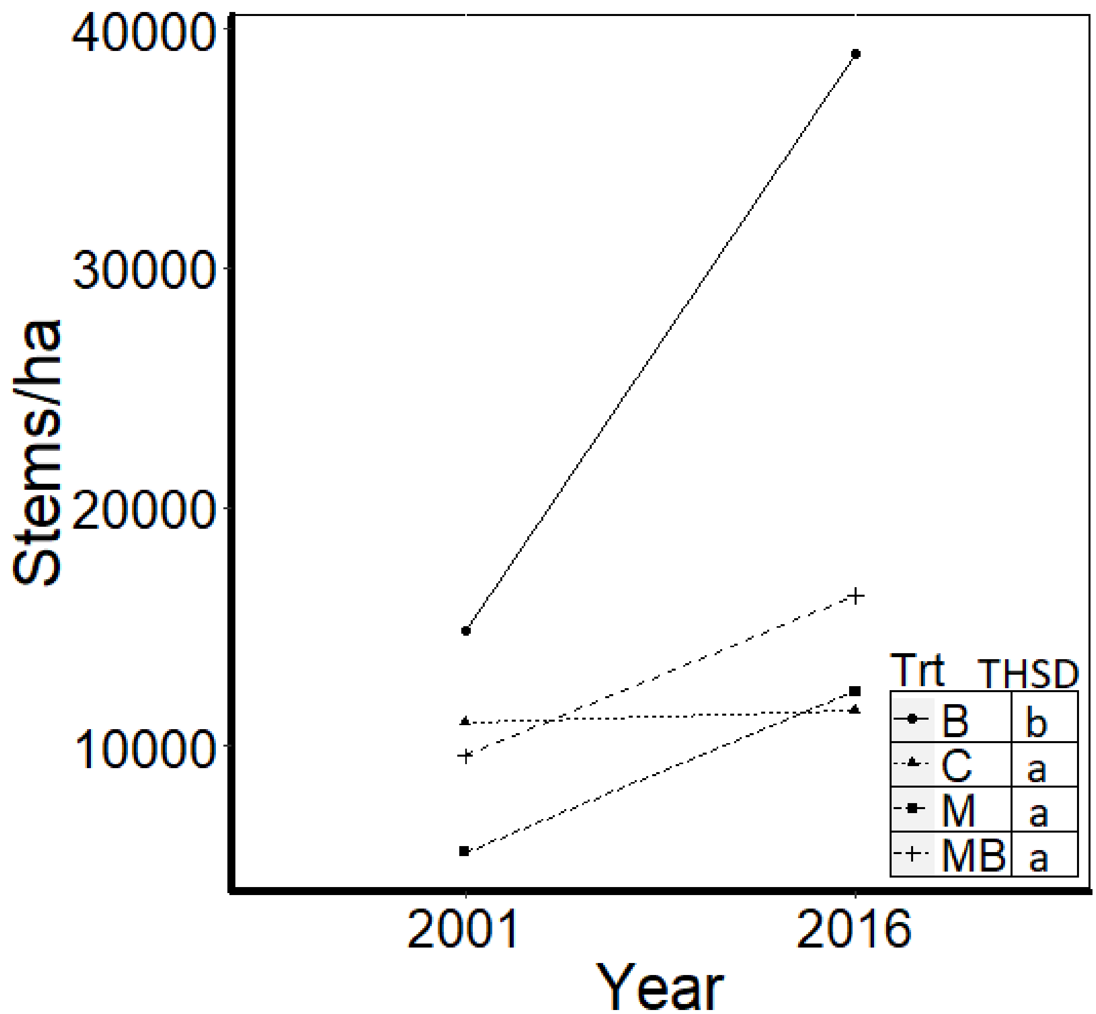


Figure 1.3: Results from the mesic hardwood primary functional group analysis of variance (ANOVA) showing overall increases of stems/ha from the pre-treatment year (2001) to post-treatment year (2016). Post-hoc comparison (Tukey's Honest Significant Difference Test; THSD) letters show and differences in stem increases between burning (B) and all other treatments (C, M, and MB).

Figures (Continued)

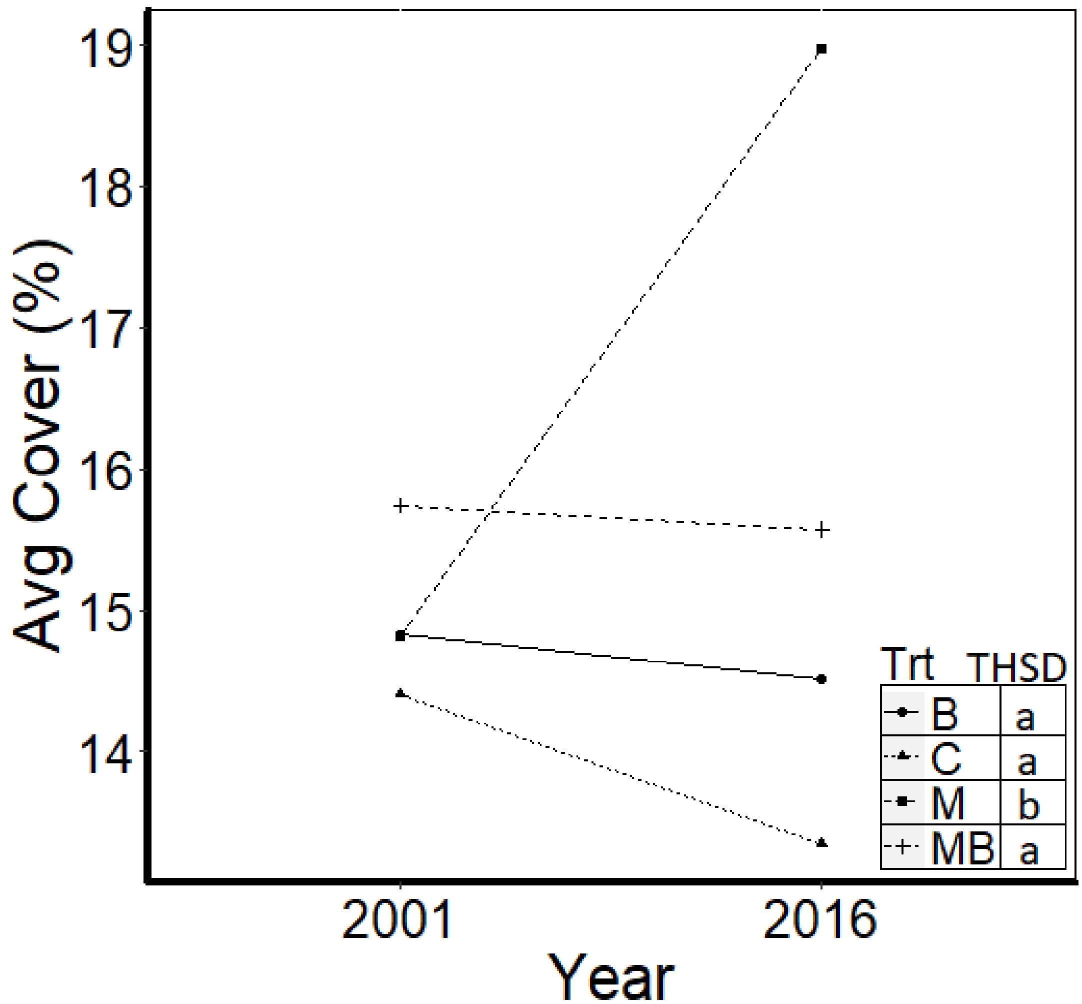


Figure 1.4: Results from the shrub primary functional group analysis of variance (ANOVA) showing significant cover % increases in M treatments from the pre-treatment year (2001) to post-treatment year (2016). Post-hoc comparison (Tukey's Honest Significant Difference Test; THSD) letters show differences between the shrubs cover % increases in the mechanical treatment (M) from all other treatments (B, C, and MB).

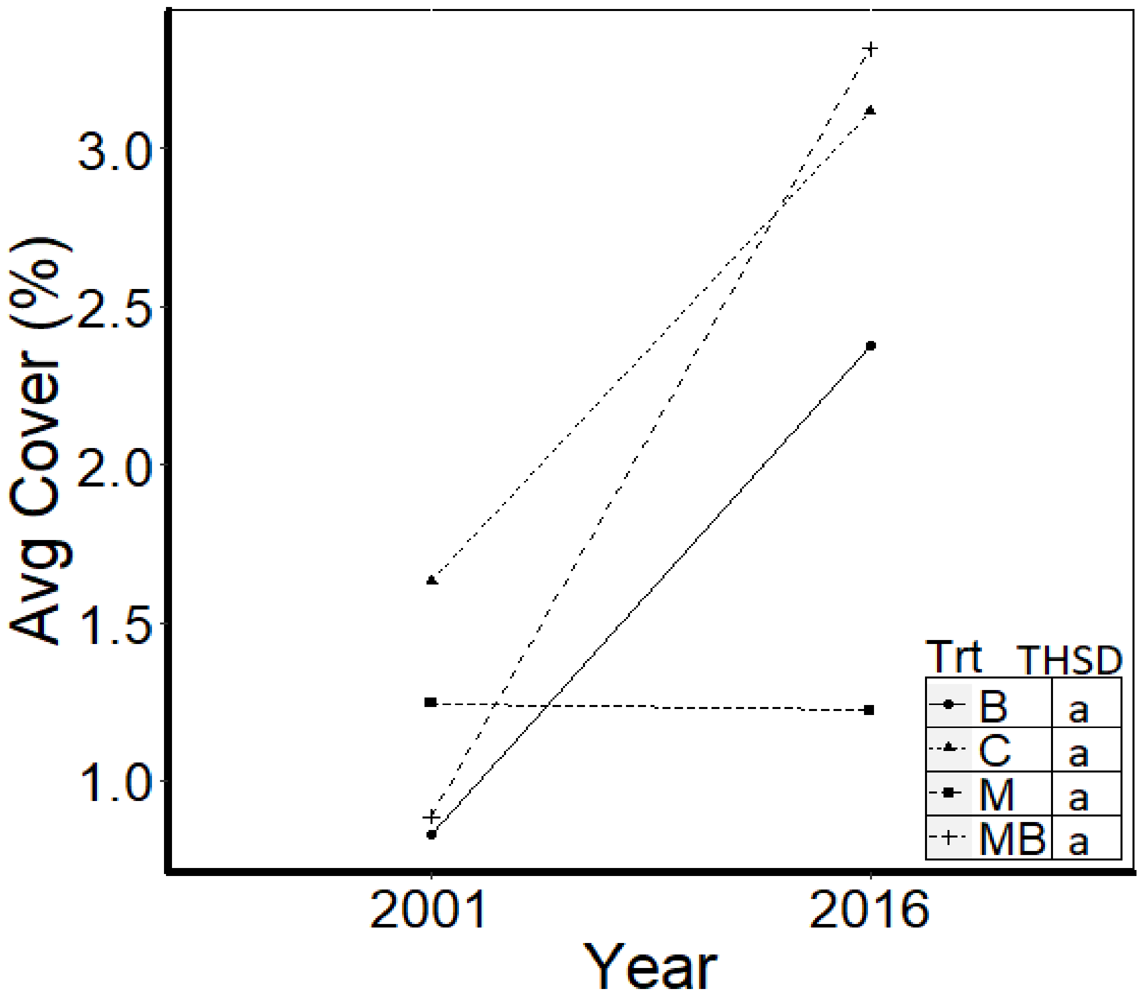


Figure 1.5: Results from the graminoid primary functional group analysis of variance (ANOVA) showing significant cover % increases in MB, B, and C treatments from the pre-treatment year (2001) to post-treatment year (2016). Post-hoc comparison (Tukey's Honest Significant Difference Test; THSD) letters show no differences in cover % increases among all treatments (B, C, M, and MB).

Figures (Continued)

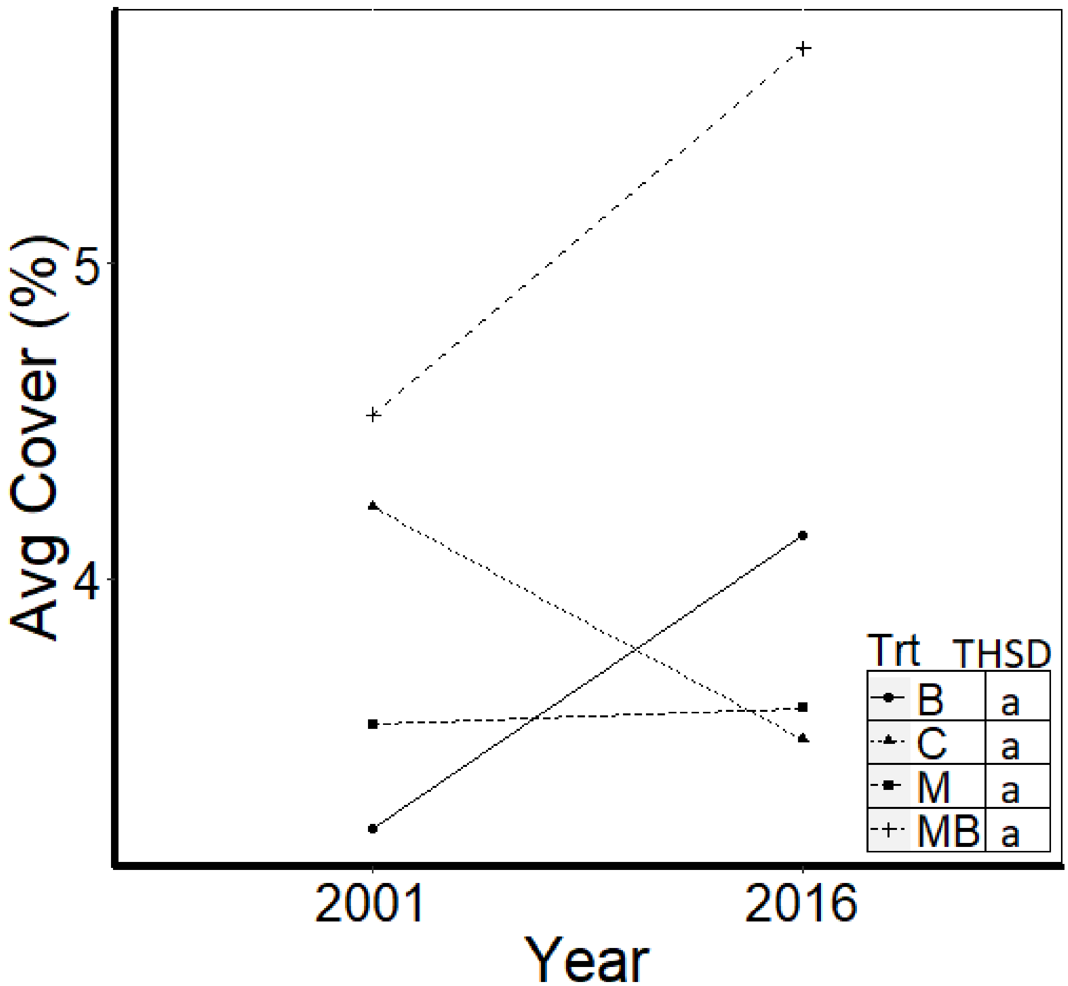


Figure 1.6: Results from the Forb primary functional group analysis of variance (ANOVA) showing significant increases in MB and B treatments from the pre-treatment year (2001) to post-treatment year (2016). Post-hoc comparison (Tukey's Honest Significant Difference Test; THSD) letters show no differences in cover % increases among all treatments (B, C, M, and MB).

CHAPTER TWO

UNDERSTORY COMMUNITY SHIFTS FOLLOWING 15 YEARS OF REPEATED FUEL REDUCTION TREATMENTS IN THE SOUTHERN APPALACHIAN MOUNTAINS, USA

Introduction

Natural and anthropogenic disturbance regimes influence forest ecosystems by altering resource distribution, which changes vegetation structure and composition across a range of spatial and temporal scales (Pickett et al. 1999; Willig & Walker 1999; Certini 2005). Variations in disturbance type, intensity and frequency often create mosaics in a landscape, increasing community-level heterogeneity (Willig and Walker 1999; Greenburg et al. 2016). Repeated disturbance has shaped many of these complex ecosystems over time, often resulting in the development of disturbance-dependent forest communities (Horn 1974).

Among disturbance regimes, fire and its effects on plant community assemblage in forested ecosystems have been well documented (Rogers 1996; Johnstone 2006; Stark et al. 2006; van Leeuwen et al. 2010; Kim & Holt 2012). Some roles of fire in forested ecosystems include the creation and maintenance of early successional communities, the control of fire sensitive-species, and the promotion of landscape-level community heterogeneity (Horn 1974; Pyne 1997; Weber & Flannigan 1997). Historical records suggest that the consistent presence of fire in a forested landscape favors the establishment, growth, and dominance of certain plant species (Wright & Bailey 1982; Greenburg et al. 2016; Lafon et al. 2017). Thus, fire acts as a selective pressure that has driven the evolution of adaptations and/or physiological strategies that make them more resilient to – if not dependent upon – burning (Pyne 1997). Many vegetative species possess traits that make them less susceptible to fire-related mortality. For example, thick bark, hypogeal germination, large root-shoot ratios, and heat-resistant seeds indicate characteristics of fire tolerance (Lorimer 1992; Pyne 1997; Hutchinson et al. 2005). Other species possess traits that benefit from fire, like serotinous/semi-serotinous cones and fire-stimulated flowering and seed production, which indicate fire-dependency (Abrams et al. 1995; Bourg & Gill 2000; Mohr et al. 2002; Fitzgerald 2005). Over time, these traits have allowed certain species to remain dominant in many fire-maintained forest communities.

Within the last several centuries, fire regime changes led to a compositional shift in many fire-dependent forests. During this time, frequent low-intensity fires were long used by Native Americans and early European colonizers (Brose et al. 2001; Schwilk et al. 2009). European settlement in the late 1800’s was followed by a short period of high-intensity stand-replacing fires from heavy logging practices (Lorimer 2001; Pausas & Keeley 2009). This brought about a period of fire suppression that began in the early 1900’s and persisted until the early 1970’s (Guyette et al. 2002; van Wagtendonk 2007). Since the suppression movement began, many terrestrial ecosystems throughout the US have experienced increased fuel loading, increased wildfire risk, changes in community composition, and landscape-level homogenization (Arno & Gruell 1986; Baker 1992; Huddle & Pallardy 1999; Taylor 2007). In the United States, one of the regions most heavily affected by fire suppression is the southern Appalachian Mountain region (Garren 1943; Greenburg et al. 2016).

Characterized by extreme topography, varying soil types, and large precipitation gradients, the southern Appalachian region is supports high levels of biodiversity and heterogeneity (Van Lear & Waldrop 1989; Waldrop et al. 2014). However, these complexities – in concert with extensive exurban development – also make the southern Appalachians one of the most difficult regions to manage with fire (Stanturf et al. 2002). Due to the fire suppression movement, forests in the southern Appalachians have experienced increases in midstory and overstory density, excess fuel loading, oak and yellow pine regeneration failure, forest homogenization, and encroachment of mesic hardwood species (Brose & Van Lear 1998; Elliott et al. 1999). Most structural shifts documented in this region are from pine-hardwood open woodlands to predominantly closed-canopy mesic forests (Waldrop et al. 2008). These open woodlands were typically comprised of fire-dependent species, such as oaks (*Quercus* spp.), yellow pines (*Pinus* spp.), and various herbaceous vegetation (sedges, grasses, and forbs) (Ayers & Ashe 1905). Conversely, contemporary mesic hardwood forests support the growth of undesirable fire-sensitive species like red maple (*Acer rubrum*) and yellow-poplar (*Liriodendron tulipifera*) (Nowacki & Abrams 2008). Additionally, there have been increases of flammable midstory mesic shrubs like mountain laurel (*Kalmia latifolia*) and rhododendron (*Rhododendron* spp.), which outcompete fire-dependent species and create potential for higher intensity fires (Monk et al. 1985; Brose et al. 2001).

Despite a majority of vascular plant diversity being found in the understory layer, this forest component is comparatively understudied in southern Appalachian forests (Gilliam & Roberts 2003). Understory vegetative communities in this region can be comprised of a wide array of tree, shrub and herbaceous species regeneration (Waldrop et al. 2016). This understory community is largely shaped by topography-related microsite characteristics that include abiotic factors like microclimate, soil moisture, and soil fertility (Hutchinson et al. 1999). Repeated fires of varied intensity and severity often contribute to fluctuations in these microsite characteristics by increasing light availability from canopy openings, which alter understory community assembly (Small & McCarthy 2002; Gilliam & Roberts 2003). However, many biotic factors also contribute to the formation of understory community assembly (Azeria et al. 2011). For example, species-specific characteristics, like shade intolerance or prolific resprouting abilities, can influence the dynamics of post-fire successional community assemblage (Van Lear et al. 2000; Hutchinson et al. 2005).

Many herbaceous species are classified as early seral and ephemeral, and thus respond positively – albeit temporarily – to recent canopy openings from disturbance (Roberts 2004; Hutchinson et al. 2005). These species often grow well in open and drier site conditions or in poor nitrogen-deficient soils, indicating some level of change in abiotic microsite characteristics (Pavlovic et al. 2011). Understory shrubs often respond positively to cutting or dormant season burning by sprouting after topkill (Chapman 1950; Moser et al. 1996). These shade-tolerant shrubs are often associated with moist or mesic sites that have thick litter and duff layers, making regeneration difficult for hardwoods and herbaceous species (Monk et al. 1985). Understory hardwoods and softwoods often respond positively under open conditions in the mid and overstory created by disturbance (Kuddes-Fischer & Arthur 2002). For example, Oaks and yellow pines are more tolerant of fire effects and respond more positively to higher intensity fire treatments than mesic hardwoods and softwoods (Elliott & Vose 2005; Holzmueller et al. 2014). However, many hardwoods and a few softwoods are shade tolerant, making them more vulnerable to competition from light demanding species when light availability is increased (Van Lear et al. 2000; Elliott & Vose 2010). Mesic hardwood or softwood presence would suggest canopy changes, as upland oaks typically regenerate well in more xeric or open sites; while most mesic hardwoods are shade-tolerant or generalist species and regenerate well in a variety of site types (Lorimer 1992; Huddle & Pallardy 1999; Iverson et al. 2008) These characteristics influence competitive species-level interactions and can indicate abiotic and biotic community changes (Van Lear et al. 2000; Kuddes-Fischer & Arthur 2002; Dey & Hartman 2005; Blankenship & Arthur 2006). Therefore, identifying species-specific responses to fire treatments can indicate post-fire successional dynamics of understory community assemblage (Keyser et al. 2008; Azeria et al. 2011).

In the southern Appalachians, fuel reduction is the primary management objective to reduce the risk of destructive wildfires, especially near populated areas (Christensen 1993). Other goals include the establishment of early successional habitat, the control of fire-sensitive mesophytic species, and the restoration of historic open woodland communities (Monk et al. 1985, Nowacki & Abrams 2008). Although the most prevalent technique for achieving management goals in the southern Appalachians has been dormant season burning (January-March), scientists and managers in the region have also expressed interest in using other fuel reduction methods to reverse the effects of fire suppression (Wade & Lunsford 1989; Brose & Van Lear 1998; Schwilk et al. 2009). Numerous studies have reported increases in woody and herbaceous species cover with reductions in overstory and midstory basal area (Keyser et al. 1996; Barnes & Van Lear 1998; Van Lear et al. 2000; Phillips & Waldrop 2008). However, short-term infrequent burning has been found to only minimally reduce overstory and midstory basal area, resulting in insignificant changes to the understory community (Arthur et al. 1998;

Kuddes-Fischer & Arthur 2002; Franklin et al. 2003; Gilbert et al. 2003; Dolan & Parker 2004). Additionally, many dormant season prescribed fire studies in this region report unsuccessful reduction of mesic hardwoods and ericaceous shrubs (Van Lear & Waldrop 1989; Wade & Lunsford 1989; Dey & Hartman 2005; Blankenship & Arthur 2006). This is predominantly due to root stocks remaining viable during the dormant season, which often results in prolific resprouting of competitive hardwood species (Drewa et al. 2002; Burton & Hallgren 2011). Low-intensity burning has also been reported to cause little overstory mortality, which may inhibit the establishment of shade-sensitive species such as grasses, forbs, oaks, yellow pines, and fruit-producing shrubs (Waldrop et al. 2008). Many knowledge gaps still exist regarding these management goals, such as long-term impacts on wildlife species, oak and pine regeneration, the control of competitive fire-sensitive species, and treatment effects on understory herbaceous vegetation (Waldrop & McIver 2006; Stephens et al. 2012; Waldrop et al. 2016). Thus, the exploration of additional fuel reduction methods, like mechanical treatments and burning in different seasons, has increasingly become the focus of most management-related research in this region (Barnes & Van Lear 1998; Van Lear et al. 2000; Alexander et al. 2008).

To explore questions concerning fuel reduction treatments, a program called the Fire and Fire Surrogate Study (FFSS) was started in 2000 by a group of scientists (federal, state, university and private) and land managers across the US (Washington state to Florida) (Youngblood et al. 2005; Schwilk et al. 2009). These studies focus on the effects of various treatment strategies, such as mechanical treatment (e.g. overstory thinning or midstory removal), prescribed burning, and combined mechanical and burning treatments on fuel loading, vegetation dynamics, and fire behavior (Barnes & Van Lear 1998; Brose

& Van Lear 1998; Clinton et al. 1998; Elliot et al. 1999; Waldrop et al. 2008; Schwilk et al. 2009; Waldrop et al. 2016). Originally, there were 13 independent FFSS sites, ranging from Washington to Ohio, with one site located in the Southern Appalachian Mountains of North Carolina (Waldrop & McIver 2006). The southern Appalachian FFSS site is one of the only sites that remains active, with several repeated treatments and data collected nearly every year since 2001 (Waldrop et al. 2016).

The purpose of this study is to investigate community-level responses of treatment effects on understory vegetation using 15 years of data collected at the southern Appalachian FFSS site in Polk County, North Carolina. We use non-parametric ordination to demonstrate community-level heterogeneity and community structure in response to 4 fuel reduction treatments: burn-only (B), mechanical (M), combination of mechanical and burning (MB), and control (C). To examine similarities in species assemblage within the study site, we also characterize plot-level clustering patterns in response to the repeated treatments. Finally, we identify species that are representative of community-level treatment responses using indicator species analyses. These results represent the species that are the most influenced by treatment, therefore, we posit that these species can be used as a proxy for community changes in response to treatment type.

Methods

Location

The study area is located in Polk County, North Carolina on the Green River Game Land, which is managed for wildlife, public recreation, timber and other resources by the North Carolina Wildlife Resources Commission (NCWRC) (Appendix A-1). The Game

Land covers 5,841 hectares and is classified as a mountainous region, where elevations range from about 300 m to 800 m. When the study was initiated in 2001, forests in the study area were about 80 to 120 years old and consisted of mixed- xeric or mesic oak (*Quercus* spp.) and pine (*Pinus* spp.) species depending on the topographic position. Shortleaf pine (*Pinus echinata*), pitch pine (*P. rigida*) and Virginia pine (*P. virginiana*) can be found on dry ridge tops while eastern white pine (*P. strobus*) can be found in moist coves. Ericaceous shrubs, like mountain laurel (*Kalmia latifolia*) and great rhododendron (*Rhododendron maximum*), made up a dense mid-story layer throughout the study area. Most of the soils are of the Evard series (fine-loamy, oxidic, mesic, Typic Hapludults) in areas that can be described as moderately deep, well-drained, mountain uplands.

Study Design

This study utilizes a randomized complete block design, which includes 4 treatments units in each of 3 replicate blocks for a total of 12 treatment sites. Each treatment site covers an average of 12 hectares (ha), which includes a 4-ha buffer zone. Both the buffer zone and the site receive the same experimental treatment. Within the replicate blocks, each of the 4 treatment units were randomly assigned to one of the treatments: control (C), prescribed burning only (B), mechanical fuel reduction (M), and prescribed burning plus mechanical fuel reduction (MB) (Appendix A-2). The B treatment was repeated 4 times by 2016, having been applied in February or March of 2003, 2006, 2012, and 2015. The M treatment was applied twice in the winters of 2001-2002 and 2011-2012 and included cutting of all woody vegetation >1.4 m tall and <10.2 cm in diameter at breast height (dbh) with a chainsaw. The MB treatment was initiated with the first mechanical cutting in 2001-2002, treated with the 2 repeated prescribed burns in 2003 and 2006, included the second mechanical cutting in 2011-2012, and was followed by the final 2 prescribed burns (2012 and 2015). All treatment areas were sampled in the pre-treatment year (2001), and in the growing seasons following each treatment (2003, 2005, 2006, 2008, 2011, 2012, 2013, 2014, 2015, and 2016).

A 50 x 50 m grid was established in the treatment areas, with grid points permanently marked and georeferenced. Ten 0.1-ha sample plots were established at randomly selected grid points within each treatment area. The sample plots are 50 m x 20 m and divided into ten subplots, each about 10 m^2. Within each subplot, two 1m^2 quadrats were established in the northwest and southeast corners to measure understory vegetation (<1.4 m tall) using Modified Whittaker plots (FFS Study Plan 2001). Composition and abundance data were collected in each 1m^2 quadrat; this included recording the cover values of all species and additionally recording the stem counts for tree species. Cover values used in this sampling method were recorded as classification values: 1 (<1%), 2 (1% - 10%), 3 (>10% - 25%), 4 (>25% - 50%), 5 (>50% - 75%), and 6 + (>75%). To generate workable cover class values for analysis, we used the median of the percent ranges for each cover class; for example, 5.5 % would be used for the cover class 2, etc.

Analysis

All species in the understory (<1.4 m tall) were classified into 3 general life form categories for analysis: trees, shrubs, and herbaceous vegetation. Cover % for trees, shrubs and herbaceous vegetation were derived from cover classification values recorded in the post treatment year, 2016. The median percentage represented in each cover class was then averaged across each 0.1-ha plot for all treatments (n = 120). Because stem density data were not collected for shrubs and herbaceous species, cover % was used for all three vegetation groups in the non-metric multidimensional scaling analyses. Cover % was also used for the shrubs and herbaceous species groups for the cluster and indicator species analyses. Stem densities (stems/ha) for tree species were derived from the average of the total stem counts in each 0.1-ha plot and used in the tree species cluster and indicator species analyses.

Non-metric multidimensional scaling

To assess understory plant community response to different long-term repeated fuel reduction treatments, we first visualized the data in a nonmetric multidimensional scaling (NMDS) ordination (McCune et. al. 2000, Kruskal 1964). Cover % data from the 2016 survey period was used to construct a Bray-Curtis dissimilarity matrix for all understory species (Faith et al. 1987, Oksanen et al. 2015). Each functional group was analyzed using the *metaMDS* function from the *vegan* package in R software and overlaid in a single figure (version 3.2.2, R Core Team 2015). Stability for each functional group was assessed using the *scree.plot* function with 20 randomized runs and 3 final axes iterations. Though 3 axes reduced the most stress (0.16), a 2-dimensional solution (0.22) was sufficient for explaining similarities among species across ecological gradients and was used in the final ordination figure. The final ordination contains three components: maximum convex polygons associated with each treatment, which represent variation in plant community responses and community heterogeneity among plots in each treatment; species points, which are positioned in a manner to reflect sites they are most closely associated with; and axes, which can represent some type of environmental gradient that can be used to describe separation among species points and treatment polygons.

Agglomerative hierarchical cluster analysis

To further examine functional group responses to treatments, we used aggolomerative hierarchical cluster analyses (AHCA) which forms clusters based on shared species within plots (McCune & Grace 2002). AHCAs were performed with the *agnes* function in the *cluster* package (Maechler et al. 2015). Each of the 3 functional groups were analyzed separately at the plot level (n_{trees} = 120, n_{shrubs} = 120, $n_{\text{herbaceous veg.}}$ = 117) to compare between-plot similarity of plant species composition across all functional groups (Oksanen et al. 2015). Three plots were omitted within the herbaceous vegetation dataset due to the absence of understory vegetation in those plots. Stem densities (stems/ha) were used in the tree species AHCA, and cover % were used during the shrubs and herbaceous vegetation AHCA. Cover % was normalized, making the marginal sum of squares equal to zero for a better fitting distribution. With an AHCA technique, each plot is considered an individual cluster and therefore plots with similar species are grouped into larger clusters resulting in a single dendrogram (McCune & Grace 2002). We used Bray-Curtis as the distance metric and flexible beta linkage (β = - 0.25) as the fusion strategy to determine the appropriate number of clusters for each functional group based on local group structure (Oksanen et al. 2015). The cluster number will be determined from fusion height, a visualization method that shows natural breaks in the data, indicating the highest number of plot similarities. To better explain functional group responses to the 4 treatments, clusters were then grouped into descriptive categories (e.g. positive treatment response, negative treatment response, or indifferent treatment response) based on the proportions of treatment plots in each cluster (McCune et al. 2000; Gonzalez-Tagle et al. 2008).

Indicator species analysis

To detect species-level drivers of community assemblage within each treatment, indicator species analyses (ISA) were conducted in the *indicspecies* package in R for each functional group (De Cáceres & Legendre 2009; Dufrene & Legendre 1997). Indicator species are those that have high specificity and fidelity to a given site, and, therefore, are the most indicative members of that treatment (Costanza et al. 2017). For shrubs and herbaceous vegetation, cover % data were used to calculate the highest indicator value (IV_{max}), *P*-value, specificity (*A*) and fidelity (*B*) for each species using the *multipatt* function (*duleg* = TRUE) and *strassoc* function in R (De Cáceres & Legendre 2009). A species' indicator value (IV) (0 – 1) is the square root of the product of that species' *A* and *B*; *A* indicates the probability that the given species is in a given cluster when it is found, and *B* indicates the probability of finding the given species in a given cluster (Shearman et al. 2017). The *P*-values ($\alpha = 0.05$) represent the probability of obtaining an indicator value by chance that is equal to 1 (Kane et al. 2010). Data were computed based on a Monte Carlo test with 999 permutations (McCune & Grace 2002).

Results and Discussion

Community-level ordination (NMDS)

The NMDS ordination was resolved by two axes, with a stress value of 0.20 which is considered acceptable based on Clarke (1993) (Figure 2.1). The overlapping treatment polygons suggest similarity among vegetative community responses, which also suggests that treatments share many species. Much of the tree, shrub, and herbaceous vegetation showed little variability in ordination space, suggesting somewhat differential but mostly shared responses to treatments. Additionally, the relatively low distances between species made the ecological trends represented by each axes difficult to interpret. Overall, the resulting NMDS showed little separation between treatment polygons and little variability in species spread when all species were included in the ordination. This suggests low detectability of community responses to treatments and that separation of functional groups is necessary to further examine changes in community assemblage.

Functional group clustering (AHCA)

Considering the NMDS results, we used an AHCA to further break down functional groups into discrete clusters at the plot level (P). Clustering similar species composition among all treatment plots helps to describe small-scale community assemblage patterns that may have otherwise been overlooked within the NMDS ordination (Shearman et al. 2017). These cluster-level responses were generalized into 2 broad categories based on species assemblages: 1) those that respond similarly to the C treatment, and 2) those that respond differently from the C treatment.

Among the clusters that respond differently to C, one or more distinct responses can be described by treatment plot proportions (e.g. if a cluster only contains B or MB plots). Clusters 2 and 5 in the tree dendrogram responded similarly to C (51 % TP_{Total}), while clusters 1, 3, and 4 responded differently to B, M and MB treatments (49.2 % TP_{Total}) (Figure 2.2). Plots in cluster 1 appeared to respond similarly to the effects of B and MB (T_{C1} = 11_{B}, 6_{C}, 2_{M}, 15_{MB}), plots in cluster 3 responded similarly to the effects of

B (T_{C3} = 6_B, 1_C, 1_M, 0_{MB}), and plots in cluster 4 responded distinctly to M effects (T_{C4} = 0_B, 7_C, 10_M, 0_{MB}).

In the shrub dendrogram, cluster 4 responded similarly to C (16 % SP_{Total}), while clusters 1, 2 and 3 responded differently to B, M and MB treatments (60 % SP_{Total}) (Figure 2.3). Plots in clusters 1 and 2 responded similarly to the effects of B and MB (S_{C1} = 6_B, 3_C, 1_M, 7_{MB}; S_{C2} = 10_B, 6_C, 4_M, 9_{MB}), and plots in cluster 3 responded distinctly to the effects of M (S_{C3} = 8_B, 15_C, 23_M, 9_{MB}).

In the herbaceous species dendrogram, clusters 4 and 5 responded similarly to C (36 % HVP_{Total}), while clusters 1, 2, and 3 responded distinctly to MB, B and M treatments (64 % HVP_{Total}) (Figure 2.4). Cluster 1 responded distinctly to B and M effects (HV_{C1}= 9_B, 14_C, 12_M, 0_{MB}), cluster 2 responded distinctly to MB effects (HV_{C2} = 8_B, 0_C, 0_M, 20_{MB}), and cluster 3 responded similarly to B and MB effects (HV_{C3} = 1_B, 6_C, 4_M, 1_{MB}). Overall, many clusters show similarities in community response among treatments, however, the few clusters that show community divergence from the C treatments suggest distinct treatment effects on understory vegetation.

While the NMDS showed only modest differences in understory community assembly between treatments, the HCA resulted in more evidence of specific treatment effects within each functional group. The largest portion of clusters was associated with category 1, which suggests that 15 years of treatments result in more similar responses across functional groups than previously assumed (Van Lear et al. 2000; Dey & Hartman 2005; Stephens et al. 2012). This also shows that many clusters were generally responsive to all treatments, suggesting the presence of generalist species assemblages that are largely unaffected by treatment type. Additionally, many clusters responded similarly to fire-related disturbance (B or MB), suggesting community assemblage shifts to a more ruderal or fire-tolerant community. This may predominantly be due to the overstory and midstory canopy openings which were mostly created by MB treatments, as reported in Waldrop et al. (2016). However, identifying individual species that are driving these responses would give more indication of community assemblage and abiotic changes in response to repeated treatments (Gilliam & Roberts 2003; Keyser et al. 2008; Azeria et al. 2011).

Community indicators for treatment response (ISA)

The ISA for trees identified 11 indicator species total: 3 species in B, 1 species in M, 7 species in MB, and no species in C (Table 2.1). The indicator species in B (*A. rubrum, Amelanchier arborea,* and *L. tulipifera*) indicate that B sites are largely differentiated by mesic species that grow well under conditions created by low intensity prescribed fire. This suggests that 4 repeated dormant season burns are not sufficient for meeting the management objectives of creating understory conditions that favor more desirable species, like oaks and yellow pines (Kuddes-Fischer & Arthur 2002; Dolan & Parker 2004). The indicator species in M (*P. strobus*) suggests that these sites are differentiated by mesic white pine species that grow well under more moist or mesic conditions with low light availability (Phillips et al. 2007). This also suggests that long-term M treatments may not create conditions that are favorable for fire-tolerant species as they need a more open canopy and drier microsite conditions to grow optimally (Vose et al. 1993). The indicator species in MB (*Sassafras albidum, Diospyros virginiana, N. sylvatica, Oxydendrum arboreum, Quercus coccinea, Q. montana,* and *Robinia pseudoacacia*) indicates that these sites are differentiated by more xeric species that are light-responsive and grow well under poor site conditions (Clinton & Vose 2000). More specifically, *Q. montana, Q. coccinea,* and *S. albidum* grow best in open dry conditions, and *R. pseudoacacia* grow best in poor nitrogen deficient soils, suggesting some level of mesophication reversal in the MB treatment (Boring & Swank 1984; Dey & Hartman 2005).

The ISA for shrubs identified 6 indicator species total: 2 species in M, 4 species in MB, and no species in B or C (Table 2.1). The indicator species in M (*K. latifolia* and *R. maximum*) indicate that these sites are differentiated by ericaceous shrubs that grow well under light-limited mesic conditions, and resprout prolifically when cut (Vose et al. 1993; Dey & Hartman 2005). This suggests that M treatments may not be reducing ericaceous shrub competitors, which is a priority management objective in this region (Waldrop et al. 2016). The indicator species in MB (*Ceanothus americanus, Hypericum hypericoides, Lyonia ligustrina,* and *Rhus glabra*) suggests that these sites are differentiated by more light-responsive and opportunistic species that grow well in open, xeric sites (Hutchinson et al. 2005; Keyser et al. 2008).

The ISA for herbaceous vegetation identified 17 indicator species total: 1 species in C, 16 species in MB, and no species in B or M (Table 2.1). The indicator species in C (*Arundinaria* sp.) usually comprise the midstory in more open sites or in low elevation sites near streams (Taylor 2006). This may be due to increases of hardwoods, shrubs, and herbaceous species outcompeting this species in the other treatments. The indicator species in MB suggests a high level of understory diversity, with 4 grasses (*Dichanthelium* spp., *Piptochaetium avenaceum, Schizachyrium scoparium,* and *Scleria* spp.), 1 sedge (*Carex* spp.), and 12 forb species (*Cassia* spp., *Conyza canadensis*, *Coreopsis major, Desmodium nudiflorium, Erechtites hieraciifolius, Helianthus divaricatus, Houstonia purpurea, Lespedeza bicolor, Potentilla canadensis, Rubus argutus,* and *Solidago* spp.), 3 of which are nitrogen-fixing (*Cassia* spp., *D. nudiflorum,* and *L. bicolor*). This suggests that MB treatments are facilitating the establishment of a different set of species that are largely unique to this treatment (Burton et al. 2011). The indicator species in MB also possess certain characteristics that suggest site xerification. For example, nitrogen-fixing forbs indicate poorer nitrogen-deficient soil conditions that often occur following a large disturbance (Peterson et al. 2007). The forbs, mainly *Asteraceae*, often respond well to larger disturbances indicating larger openings in the canopy and midstory (Hutchinson et al. 2005). Many graminoids often grow well in drier poorer sites, indicating increased light availability and the reversal of mesophication in the understory (Peterson et al. 2007).

Conclusions

At the community level, 15 years of repeated treatments have not resulted in much change in understory community assemblage as observed from our NMDS results. We observed an overall positive response from understory functional groups to more intense fire treatments (MB). We also observed an increase in early seral, fire-tolerant/dependent herbaceous species in the understory with MB treatments. This indicates that MB treatments are creating the correct physical environment for these species to respond, specifically, opening the midstory and overstory strata and increasing light availability to the forest floor (Van Lear et al. 2000; Hutchinson et al. 2005; Waldrop et al. 2016). MB was the only treatment that produced site conditions favorable to our management objectives, in which upland oaks, xeric shrubs and numerous perennial and ruderal herbaceous species were indicators of mesophication reversal. Additionally, we found that 15 years of repeated B and M treatments are not causing substantial understory community changes, as competitive mesic species like *A. rubrum, A. arborea, L. tulipifera, K. latifolia,* and *R. maximum* remain prevalent. However, other longer-term studies suggest that an open woodland community can be obtained through additional repeated burning treatments that result in post-fire overstory mortality (Stratton 2007). Our results describe modest community changes in response to B, thus suggesting early signs of long-term community shifts from dormant season burning.

Overall, reversing decades of fire suppression takes extensive amounts of time. Even after more than 15 years of repeated treatments, we are seeing modest changes, most of which are observed in the most intensive treatments (B and MB). In general, this study confirms the idea that change needs to occur in the abiotic environment before changes will be seen in the biotic community. We are seeing this in the MB treatments, further indicated by our desirable indicator species responses. However, further research should be conducted on the longer-term effects of B to determine if B treatments will eventually equate to the more prevalent effects of MB treatments. Additionally, the minute similarities in understory responses observed between M and B treatments suggests that M is only somewhat of a surrogate for B treatments. This study provides valuable evidence for management, but longer-term studies are still needed to determine disturbance effects on vegetative community responses. Observing vegetation changes is a slow process, as expected, and should be continued for effective management practices in the southern Appalachians.

TABLES

Table 2.1 Site specificity ($A_{specificity}$), fidelity ($B_{fidelity}$), relative indicator value (IV_{rel} ($A * B$), abundance (stems/ha or cover %), and the maximum indicator value (IV_{max}) results from the indicator species analyses (ISA) on understory tree, shrub, and herbaceous species. These results indicate species assemblages in response to 4 repeated treatments (Control = C, Burned = B, Mechanical removal of stems <10 cm dbh = M, and combination of M + B = MB).

Group			$A_{specificity}$	$B_{fidelity}$	IV_{rel} ($A*B$)	Abundance stems/ha	IV_{max}
Trees (3 Treatments)	B						
		Acer rubrum	0.40	1.00	0.40	2376.40	0.63 ***
		Liriodendron tulipifera	0.39	0.77	0.30	526.43	0.55 **
		Amelanchier arboreum	0.71	0.33	0.23	413.89	0.49 **
	M						
		Pinus strobus	0.64	0.20	0.13	116.67	0.36 *
	MB						
		Quercus coccinea	0.34	0.97	0.33	1360.43	0.58 ***
		Sassafras albidum	0.38	1.00	0.38	1173.89	0.62 ***
		Nyssa sylvatica	0.40	0.90	0.36	966.98	0.60 ***
		Oxydendrum arboreum	0.49	0.60	0.29	838.89	0.54 **
		Diospyros virginiana	0.66	0.30	0.20	261.10	0.44 **
		Quercus montana	0.37	0.87	0.32	1709.09	0.56 *
		Robinia pseudoacacia	0.53	0.30	0.16	245.00	0.40 *
						Abundance cover %	
Shrubs (2 Treatments)	M						
		Rhododendron maximum	0.84	0.37	0.31	287.27	0.55 ***
		Kalmia latifolia	0.36	0.87	0.31	780.66	0.56 **
	MB						
		Rhus glabra	0.81	0.6	0.49	135.44	0.70 ***
		Ceanothus americanus	0.88	0.27	0.24	39.21	0.49 ***
		Lyonia ligustrina	0.84	0.17	0.14	179.15	0.38 *
		Hypericum hypericoides	0.56	0.27	0.15	20.88	0.39 *
Herbaceous Vegetation	C						
		Arundinaria gigantea	0.83	0.23	0.19	111.50	0.44 **
	MB						
		Rubus argutus	0.67	0.70	0.47	122.93	0.69 ***
		Coreopsis major	0.73	0.77	0.56	91.41	0.75 ***
		Carex spp.	0.48	0.73	0.35	83.50	0.60 ***
		Dichanthelium spp.	0.46	0.97	0.45	71.57	0.67 ***
		Schizachyrium scoparium	0.68	0.50	0.34	69.86	0.58 ***

	Helianthus divaricatus	0.98	0.27	0.26	53.50	0.51 ***
(2 Treatments)	*Potentilla canadensis*	0.56	0.53	0.30	47.18	0.55 ***
	Lespedeza bicolor	1.00	0.20	0.20	37.17	0.45 ***
	Solidago spp.	0.60	0.33	0.20	70.08	0.45 **
	Desmodium nudiflorum	0.64	0.43	0.28	52.79	0.51 **
	*Scleria*spp.	0.55	0.53	0.29	23.50	0.54 **
	Conyza canadensis	0.70	0.30	0.21	5.46	0.46 **
	Erechtites hieraciifolius	0.73	0.27	0.20	3.00	0.44 **
	Piptochaetium avenaceum	0.63	0.23	0.14	43.00	0.39 *
	Houstonia purpurea	0.62	0.43	0.27	30.50	0.46 *
	Cassia spp.	0.90	0.13	0.12	4.50	0.35 *

2.1

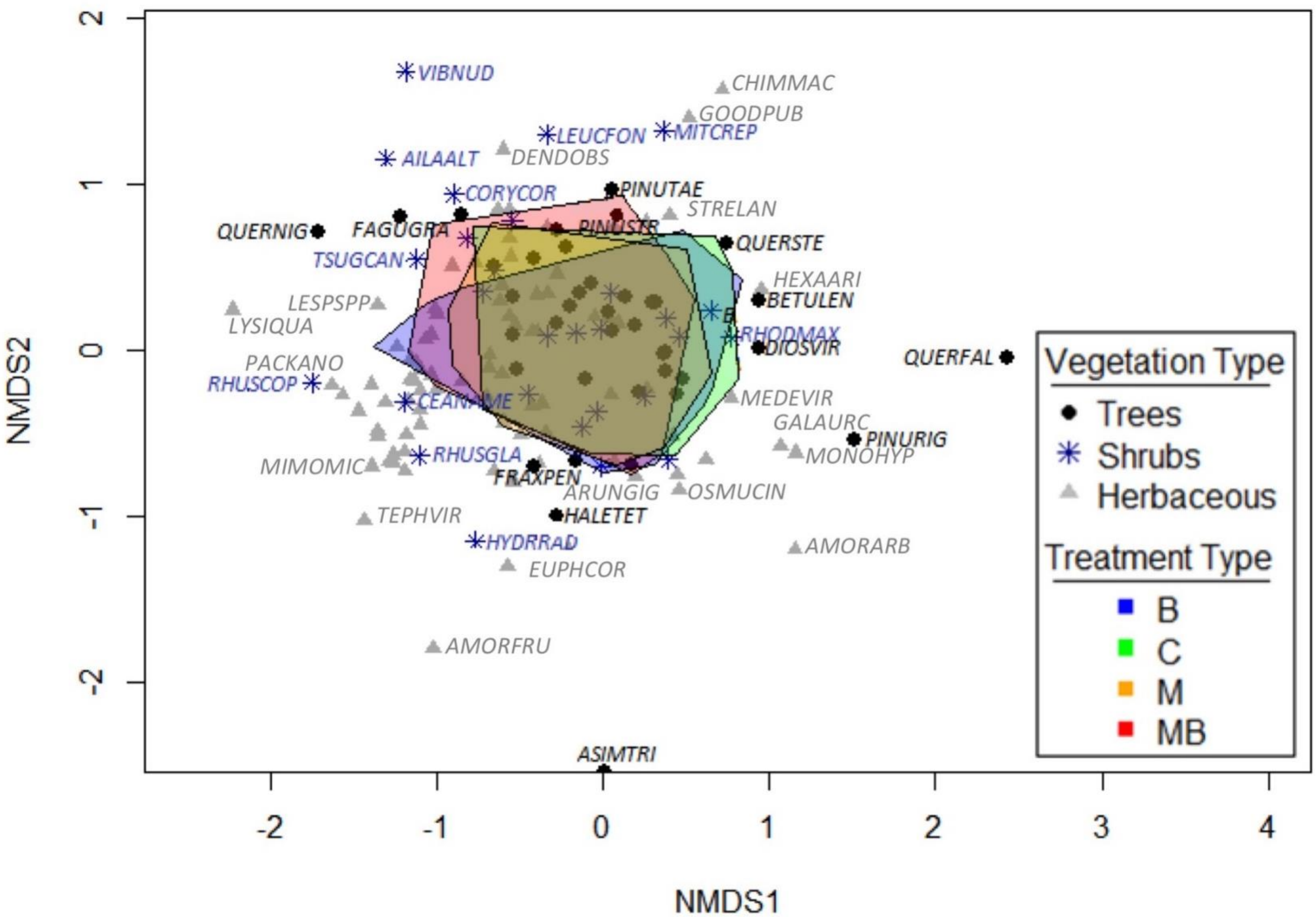


Figure 2.1 Results from the non-metric multidimensional scaling (NMDS) analysis show overlap among understory vegetation community responses to treatments, indicated by blue (B), green (C), orange (M), and red (MB) polygons. Trees (filled circles), shrubs (blue stars), and herbaceous vegetation (grey triangles) also show little variability in ordination space, suggesting similarities in responses to B, C, M, and MB treatments

2.2

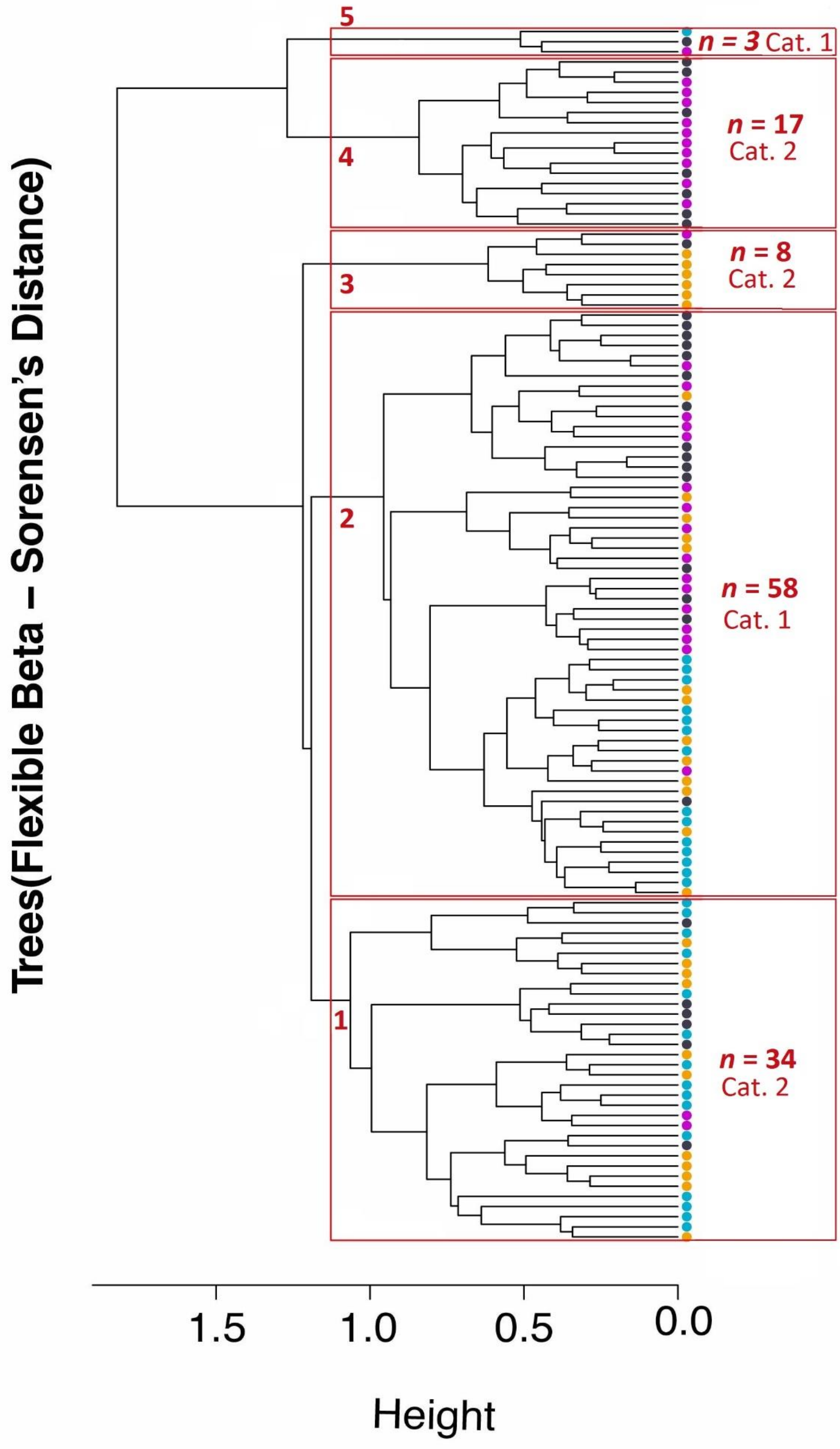
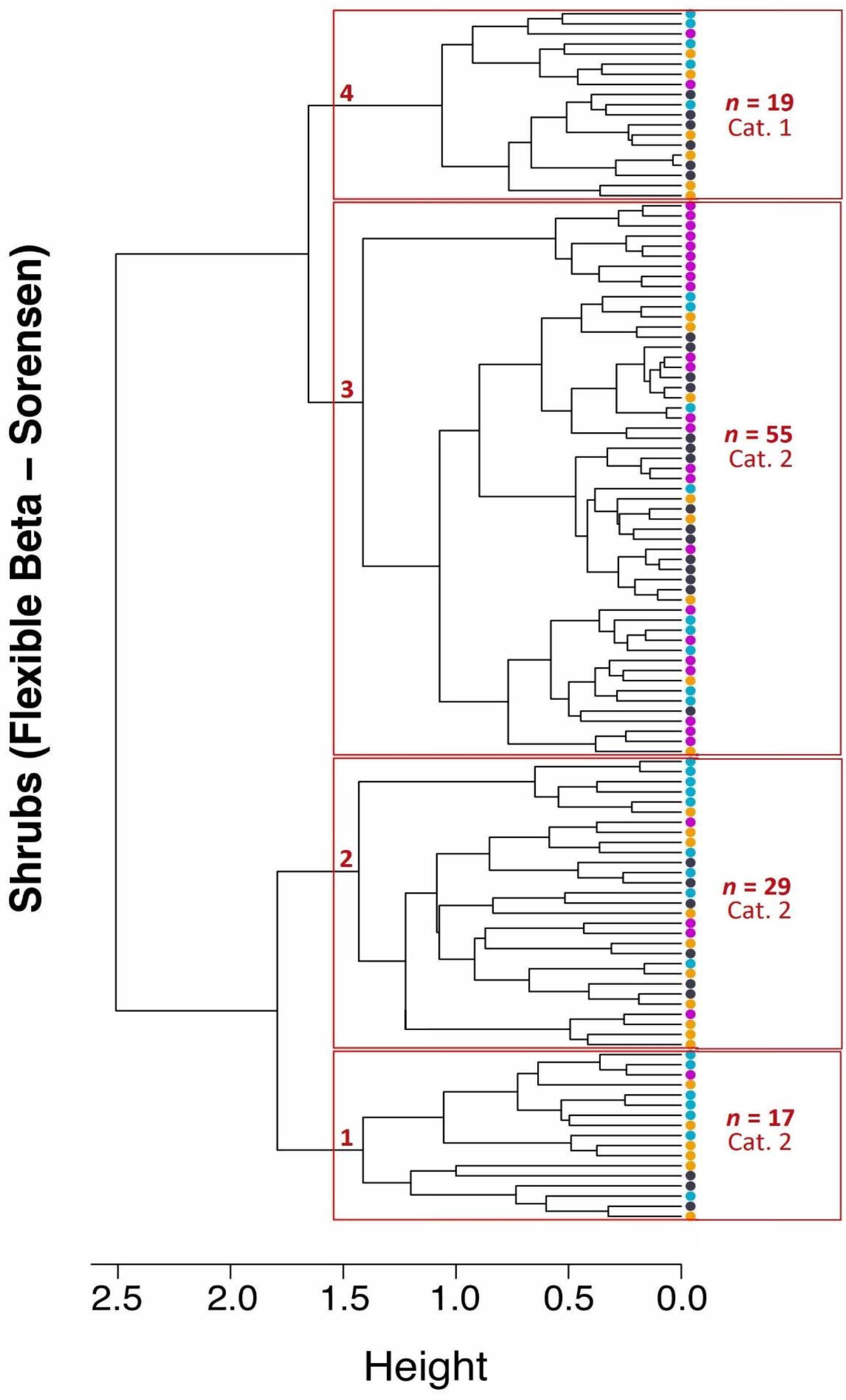

5
n = 3 Cat. 1
4
n = 17
Cat. 2
3
n = 8
Cat. 2
2
n = 58
Cat. 1
1
n = 34
Cat. 2
Trees(Flexible Beta – Sorensen's Distance)
1.5
1.0
0.5
0.0
Height

Figure 2.2 Results from the agglomerative hierarchical cluster analysis (HCA) for trees shows 5 distinct clusters in response to treatments. Each line on the dendrogram is denoted by orange (B), grey (C), purple (M), and blue (MB) dots that indicate which treatment was applied to that plot. Cluster 1had 11 plots in B, 6 in c, 2 in M, and 15 in MB, categorizing it as having a different response from C (Cat. 2). Cluster 2 had 13 plots in B, 15 in C, 16 in M, and 14 in MB, categorizing it as having a similar response to C (Cat. 1). Cluster 3 had 6 plots in B, 1 in C, 1 in M, and 0 in MB, falling under category 2. Cluster 4 had 0 plots in B, 7 in C, 10 in M, and 0 in MB, falling under category 2. Cluster 5 had 0 plots in B, 1 in C, 1 in M, and 1 in MB, falling under category 1.

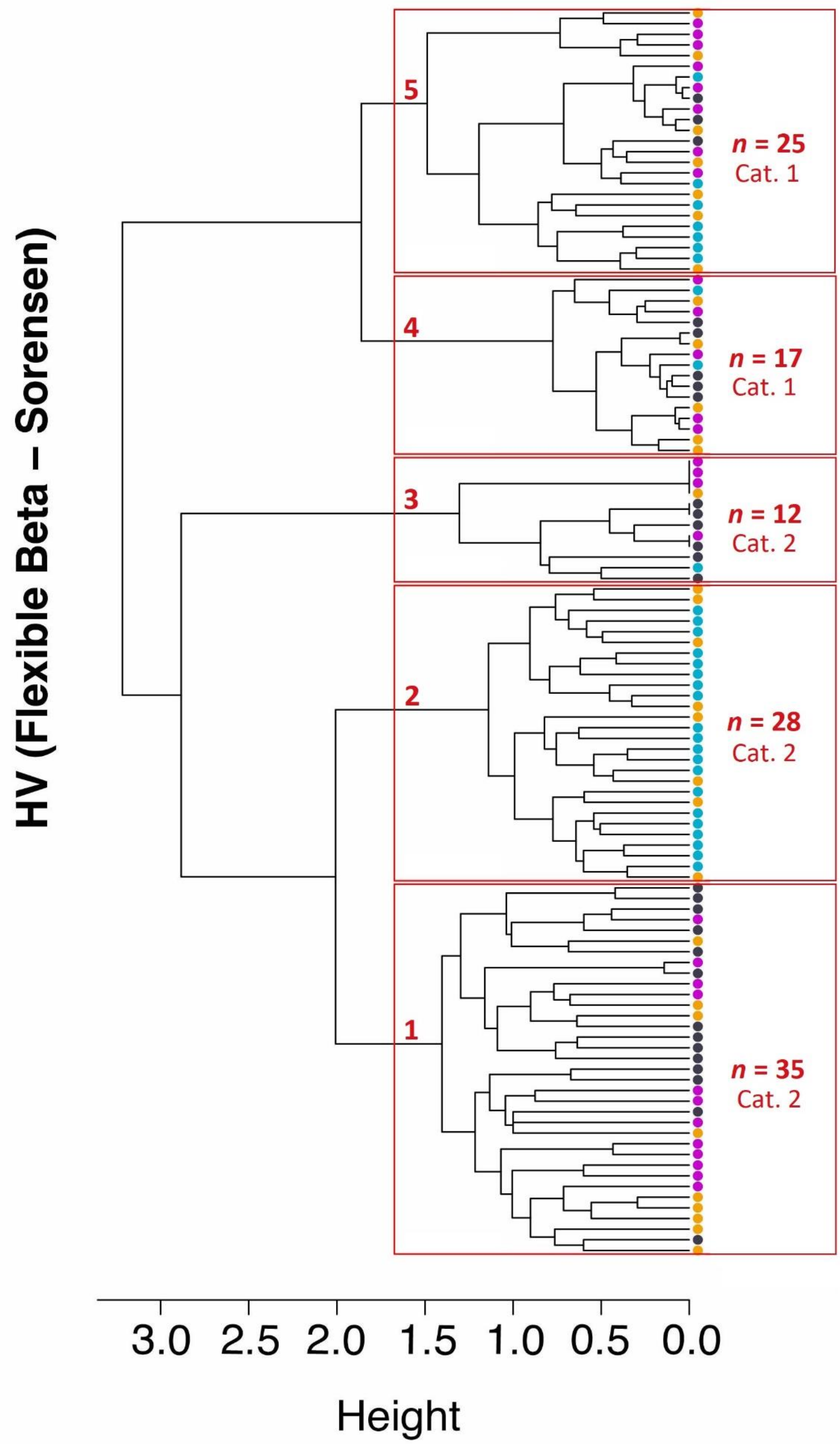
2.3



Shrubs (Flexible Beta – Sorensen)
4
n = 19
Cat. 1
3
n = 55
Cat. 2
2
n = 29
Cat. 2
1
n = 17
Cat. 2
2.5
2.0
1.5
1.0
0.5
0.0
Height

Figure 2.3 Results from the agglomerative hierarchical cluster analysis (HCA) for shrubs shows 4 distinct clusters in response to treatments. Each line on the dendrogram is denoted by orange (B), grey (C), purple (M), and blue (MB) dots that indicate which treatment was applied to that plot. Cluster 1 had 6 plots in B, 3 in c, 1 in M, and 7 in MB, categorizing it as having a different response from C (Cat. 2). Cluster 2 had 10 plots in B, 6 in C, 4 in M, and 9 in MB, falling under category 2. Cluster 3 had 8 plots in B, 15 in C, 23 in M, and 9 in MB, falling under category 2. Cluster 4 had 6 plots in B, 6 in C, 2 in M, and 5 in MB, categorizing it as having a similar response to C (Cat. 1).

2.4



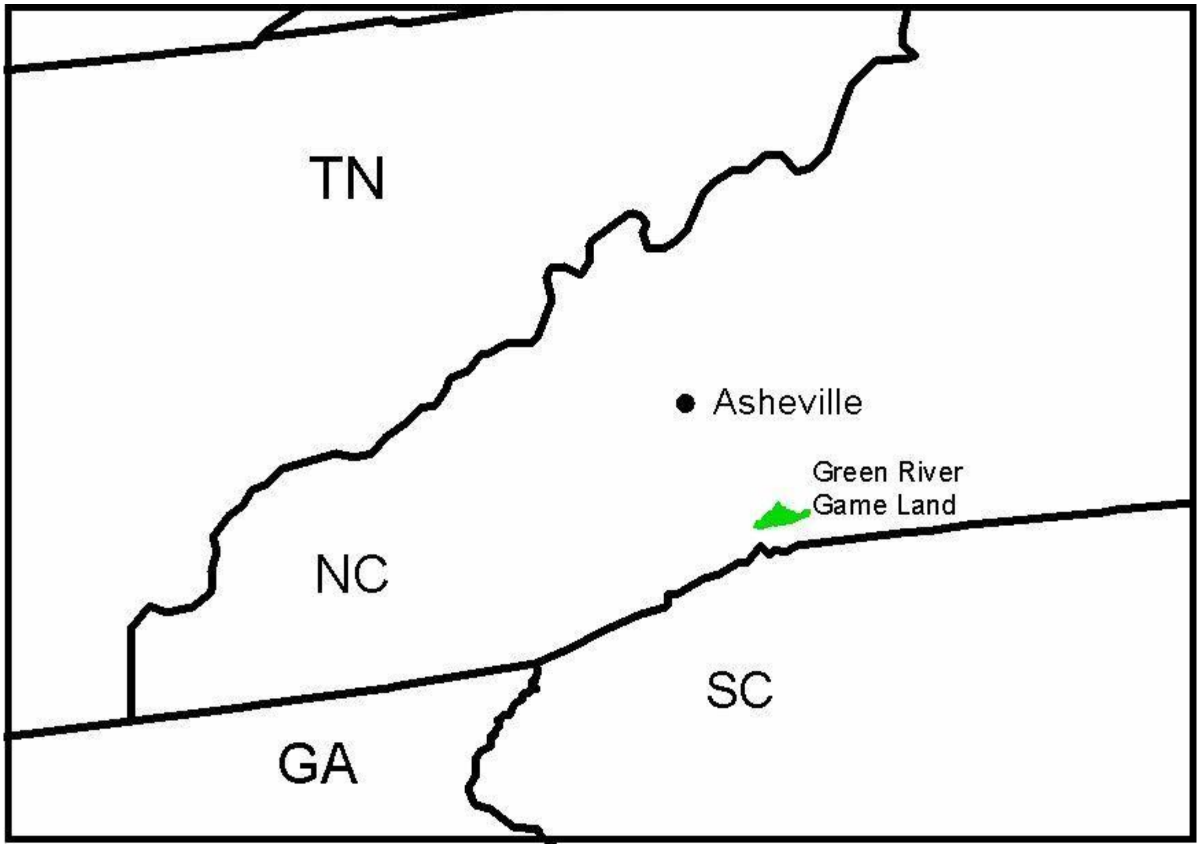
HV (Flexible Beta – Sorensen)
5
n = 25
Cat. 1
4
n = 17
Cat. 1
3
n = 12
Cat. 2
2
n = 28
Cat. 2
1
n = 35
Cat. 2
3.0 2.5 2.0 1.5 1.0 0.5 0.0
Height

Figure 2.4 Results from the agglomerative hierarchical cluster analysis (HCA) for herbaceous species shows 5 distinct clusters in response to treatments. Each line on the dendrogram is denoted by orange (B), grey (C), purple (M), and blue (MB) dots that indicate which treatment was applied to that plot. Cluster 1 had 9 plots in B, 14 in c, 12 in M, and 0 in MB, categorizing it as having a different response from C (Cat. 2). Cluster 2 had 8 plots in B, 0 in C, 0 in M, and 20 in MB, falling under category 2. Cluster 3 had 1 plot in B, 6 in C, 4 in M, and 1 in MB, falling under category 2. Cluster 4 had 5 plots in B, 5 in C, 5 in M, and 2 in MB, categorizing it as having a similar response to C (Cat. 1). Cluster 5 had 7 plots in B, 3 in C, 8 in M, and 7 in MB, falling under category 1.

APPENDIX

Appendix A

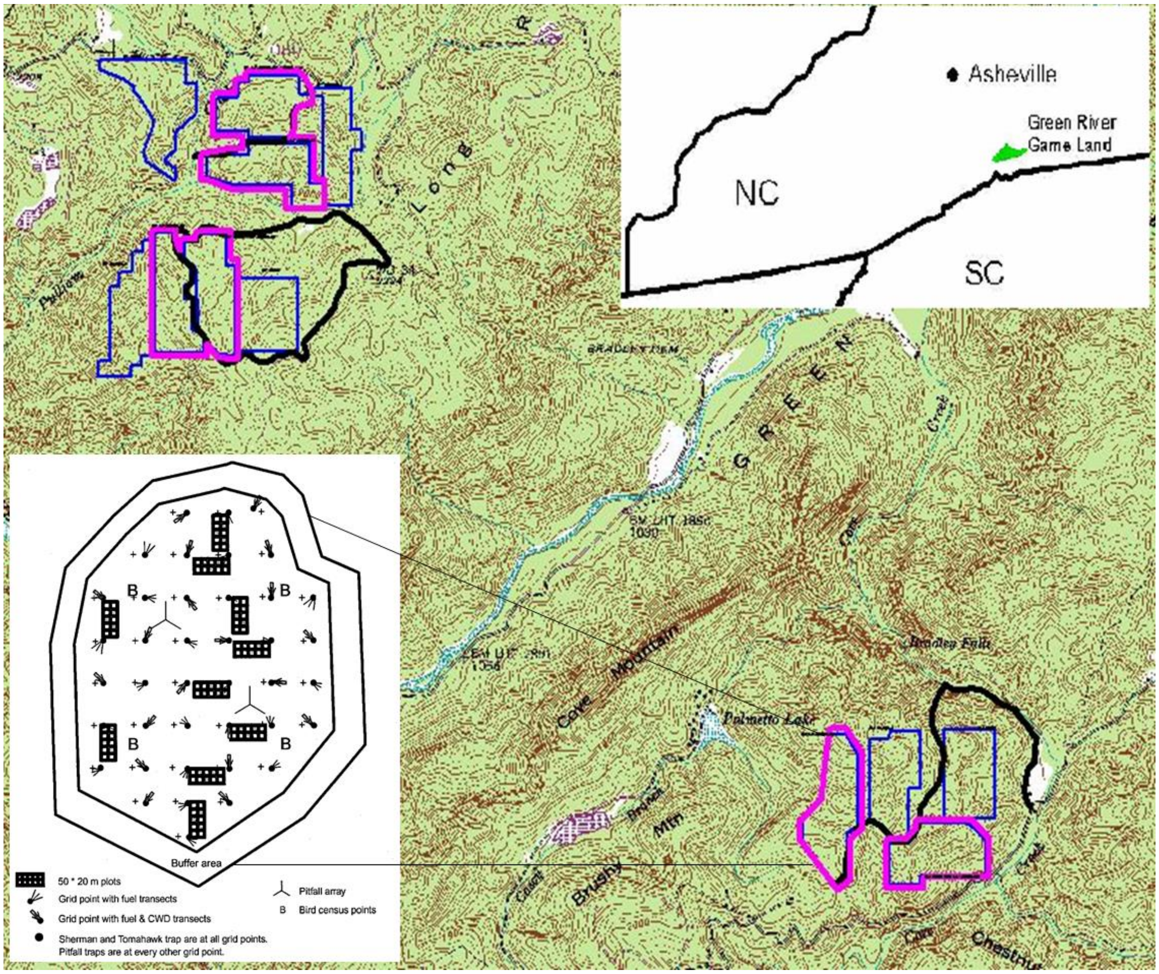
Study Location



A-1: The study location in the Green River Game Lands, NC, USA.

<u>Appendix A</u>

<u>Treatment and Study Design Layout</u>



A-2: Treatment layout at Green River Game Land. Randomized complete block design: with 3 treatment areas, 4 treatments in each area (Control = C, Burned = B, Mechanical removal of stems <10 cm dbh = M, and Mechanical removal of stems <10 cm dbh + burned = MB), 10 plot origins in each area, 10 subplots in each origin, and 2 m2 vegetation plots per subplot. Pink outline shows the MB treatment areas, blue outline shows the M treatment areas, and black shows the B areas.

REFERENCES

Chapter 1

Abrams, M. D., D. A. Orwig and T. E. DeMeo. 1995. Dendrochronology and successionaldynamics of a presettlement-origin white pine-mixed oak forest in the southern Appalachians USA. Journal of Ecology 83:123-133.

Ayers, H. B., and W. W. Ashe. 1905. The southern Appalachian forests. US Geological Survey professional paper, 37. Reston: US Geological survey.

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

Barden, L. S., and F. W. Woods. 1976. Effects of fire on pine and pine-hardwood forests in the southern Appalachians. Forest Science 22(4):399-403.

Beck, D. E., and R. M. Hooper. 1986. Development of a southern Appalachian hardwood stand after clearcutting. Southern Journal of Applied Forestry 10:168-172.

Boerner, R. E. J., J. A. Brinkman, and D. A. Yaussy. 2007. Ecosystem restoration treatments affect soil physical and chemical properties in Appalachian mixed-oak forests. Proceedings, 15th Central Hardwood Forest Conference. e-General Technical Report SRS-GTR-101. USDA Forest Service, Southern Research Station. p. 107-115.

Brockway, D. G., K. W. Outcalt, and R. N. Wilkins. 1998. Restoring longleaf pine wiregrass ecosystems: plant cover, diversity and biomass following low-rate hexazinone application on Florida sandhills. Forest Ecology and Management. 103:159–175.

Brose, P. H., D. H. Van Lear. 1998. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research 28(3):331-339.

Brose, P. H., T. M. Schuler, D. H. Van Lear, and J. Berst. 2001. Bringing fire back: the changing regimes of the Appalachian mixed oak forest. Journal of Forestry 99: 30-35.

Brose, P. H. 2010. Long-term effects of single prescribed fires on hardwood regeneration in oak shelterwood stands. Forest Ecology and Management 260(9):1516-1524. Cronon, W. 1983. Changes in the land: Indians, colonists, and the ecology of New England. New York: Hill and Wang.

Clinton, B. D., J. M. Vose, and W. T. Swank. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands: vegetation composition and diversity of 13-year-old stands. Canadian Journal of Forest Restoration 25: 2211-2217.

Coates, T. A., V. B. Shelburne, T. A. Waldrop, B. R. Smith, H. S. Jr. Hill, and D. M. Simon. 2010. Forest soil response to fuel reduction treatments in the Southern Appalachian Mountains. In: Stanturf, John A., ed. 2010. Proceedings of the 14th biennial southern silvicultural research conference. Gen. Tech. Rep. SRS–121. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 283-287.

Cronon, W. 1983. Changes in the land: Indians, colonists, and the ecology of New England. Hill and Hang, New York.

Delcourt, P. A., and H. R. Delcourt. 1998. The influence of prehistoric human-set fires in oak-chestnut forests in the southern Appalachians. Castanea 63(3): 337-345.

Denevan, W. M. 1992. The pristine myth: the landscape of the Americas in 1492. Annals of the Association of American Geographers 82: 369-385.

Dolan, E. 2002. Soil and site variability in the northeast Wenatchee Mountains. Seattle, WA: University of Washington. 83 p. M.S. thesis.

Dolan, B. J., and G. R. Parker. 2004. Upland oak ecology symposium: history, current conditions, and sustainability. Gen. Tech. Rep. SRS–73. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 311 p. 285-291.

Ducey, M. J., W. K. Moser, and P. M. Ashton. 1996. Effect of fire intensity on understory competition and diversity in a Kalmia-dominated oak forest, New England, USA. Vegetatio 123: 81-90.

Elliott, K. J., and J. M. Vose. 2005. Effects of understory prescribed burning on shortleaf pine (*Pinus echinate* Mill.)/mixed-hardwood forests. Journal of the Torrey Botanical Society 132(2): 236-251.

Elliott, K. J., J. M. Vose, and B. D. Clinton. 2002. Growth of eastern white pine (*Pinus strobus* L.) related to forest floor consumption by prescribed fire in the southern Appalachians. Southern Journal of Applied Forestry 26: 18-25.

Elliott, K. J., R. L. Hendrick, A. E. Major, J. M. Vose, and W. T. Swank. 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. Forest Ecology and Management 114: 199-213.

Fire and Fire Surrogate [FFS] Study Plan. 2001. A national study on the consequences of fire and fire surrogate treatments. U.S. Department of the Interior, U.S. Department of Agriculture, Joint Fire Science Program. www.fs.fed.us/ffs/.

Fowler, C. and E. Konopik. 2007. The history of fire in the southern United States. Human Ecology Review 14 (2): 165-176.

Gilbert, N. L., S. L. Johnson, S. K. Gleeson, B. A. Blankenship, M. A. Arthur. 2003. Effects of prescribed fire on physiology and growth of Acer rubrum and Quercus spp. seedlings in an oak-pine forest on the Cumberland Plateau, KY. Journal of the Torrey Botanical Society 130(4):253-64.

Greenberg, C. H., A. L. Tomcho, J. D. Lanham, T. A. Waldrop, J. Tomcho, R. J. Phillips, and D. Simon. 2007. Short-term effects of fire and other fuel reduction treatments on breeding birds in a southern Appalachian upland hardwood forest. Journal of Wildlife Management. 71(6): 1906-1916.

Greenberg, C. H., C. E. Moorman, C. E. Matthews-Snoberger, T. A. Waldrop, D. Simon, A. Heh and D. Hagan. 2017. Long-term herpetofaunal response to repeated fuel reduction treatments. The Journal of Wildlife Management. DOI: 10.1002/jwmg.21402

Hagan, D. L., T. A. Waldrop, M. Reilly, and T. M. Shearman. 2015. Impacts of repeated wildfire on long-unburned Appalachian plant communities of the southern Appalachian Mountains. International Journal of Wildland Fire 24: 911-920. doi: 10.1071/wf14143

Hammond, D. H., J. M. Varner, J. S. Kush, and Z. Fan. 2015. Contrasting sapling bark allocation of five southeastern USA hardwood tree species in a fire prone ecosystem. Ecosphere, v. 6, no. 7, p. 112.

Harrod, J., P. S. White, and M. E. Harmon. 1998. Changes in xeric forests in western Great Smoky Mountains National Park. Castanea 63: 346-360.

Hille Ris Lambers, J., J. S. Clark, and M. Lavine. 2005. Implications of seed banking for recruitment of southern Appalachian woody species. Ecology 86: 85-95.

Hutchinson, T. F., E. K. Sutherland, and D. A. Yaussy. 2005. Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed-oak forests in Ohio. Forest Ecology and Management 218:210-228.

Hutchinson, T. 2006. Fire and the herbaceous layer of eastern oak forests. Pages 136-151 in: M.B. Dickinson, editor. Fire in eastern oak forests: delivering science to land managers, proceedings of a conference. USDA Forest Service General Technical Report NRS-P-1, Northern Research Station, Newtown Square, Pennsylvania, USA.

Hutchinson, T. F., R. P. Long, R. D. Ford, And E. K. Sutherland. 2008. Fire history and the establishment of oaks and maples in second growth forests. Canadian Journal of Forest Resources. 38: 1184–1198.

Iverson, L. R., T. F. Hutchinson, A. M. Prasad, and M. P. Peters. 2008. Thinning, fire, and oak regeneration across a heterogeneous landscape in the eastern US: 7-year results. Forest Ecology and Management 255: 3035-3050.

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

Jones, M. D. 2005. Hard and soft mast survey report western North Carolina, summer and fall 2005. http://www.ncwildlife.org/.

Kane, J. M., J. M. Varner, E. E. Knapp, and R. F. Powers. 2010. Understory vegetation response to mechanical mastication and other fuels treatments in a ponderosa pine forest. Applied Vegetation Science 13:207-220.

Keeley, J. E. and C. J. Fotheringham. 2005. Plot shape effects on plant species diversity measurements. Journal of Vegetation Science 16(2):249-256.

Kennedy, P. L. and J. B. Fontaine. 2009. Synthesis of knowledge on the effects of fire and fire surrogates on wildlife in U. S. dry forests. Special Report 1096, Oregon State University, Corvallis, Oregon.

Knapp, E. E., B. L. Estes, and C. N. Skinner. 2009. Ecological effects of prescribed fire season: aliterature review and synthesis for managers. USDA Forest Service General Technical Report PSW-GTR-224, Pacific Southwest Research Station, Albany, California, USA.

Kreye J., M. Varner, K. Hiers, and J. Mola. 2013. Toward a mechanism for eastern North American forest mesophication: differential litter drying across 17 species. Ecological Applications 23(8): 1976-1986.

Kuznetsova A., P. B. Brockhoff and R. H. B. Christensen. 2016. lmerTest: Tests in Linear Mixed Effects Models. R package version 2.0-33.

Lafon, C. W., A. T. Naito, H. D. Grissino-Mayer, S. P. Horn, and T. A. Waldrop. 2017. Fire history of the Appalachian region: a review and synthesis. USDA Forest Service General Technical Report SRS-219, Southern Research Station, Asheville, North Carolina, USA.

Lorimer, C. G. 1992. Causes of the oak regeneration problem. P. 14–39 in Oak regeneration: Serious problems, practical recommendations. USDA For. Serv., Gen. Tech. Rep. SE-84, Southeastern Forest Experiment Station, Asheville, NC.

McGinty, D. T. 1972. The ecological roles of Kalmia latifolia L. and Rhododendron maximum L. in the hardwood forest at Coweeta. M.S. Thesis, University of Georgia, p. 81.

Mohr, H. H., T. A. Waldrop, S. Rideout, R. J. Phillips, and C. T. Flint. 2004. Effectiveness of fire and fire surrogate treatments for controlling wildfire behavior in Piedmont forests: a simulation study. Gen. Tech. Rep. SRS-71. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 71-73.

Monk, C. D., D. T. McGinty, and F. P. Day. 1985. The ecological importance of Kalmia latifolia and Rhododendron maximum in the deciduous forest of the Southern Appalachians. Bulletin of the Torrey Botanical Club 112: 187-193.

Nowacki, G. J., and M. D. Abrams. 2008. The demise of fire and mesophication of forests in the eastern United States. Bioscience 58(2): 123-38.

Olsen, C. S., J. D. Kline, A. A. Ager, K. A Olsen, and K. C. Short. 2017. Examining the influence of biophysical conditions on wildland-urban interface homeowners' wildfire risk mitigation activities in fire-prone landscapes. Ecology and Society 22(1):21.

Overby, S. T. and S. C. Hart. 2016. Short-term belowground responses to thinning and burning treatments in southwestern ponderosa pine forests of the USA. Forests 7 (45).

Palik, B. J. and K. S. Pregitzer. 1992. The age and height structure od red maple (Acer rubrum) populations in northern Michigan bigtooth aspen (Populus grandidentata) forests. Canadian Journal of Forest Research. 22:1449-1462.

Parresol, B. R., J. I. Blake, and A. J. Thompson. 2012. Effects of overstory composition and prescribed fire on fuel loading across a heterogeneous managed landscape in the southeastern USA. Forest Ecology and Management 273:29-42.

Peet, R. K., M. T. Lee, M. F. Boyle, T. R. Wentworth, M. P. Schafale, and A. S. Weakley. 2012. Vegetation-plot database of the Carolina Vegetation Survey. Biodiversity & Ecology 4: 243–253.

Phillips, R. J., T. A. Waldrop, and D. M. Simon. 2006. Assessment of the FARSITE model for predicting fire behavior in the southern Appalachian Mountains. Pages 521-525 in: K.S. Connor, editor. Proceedings of the 13th biennial southern silvicultural research conference. USDA Forest Service General Technical Report SRS-92, Southern Research Station, Asheville, North Carolina, USA.

Phillips, R. J., T. A. Waldrop, and D. M. Simon. 2010. Third-year responses of understory woody regeneration to fuel reduction treatments in the Southern Appalachian Mountains. Pages 289-293. In: Stanturf, John A. (editor). Proceedings of the 14th biennial southern silvicultural research conference. General Technical Report SRS-GTR-121. Asheville, NC: USDA Forest Service, Southern Research Station. 614 p.

Pile, L. S., and T. A. Waldrop. 2016. Shortleaf pine and mixed hardwood stands: thirty-four years after regeneration with the fell-and-burn technique in the southern Appalachian Mountains. USDA Forest Service Research Paper SRS-56, Southern Research Station, Asheville, North Carolina, USA.

Pile, L. S., G. G. Wang, B. O. Knapp, J. L. Walker, and M. C. Stambaugh. 2017. Chinese tallow (*Triadica sebifera*) invasion in maritime forests: The role of anthropogenic disturbance and its management implications. Forest Ecology and Management 398:10-24.

Reilly, M. J., K. Outcalt, J. J. O'Brien, and D. Wade. 2017. Effects of repeated growing season prescribed fire on the structure and composition of pine-hardwood forests in the southeastern Piedmont, USA. Forests 8(1):8.

Schiffman, P. M., and W. C. Johnson. 1992. Sparse buried seed bank in a southern Appalachian oak forest: Implications for succession. The American Midland Naturalist 127(2): 258-267.

Schwilk, D. W., J. E. Keeley, E. E. Knapp, J. McIver, J. D. Bailey, C. J. Fettig, C. E. Fiedler, R. J. Harrod, J. J. Moghaddas, K. W. Outcalt, C. N. Skinner, S. L. Stephens, T. A. Waldrop, D. A. Yaussy, and A. Youngblood. 2009. The national fire and fire surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels. 2009. Ecological Applications 19: 285-304.

Stanturf, J. A., D. D. Wade, T. A. Waldrop, D. K. Kennard, and G. L. Achtemeier. 2002. Fire in southern forest landscapes. Pages 607-630 in: D.M. Wear and J. Greis, editors. Southern forest resource assessment. USDA Forest Service General Technical Report SRS-53, Southern Research Station, Asheville, North Carolina, USA.

Stevens, J. T., B. M. Collins, J. W. Long, M. P. North, S. J. Prichard, L. W. Tarnay, and A. M. White. 2016. Evaluating potential trade-offs among fuel treatment strategies in mixed-conifer forests of the Sierra Nevada. Ecosphere 7(9):e01445. 10.1002/ecs2.1445

Thomas, S. C., C. B. Halpern, D. A. Falk, D. A. Liguori, and K. A. Austin. 1999. Plant diversity in managed forests: understory responses to thinning and fertilization. Ecological Applications 9: 864–879.

Tift, B. D., and M. A. Fajvan. 1999. Red maple dynamics in Appalachian hardwood stands in West Virginia. Canadian Journal of Forest Research 29(2):157-165.

Van Lear, D. H. 2004. Upland oak ecology and management. USDA Forest Service General Technical Report SRS-73, Southern Research Station, Asheville, North Carolina, USA.

Van Lear, D. H., and T. A. Waldrop. 1989. History, use, and effects of fire in the southern Appalachians. USDA Forest Service General Technical Report SE-54, Southeastern Forest Experiment Station, Asheville, North Carolina, USA.

Varner J. M., M. A. Arthur, S. L. Clark, D. C. Dey, J. L. Hart, and C. J. Schweitzer. 2016. Fire in eastern North American oak ecosystems: filling the gaps. Fire Ecology 12(2). doi: 10.4996/fireecology.1202001

Vose, J. M., W. T. Swank, B. D. Clinton, R. L. Hendrick, and A. E. Major. 1995. Using fire to restore pine/hardwood ecosystems in the southern Appalachians of North Carolina. Proceedings of Fire Effects on Rare and Endangered Species and Habitats Conference. Coeur d' Alene, Idaho.

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

Waldrop, T. A., L. Brudnak, and S. Rideout-Hanzak. 2007. Fuels on disturbed and undisturbed sites in the southern Appalachian Mountains, USA. Canadian Journal of Forest Research 37: 1134-1141. doi: 10.1139/X06-302

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

Waldrop, T., R. Phillips, and D. Simon. 2010. Fuels and predicted fire behavior in the southern Appalachian Mountains after fire and fire surrogate treatments. Forest Science. 56(1): 32-45.

Waldrop, T. A., D. Hagan, and D. M. Simon. 2016. Repeated application of fuel reduction techniques in the southern Appalachian Mountains, USA. Implications for achieving management goals. Fire Ecology 12 (2): 28-47. doi: 10.4996/fireecology.1202028

White, D. L., T. A. Waldrop, and S. M. Jones. 1990. Forty years of prescribed burning on the Santee fire plots: effects on understory vegetation. Proceedings of an International Symposium. Knoxville, TN. Gen. Tech. Report. SE-69.

Youngblood, A., K. L. Metlen, E. E. Knapp, K. W. Outcalt, S. L. Stephens, T. A. Waldrop, and D. Yaussy. 2005. Implementation of the fire and fire surrogate study—a national research effort to evaluate the consequences of fuel reduction treatment. Pages 315–321 in: C.E. Peterson and D.A. Maguire, editors. Balancing ecosystem values: innovative experiments for sustainable forestry—proceedings of a conference. USDA Forest Service General Technical Report PNW-GTR-635, Pacific Northwest Station, Portland, Oregon, USA.

Zenner, E. K., J. M. Kabrick, R. G. Jensen, J. E. Peck, and J. K. Grabner. 2006. Responses of ground flora to a gradient of harvest intensity in the Missouri Ozarks. Forest Ecology and Management 222: 326-334.

Chapter 2

Abrams, M. D., D. A. Orwig and T. E. DeMeo. 1995. Dendrochronology and successional dynamics of a presettlement-origin white pine-mixed oak forest in the southern Appalachians USA. Journal of Ecology 83,123-133.

Alexander, H. D., M. A. Arthur, D. L. Loftis, and S. R. Green. 2008. Survival and growth of upland oak and co-occurring competitor seedlings following single and repeated prescribed fires. Forest Ecology and Management 256: 1021-1030.

Arno, S. F. and G. E. Gruell. 1986. Douglas-fire encroachment into mountain grasslands in southwestern Montana. Journal of Range Management 39(3): 272-276.

Azeria, E. T., M. Bouchard, D. Pothier, D. Fortin, and C. Hebert. 2011. Using biodiversity deconstruction to disentangle assembly and diversity dynamics of understorey plants along post-fire succession in boreal forest. Global Ecology and Biogeography 20: 119-133.

Ayers, H. B., and W. W. Ashe. 1905. The southern Appalachian forests. US Geological Survey professional paper, 37. Reston: US Geological survey.

Baker, W. L. 1992. Effects of settlement and fire suppression on landscape structure. Ecology 73(5): 1879-1887.

Bartuszevige A. M., and P. L. Kennedy. 2009. Synthesis of knowledge on the effects of fire and thinning treatments on understory vegetation in U.S. dry forests. Oregon State University Agricultural Experiment Station. Special Report no. 1095.

Blankenship, B. A. and M. A. Arthur. 2006. Stand structure over 9 years in burned and fire-excluded oak stands on the Cumberland Plateau, Kentucky. Forest Ecology and Management. 225: 134-145.

Boring, L. R., and W. T. Swank. 1984. The role of black locust (Robinia pseudoacacia) in forest succession. Journal of Ecology 72:749-766.

Bourg, N. A., and D. E. Gill. 2000. Ecological effect of fire on Xerophyllum asphodeloides, a rare Appalachian lily [poster abstract], in Yaussy, D. A., Proceedings: Workshop on Fire, People, and the Central Hardwoods Landscape. Richmond, Kentucky. USDA Forest Service, Northeastern Research Station, Newtown Square, PA. p. 123, General Technical Report NE-274.

Brose, P. H., D. H. Van Lear. 1998. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. Canadian Journal of Forest Research 28(3):331-339.

Brose, P. H., T. Schuler, D. Van Lear, and J. Berst. 2001. Bringing fire back: the changing regimes of the Appalachian mixed-oak forests. Journal of Forestry 99: 30-35.

Burton, J. A., S. W. Hallgren, S. D. Fuhlendorf, and D. M. Leslie Jr. 2011. Understory response to varying fire frequencies after 20 years of prescribed burning in an upland oak forest. Plant Ecology 212(9): 1513-1525.

Certini, G. 2005. Effects of fire on properties of forest soils: A review. Oecologia, 143, 1-10. http://dx.doi.org/10.1007/s00442-004-1788-8

Chapman, G. L. 1950. The influence of mountain laurel undergrowth on environmental conditions and oak reproduction. Ph.D. dissertation, Yale University, New Haven, CT.

Christensen, N. L. 1993. The effects of fue on nutrient cycles in longleaf pine ecosystems. In: S.M. Hermann (ed.), Proceedings of the 18th Tall Timbers Fire Ecology Conference. Tall Timbers Research Station, Tallahassee, FL, 18: 205-214.

Clarke, K. R. 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18:117–117.

Clinton, B. D., J. M. Vose, W. T. Swank, E. C. Berg, and D. L. Loftis. 1998. Fuel consumption and fire characteristics during understory burning in a mixed white pine-hardwood stand in the southern Appalachians. USDA Forest Service Southern Research Station

Clinton, B. D., and J. M. Vose. 2000. Plant succession and community restoration following felling and burning in the southern Appalachian Mountains. Pages 22-29 in W. Keith Moser and Cynthia F. Moser (eds.). Fire and forest ecology: innovative silviculture and vegetation management. Tall Timbers Fire Ecology Conference Proceedings, No. 21. Tall Timbers Research Station, Tallahassee, FL.

Costanza, J. K., J. W. Coulston, and D. N. Wear. 2017. An empirical, hierarchical typology of tree species assemblages for assessing forest dynamics under global change scenarios. PLoS ONE 12(9).

Dale, V. H., S. C. Beyeler, and B. Jackson. 2002. Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. Ecological Indicators 1: 155-170.

Dey, D. C. and G. Hartman. 2005. Returning fire to Ozark Highland forest ecosystems: effects on advance regeneration. Forest Ecology and Management 217: 37-53.

De Cáceres, M. and Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. Ecology 90(12): 3566-3574.

Certini, G. 2005. Effects of fire on properties of forest soils: a review. Oecologia 143:1-10.

Dolan, M. M. and A. J. Parker. 2004. Evergreen understory dynamics in Coweeta Forest, North Carolina. Physical Geography 6: 481-498.

Drewa, P. B., W. J. Platt, and E. B. Moser. 2002. Fire effects on resprouting of shrubs in headwaters of southeastern longleaf pine savannas. Ecology. 83(3): 755–767.

Dufrêne, M. and P. Legendre. 1997. Species assemblages and indicator species: the need for a flexible asymetrical approach. Ecological Monographs 67: 345-366.

Elliott, K. J., R. L. Hendrick, A. E. Major, J. M. Vose, and W. T. Swank. 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. Forest Ecology and Management 114: 199-213.

Elliott, K. J., J. M. Vose, and B. D. Clinton. 2002. Growth of eastern white pine (Pinus strobus L.) related to forest floor consumption by prescribed fire in the southern Appalachians. Southern Journal of Applied Forestry 26(1): 18-25.

Elliott, K. J., and J. M. Vose. 2005. Effects of understory prescribed burning on shortleaf pine (Pinus echinata Mill.)/mixed-hardwood forests. Journal of the Torrey Botanical Society 132(2): 236-251.

Elliott, K. J. and J. M. Vose. 2010. Short-term effects of prescribed fire on mixed oak forests in the southern Appalachians: vegetation response. Journal of the Torrey Botanical Society 137(1): 49-66.

Faith, D. P., P. R. Minchin, L. and Belbin. 1987. Compositional dissimilarity as a robust measure of ecological distance. Vegetatio 69, 57–68.

Fire and Fire Surrogate [FFS] Study Plan. 2001. A national study on the consequences of fire and fire surrogate treatments. U.S. Department of the Interior, U.S. Department of Agriculture, Joint Fire Science Program. www.fs.fed.us/ffs/.

Fitzgerald, S. A. 2005. Fire ecology of ponderosa pine and the rebuilding of fire-resilient ponderosa pine ecosystems. USDA Forest Service General Technical Report PSW-GTR-198.

Franklin, S. B., P. A. Robertson, and J. S. Fralish. 2003. Prescribed burning effects on upland Quercus forest structure and composition. Forest Ecology and Management 184: 315–335.

Garren, K. H. 1943. Effects of fire on vegetation of the southeastern United States. Botanical Review 9(9): 617-654.

Gilbert, N. L., S. L. Johnson, S. K. Gleeson, B. A. Blankenship, M. A. Arthur. 2003. Effects of prescribed fire on physiology and growth of Acer rubrum and Quercus spp. seedlings in an oak-pine forest on the Cumberland Plateau, KY. The Journal of the Torrey Botanical Society 130(4):253-64.

Gilliam, F. S. and M. R. Roberts. 2003. The dynamic nature of the herbaceous layer: synthesis and future directions for research. *In* The herbaceous layer in forests of eastern North America. *Edited by* F. S. Gilliam and M. R. Roberts. Oxford University Press, New York 323-337.

Gonzalez-Tagle, M. A., L. Schwendenmann, J. J. Perez, and R. Schulz. 2008. Forest structure and woody plant species composition along a fire chronosequence in mixed pine-oak forest in the Sierra Madre Oriental, Northeast Mexico. Forest Ecology and Management 256:161-167.

Greenberg, C. H., B. S. Collins, W. H. McNab, D. K. Miller, and G. R. Wein. 2016. Introduction to natural disturbances and historic range of variation: type, frequency, severity, and post-disturbance structure in central hardwood forests. *In:* C. H. Greenberg, B. S. Collins (eds.). *Natural Disturbances and Historic Range of Variation, Managing Forest Ecosystems* pp. 1-32.

Guyette, R. P., R. M. Muzika, and D. C. Dey. 2002. Dynamics of an anthropogenic fire regime. Ecosystems 5:472-486.

Holzmueller, E. J., J. W. Groninger, and C. M. Ruffner. 2014. Facilitating oak and hickory regeneration in mature central hardwood forests. Forests. 5: 3344-3351.

Horn, H. S. 1974. The ecology of secondary succession. Annual Review of Ecological Systems 5: 25-37.

Huddle, J. and S. G. Pallardy. 1999. Effect of fire on survival and growth of Acer rubrum and Quercus seedlings. Forest Ecology and Management 118(1): 49-56.

Hutchinson, T.F., Boemer, R.E.J., Iverson, L.R., Sutherland, S., Sutherland, E.K., 1999. Landscape patterns of understory com- position and richness across a moisture and nitrogen mineraliza- tion gradient in Ohio (U.S.A.) Quercus forests. Plant Ecol. 144, 179-189.

Hutchinson, T. F., R. E. J. Boerner, S. Sutherland, E. K. Sutherland, M. Ortt, and L. R. Iverson. 2005. Prescribed fire effects on the herbaceous layer of mixed-oak forests. Canadian Journal of Forest Resources 35: 877-890.

Hutto RL. 2008. The ecological importance of severe wildfires: Some like it hot. Ecological Applications 18: 1827–1834.

Johnstone, J. F. 2006. Response of boreal plant communities to variations in previous fire-free interval. International Journal of Wildland Fire 15(4): 497-508.

Kane, J. M., J. M. Varner, E. E. Knapp, and R. F. Powers. 2010. Understory vegetation response to mechanical mastication and other fuels treatments in a ponderosa pine forest. Applied Vegetation Science 12(2): 207-220.

Keyser, M. J., P. J. Gould, M. E. McDill, K. C. Steiner, and J. C. Finley. 2008. Classifying patterns of understory vegetation in mixed-oak forests in two ecoregions of Pennsylvania. Northern Journal of Applied Forestry 25(1): 38-44.

Kim, T. N., and R. D. Holt. 2012. The direct and indirect effects of fire on the assembly of insect herbivore communities: examples from the Florida scrub habitat. Oecologia 168(4): 997-1012.

Kruskal, J. B. 1964. Multidimensional scaling by optimizing goodness of fit to a nonmetric hypothesis. Psychometrika, 29, 1–27; Nonmetric multidimensional scaling: A numerical method. 29, 115–129.

Kuddes-Fischer, L. M. and M. A. Arthur. 2002. Response of understory vegetation and tree regeneration to a single prescribed fire in oak-pine forests. Natural Areas Journal 22(1): 43-52.

Lafon, C.W., A.T. Naito, H.D. Grissino-Mayer, S.P. Horn, and T.A. Waldrop. 2017. Fire history of the Appalachian region: a review and synthesis. USDA Forest Service General Technical Report SRS-219, Southern Research Station, Asheville, North Carolina, USA.

Lorimer, C.G. 1992. Causes of the oak regeneration problem. P. 14–39 in Oak regeneration: Serious problems, practical recommendations. USDA For. Serv., Gen. Tech. Rep. SE-84, Southeastern Forest Experiment Station, Asheville, NC.

Lorimer, C. G. 2001. Historical and ecological roles of disturbance in eastern North American forests: 9,000 years of change. Wildlife Society Bulletin 29: 425-439.

Maechler M., P. Rousseeuw, A. Struyf, M. Hubert, and K. Hornik. 2015. Cluster: Cluster Analysis Basics and Extensions. R package version 2.0.3.

McCune, B., R. Rosentreter, J. M. Ponzetti, and D. C. Shaw. 2000. Epiphyte habitats in an old conifer forest in western Washington, USA. The Bryologist 103(3): 417-427.

McCune, B. and J. B. Grace. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, Oregon.

Mohr, H. H., T. A. Waldrop, and V. B. Shelburne. 2002. Optimal seedbed requirements for regenerating table mountain pine. Outcalt, Kenneth W. (ed.) 2002. Proceedings of the eleventh biennial southern skicultural research conference. Gen. Tech. Rep. SRS-48. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 622.

Monk, C. D., D. T. McGinty, and F. P. Day. 1985. The ecological importance of Kalmia latifolia and Rhododendron maximum in the deciduous forest of the Southern Appalachians. Bulletin of the Torrey Botanical Club 112: 187-193.

Moser, W. K., M. J. Ducey, and P. M. S. Ashton. 1996. Effects of fire intensity on competitive dynamics between red and black oaks and mountain laurel. Northern Journal of Applied Forestry 13(3): 119-123.

Murphy, P. A., and G. J. Nowacki. 1997. An old-growth definition for xeric pine and pine-oak woodlands. USDA Forest Service Southern Research Station. General Technical Report SRS-7 Asheville, NC.

Nowacki, G. J., and M. D. Abrams. 2008. The demise of fire and mesophication of forests in the eastern United States. Bioscience 58(2): 123-38.

Oksanen, J., F. Guillaume Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, E. Szoecs, and H. Wagner. 2015. Vegan: the community ecology package, v. 1.8-4

Pausas, J. G. and J. E. Keeley. 2009. A burning story: the role of fire in the history of life. BioScience 59(7): 593-601.

Pavlovic, N. B., S. A. Leicht-Young, and R. Grundel. 2011. Short-term effects of burn season on flowering phenology of savanna plants. Plant Ecology 212:611–625.

Peterson, D. W., P. B. Reich, and K. J. Wrage. 2007. Plant functional group responses to fire frequency and tree canopy cover gradients in oak savannas and woodlands. Journal of Vegetation Science 18(1): 3-12.

Phillips, R., Hutchinson, T., Brudnak, L., Waldrop, T., 2007. Fire and fire surrogate treatments in mixed-oak forests: effects on herbaceous layer vegetation. In: Butler, B.W., Cook, W. (Eds.), The Fire Environment—Innovations, Management, and Policy; Conference Proceedings, 26–30th March 2007; Destin, FL. Proceedings RMRS-P-46. Fort Collins, CO, USA (comps).

Pickett, S. T. A., J. Wu, and M. L. Cadenasso. 1999. Patch dynamics and the ecology of disturbed ground: a framework for synthesis. In: L. R. Walker (ed.). 1999. Ecosystems of Disturbed Ground. Elsevier, Amsterdam. p.707-722.

Pyne, S. J. 1997. Fire in America: A cultural history of wildland and rural fire. 2nd ed. University of Washington Press, Seattle, WA. 680.

Roberts, M. R. 2004. Response of herbaceous layer to natural disturbance in North American forests. Canadian Journal of Botany 82: 1273-1283.

Rogers, P. 1996. Disturbance ecology and forest management: a review of the literature. USDA Forest Service, Intermountain Research Station. General Technical Report INT-GTR-336.

Schwilk, D. W., J. E. Keeley, E. E. Knapp, J. McIver, J. D. Bailey, C. J. Fettig, C. E. Fiedler, R. J. Harrod, J. J. Moghaddas, K.W. Outcalt, C. N. Skinner, S. L. Stephens, T. A. Waldrop, D. A. Yaussy, and A. Youngblood. 2009. The national fire and fire surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels. Ecological Applications 19: 285-304.

Shearman, T. M., G. G. Wang, R. K. Peet, T. R. Wentworth, M. P. Schafale, and A. S. Weakley. 2017. A community analysis for forest ecosystems with natural growth of Persea spp. in the southeastern United States. Castanea 83(1): 3-27.

Small, C.J., and McCarthy. B.C. 2002. Spatial and temporal varia- tion in the response of understory vegetation to disturbance in a central Appalachian oak forest. Journal of the Torrey Botanical Society 129:136-153.

Stanturf, J.A., D.D. Wade, T.A. Waldrop, D.K. Kennard, and G.L. Achtemeier. 2002. Fire in southern forest landscapes. Pages 607-630 in: D.M. Wear and J. Greis, editors. Southern forest resource assessment. USDA Forest Service General Technical Report SRS-53, Southern Research Station, Asheville, North Carolina, USA.

Stark, K. E., A. Arsenault, and G. E. Bradfield. 2006. Soil seed banks and plant community assembly following disturbance by fire and logging in interior Douglas-fire forests of south-central British Columbia. Canadian Journal of Botany 84(10): 1548-1560.

Stephens, S. L., J. D. McIver, R. E. J. Boerner, C. J. Fettig, J. B. Fontaine, B. R. Hartsough, P. L. Kennedy, and D. W. Schwilk. 2012. The effects of forest fuel-reduction treatments in the United States. Bioscience 62: 549-560.

Stottlemyer, A. D., V. B. Shelburne, T. A. Waldrop, S. Rideout-Hanzak, and W. C. Bridges. 2009. Fuel characterization in the southern Appalachian Mountains: an application of landscape ecosystem classification. International Journal od Wildland Fire 18: 423-429.

Stratton, R.L. 2007. Effects of long-term late winter prescribed fire on forest stand dynamics, small mammal populations, and habitat demographics in a Tennessee oak barrens. Knoxville, TN: University of Tennessee. 89 p. M.S. thesis.

Taylor, J. E. 2006. Arundinaria gigantea. In: Fire Effects Information System, Online. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer).

Taylor, A. H. 2007. Forest changes since euro-American settlement and ecosystem restoration in the Lake Tahoe Basin, USA. USDA Forest Service Gen. Tech. Rep. PSW-GTR-203.

Van Lear, D.H., and T.A. Waldrop. 1989. History, use, and effects of fire in the Southern Appalachians. USDA Forest Service General Technical Report SE-54, Southeastern Forest Experiment Station, Asheville, North Carolina, USA.

Van Lear, D. H., P. H. Brose, and P. D. Keyser. 2000. Using prescribed fire to regenerate oaks. Proceedings: workshop on fire, people, and the central hardwoods landscape. General Technical Report NE-274, USDA, Forest Service, Richmond, KY.

van Leeuwen, W. J. D., G. M. Casady, D. G. Neary, S. Bautista, J. A. Alloza, Y. Carmel, L. Wittenberg, D. Malkison, and B. J. Orr. 2010. Monitoring post-wildfire vegetation response with remotely sensed time-series data in Spain, USA and Israel. International Journal of Wildland Fire 19(1): 75-93.

van Wagtendonk, J. W. 2007. The history and evolution of wildland fire use. Fire Ecology Special Issue 3(2): 3-17.

Vose, J. M., B. D. Clinton, and W. T. Swank. 1993. Fire, drought, and forest management influences on pine/hardwood ecosystems in the southern Appalachians. A paper presented at the 12th Conference on Fire and Forest Meteorology, October 26-28, 1993, at Jekyll Island, GA.

Wade D, and J. D. Lunsford. 1989. A guide for prescribed fire in Southern forests. USDA Forest Service Technical Publication R8-TP 11, Atlanta, GA.

Waldrop, T. A., L. Brudnak, and S. Rideout-Hanzak. 2007. Fuels on disturbed and undisturbed sites in the southern Appalachian Mountains, USA. Canadian Journal of Forest Research 37: 1134-1141. doi: 10.1139/X06-302

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

Waldrop, T. A., H. H. Mohr, R. J. Phillips, and D. M. Simon. 2014. The national fire and fire surrogate study: vegetation changes over 11 years of fuel reduction treatments in the Southern Appalachian Mountains. Waldrop, Thomas A., (ed.) 2014. Proceedings, Wildland Fire in the Appalachians: Discussions among Managers and Scientists. Gen. Tech. Rep. SRS-199. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 208 p.

Waldrop, T. A., D. Hagan, and D. M. Simon. 2016. Repeated application of fuel reduction techniques in the southern Appalachian Mountains, USA. Implications for achieving management goals. Fire Ecology 12 (2): 28-47. doi: 10.4996/fireecology.1202028

Waldrop, T. A., and J. D. McIver. 2006. The national fire and fire surrogate study: early results and future challenges. In 'Proceedings of the 13th Biennial Southern Silvicultural Research Conference', 28 February–4 March 2005, Memphis, TN. (Ed. KF Conner). Asheville, NC: USDA Forest Service, Southern Research Station, General Technical Report SRS-GTR-92, pp. 526–530.

Weber, M. G. and M. D. Flannigan. 1997. Canadian boreal forest ecosystem structure and function in a changing climate: impact on fire regimes. Environ. Rev. 5: 145-166.

Willig, M. R. and L. R. Walker. 1999. Disturbance in terrestrial ecosystems: salient themes, synthesis, and future directions. In L. R. Walker (Ed.). Ecosystems of disturbed ground pp. 747–767.

Wright, H. A. and A. W. Bailey (eds.). 1982. Fire Ecology: United States and Southern Canada.