Mixed-severity Fires in the Southern Appalachians

By

Thomas F. DellaRocco

Master of Natural Resources

Assessment and Analysis

Raleigh, North Carolina

2018

Approved by advisory committee:
Dr. Katherine Martin, Chair
Dr. Christopher Moorman, Member
Dr. Barry Goldfarb, Member

October 30, 2018

Abstract

DellaRocco, Thomas. Master of Natural Resources Assessment and Analysis Mixed-severity Fires in Southern Appalachians

Warmer, drier conditions and extended growing seasons are intensifying forest disturbance regimes, particularly wildfire. In the southern Appalachians, historical wildfires were primarily low severity and promoted growth of understory vegetation. Fires also promoted the dominance of fire-tolerant species with thick bark in the overstory, such as oaks (*Quercus* spp.) and hickories (*Carya* spp.) over mesic, fire-sensitive species, such as maples (*Acer rubrum* L., *Acer saccharum* Marsh.), tulip-*poplar* (*Liriodendron tulipifera* L.) and birch (*Betula lenta* L., *Betula alleghaniensis* Britton). Fire exclusion over the last century has contributed to a gradual shift toward mesic species dominance, which has likely altered the forest understory. Further, these mesophytic species might experience mortality in the overstory in the projected future conditions that include longer dry periods and more frequent and intense wildfires. In fall 2016, the southern Appalachians experienced exceptional drought and multiple wildfires, some of which were novel and included moderate and high severity burned areas. We investigated the impacts of fire severity on forest understory, seedling, and groundlayer composition. We measured the understory (woody species <5 cm dbh, >0.5 m tall), woody seedlings (<0.5 m tall), and ground-layer (herbaceous species), across burned watersheds within two large wildfire complexes and compared those responses to adjacent, unburned watersheds. Across the burned watersheds, we assigned three burn severity classes, based on tree and evergreen shrub mortality, forest floor depth, mineral soil exposure and bole scorch height. We did not find a positive effect of fire on sapling density or richness in any of the burn severity classes. However, oak saplings occurred in similar densities to maples in higher burn severity plots. Mountain laurel and rhododendron shrubs had lower densities in burned areas. Additionally, seedling densities were higher in burned areas, particularly mesophytic seedlings such as tulip-*poplar* and birch. The groundlayer cover and richness was lower in areas affected by higher severity fires.

ACKNOWLEDGEMENTS

We would like to thank Dr. Katherine Elliott from the USDA Forest Service Coweeta Hydrologic Laboratory for her guidance through this research, including the study design and fire severity analysis. Thank you Molly Suminski for field assistance. Additionally, we appreciate the assistance other employees from Coweeta Hydrologic Laboratory provided, including Patsy Clinton and Joel Scott.

TABLE OF CONTENTS

1. Introduction:

Major changes have occurred in the forests of eastern North America since European colonization. Records indicate eastern North America was repeatedly cleared of forests and converted to agriculture (McEwan et al. 2011). Based on land used cover data, these forests were substantially cut from 1850 to 1880, and most of the forest had been cleared by 1930. Even in remaining forests, understories and regeneration were altered by cattle and hog grazing. The disturbance ecology of remaining forests were also altered by European colonists, particularly wildfire regimes (Nowacki and Abrams 2008). Before European colonization of North America, wildfire ignitions were started by Native Americans and lightning strikes (Abrams 1992, Brose et al. 2014, Flatley et al. 2013, Nowacki and Abrams 2008, Nowacki and Abrams 2015). Frequent fire led to regional dominance of oak-hickory forests in eastern North America, including the southern Appalachians (Arthur et al. 2012). By the end of the 19th century, human ignited fires decreased, and fires were actively suppressed (McEwan et al. 2011). From the 1920's to present, there is a trend toward decreasing fire frequency and area burned (Nowacki and Abrams 2008, McEwan et al. 2011). As a result of this and other factors, forest composition has transitioned from mixed-oaks and hickories to shade-tolerant, mesophytic, fire-sensitive species such as maples (*Acer rubrum* L., *Acer saccharum* Marsh.), tulip-*poplar* (*Liriodendron tulipifera* L.), and birch (*Betula lenta* L., *Betula alleghaniensis* Britton), a process that has been termed *mesophication* (Abrams 1992, Brose et al. 2014, Elliott and Swank 2008, Nowacki and Abrams 2008, Nowacki and Abrams 2015).

1.1 Forest Understories

As land use and disturbance histories of forests shifted across eastern North America from the late 19th century, forest understories were affected. Due to widespread disturbance from clearing, grazing, and agricultural conversion, the groundlayer of eastern forests, including herbaceous species and woody seedlings and saplings, experienced substantial change (Elliott and Swank 2008, Flatley et al. 2013, McEwan et al. 2011). A decrease in herbaceous species richness has been observed in the Piedmont region of North Carolina from 1977 to 2000, characterized by the decline or disappearance of less abundant species (Gilliam 2014, McEwan et al. 2011, Schwartz et al. 2016).

The groundlayer of forest understories (herbaceous species and woody seedlings) play important roles in ecosystem function in forests. Within the southern Appalachians, the forest groundlayer contains the greatest plant biodiversity in the region (Gilliam 2007). This is particularly true for mesic cove forests, which have an outstanding amount of herbaceous biodiversity, with up to 43.75± 3.03 species per 0.25 m^{-2} plots in reference areas (Elliott et al. 2014, Ford et al. 2000). Although herbaceous plants account for about 1% of the above ground biomass in a forest they account for up to 20% of the foliar litter (Gilliam 2007). Additionally, herbaceous plants account for a larger portion of net primary production (NPP) relative to total above ground NPP (Muller 2003). Herbaceous plants also have a disproportional influence on nutrient cycling relative to their biomass in forests, containing higher concentrations of nitrogen, phosphorus and potassium in their foliage compared to tree species. The foliage of herbaceous plants can decompose at twice the rate of trees, further facilitating nutrient cycling in forest. Spring ephemeral plants have a unique role in forest ecosystems because they take up nitrogen in the spring and act as a release of nutrients for other plants during the growing season as they die and decompose, reducing potential nitrogen loss (Gilliam 2007). Furthermore, herbaceous vegetation provides critical food to wildlife, such as white-tailed deer (*Odocoileus virginianus*) (Edwards et al. 2004).

In addition to herbaceous species, the forest groundlayer contains the woody seedlings that may become recruited into the overstory. Within the groundlayer, competition between herbaceous species and seedlings can affect woody species recruitment (Gilliam 2007). For example, ferns have a negative effect on the emergence and survivorship of tree seedlings (George and Bazzaz 2003). Other tall herbaceous plants that form groups, such as wood nettle (*Laportea canadensis* L.), likely have a similar effect on growth and recruitment of tree seedlings (Gilliam 2007). Woody seedlings that successfully recruit in the groundlayer form the future forest overstory. Assessing seedlings growing after a disturbance can give an indication of the species that germinate or basal sprout after a disturbance and might indicate shifts in species composition.

Mature deciduous hardwood forests, including the second-growth forests of the southern Appalachians, have complex structure, including multiple sapling layers. In addition to the groundlayer, much of the southern Appalachians includes a significant shrub layer. Two shrub species are particularly dominant and important: mountain laurel (*Kalmia latifolia* L.) and great laurel or rhododendron (*Rhododendron maximum* L.) (Day et al. 1988). Both ericaceous shrubs often create dense colonies, which affect overstory regeneration by preventing recruitment of tree seedlings (Elliott et al. 2011). Rhododendron inhibits the recruitment and growth of tree seedlings by limiting light (Nilsen et al. 2001, Wurzburger and Hendrick 2007). Equally or more significantly rhododendron reduces soil water and nutrients.

1.2 Mesophication

As outlined above, eastern forests have experienced widespread change since the arrival of European colonists (Abrams 1992, Brose et al. 2014, Flatley et al. 2013, Nowacki and Abrams 2008, Nowacki and Abrams 2015. Much of this change is attributed to altered fire regimes. For example, a study of fires scars from southern Appalachian forests in Tennessee and North Carolina by Flatley and colleagues (2013) estimated low severity fires occurred in four to eight-year cycles from approximately 1756 until the 1900's when fire suppression began. Likewise, Stambaugh and colleagues (2016) reported the mean fire frequencies in post oak woodlands of south-central Tennessee ranged from three to five years prior to 1835 (records were as early as 1631). From 1824 to 1926, mean fire frequencies ranged from three to ten years. After 1926 the mean fire frequencies ranged from six to ten years. There was a 67-year fire free period from 1936 to 2003 at one site in this study.

Expanding on the "oak and fire hypothesis," some researchers including McEwan and colleagues (2011) argue that the decline in oak regeneration and recruitment is the result of multiple interacting drivers. These interacting drivers have led to conditions where mesic species such as maples have greater regeneration relative to their importance as overstory trees (Abrams 1998). One cause for the increase in mesic compared to more xeric species is a change in climatic conditions. Much of the current oak dominance might have occurred during a period characterized by severe multi-year droughts; however, the past 100 years are characterized by decreased drought and increased moisture availability, and a slight (0.5^{o} C) decline in average maximum summer (June-August) temperatures (McEwan et al.2011). This shift in climate conditions that might favor mesic species coincides with the decline in oak regeneration during the early 20th century (Abrams 1998). Increases in herbivore species populations over the 20th

century, particularly white-tailed deer, might also have contributed to declines in oak regeneration (McEwan et al. 2011). Invasive pests and pathogens have also influenced the current structure and composition of forests across the southern Appalachians. For example, a 1934 vegetation assessment of the Coweeta basin in the southern Appalachians identified American chestnut (*Castanea dentata* (Marshall) Borkh.) as the most important species (Elliott and Swank 2008). Chestnut blight was subsequently reported in the Coweeta Basin in 1926, and by the 1990's, American chestnut was functionally extinct. Throughout its range, American chestnut was replaced by chestnut oak (*Quercus montana* Willd.) and red maple (Elliott and Vose 2010). This suggests that the decline of chestnut might have contributed to the increase in red maple and other mesic species throughout much of the landscape.

The changes in forests that have been linked to mesophication also extend to the sapling layer and groundlayer in the southern Appalachians. Similar to the overstory, fire is important for maintaining the sapling layer (Holzmueller et al. 2009). Following fire there is an initial increase of nitrogen available in the soil (Knoepp et al. 2009). This increase in nitrogen is followed by an increase in herbaceous cover (Elliott and Vose 2010). Fires also increase light to the forest floor, which leads to increased growth of many species (McEwan et al. 2011). In addition, fires clear the leaf litter, which can be critical for seed germination (Nowacki and Abrams 2008). After high intensity prescribed fire, Elliott and colleagues (2009) found an increase in huckleberry (*Gaylussacia ursina*), blackberry (*Rubus allegheniensis* Porter), and grasses. Overall more intense fire led to increased herbaceous diversity (Elliott et al. 2009, Holzmueller et al. 2009).

1.3 Changing Climate and Forest Dynamics:

Climate change is expected to have multiple direct and indirect effects on forest composition and structure, through changing temperatures and precipitation, as well as increasing forest disturbances (Dale et al. 2001). The southern Appalachians are expected to experience higher annual average temperatures, increasingly variable precipitation with more frequent droughts, and a new, distinct wildfire season characterized by more frequent and severity fires (Liu et al. 2013, Mitchell et al. 2014). Additionally, wildfires are expected to be more prevalent outside the growing season. An increase in consecutive dry days is also predicted for the southern Appalachians (Vose and Elliott 2016). The shifts to increasing dry periods and droughts and more frequent and intense wildfire suggest conditions might once again favor xerophytic oaks over more mesic species (Mitchell et al. 2014, Vose and Elliott 2016). The increased dominance of oaks and other xeric species as a result of a warmer and dryer climate might occur without management; however, Vose and Elliott (2016) suggest that management might mitigate the degradation of some of the forest during this transition. A transition in forest composition back to xerophytic species might improve forest resilience in warmer, dryer conditions. The deep root structure and xylem anatomy of oaks facilitates higher drought tolerance compared to co-occurring mesic species (Roman et al. 2015, Vose and Elliott 2016).

Motivated to improve forest resilience to climate change and to improve wildlife food sources (acorns), forest managers and land owners are interested in restoring oak forest structure and composition in the southern Appalachians (Elliott and Vose 2010, Vose and Elliott 2016). Prescribed fire is seen as an important tool in this restoration process, as well as a tool to reduce fuel loading in the context of increasing wildfire risk (Stambaugh et al. 2016). Single low severity prescribed fires used to restore oaks and hickories in the southern Appalachians have been relatively ineffective. For example, following low to moderate intensity fire Elliot and Vose

(2010) found no significant difference between burned and unburned sites for shrubs, seedlings, or herbaceous plants. In their study, the seedlings and saplings of oaks and hickory decreased, while species considered more fire sensitive such as maple and sourwood (*Oxydendrum arboretum* (L.) DC.) experienced no significant changes. Chiang and colleagues (2008) determined silvicultural thinning followed by prescribed fire only had a minimal benefit to oak regeneration because of maple recruitment in the gaps that were formed. However, Phillips and colleagues (2007) determined thinning followed by prescribed fire increased herbaceous cover and species richness. Oaks have been observed to benefit from high intensity prescribed fire in the southern Appalachians (Elliott et al. 2009). High intensity prescribed fire also leads to an increase in ericaceous shrubs and greater herbaceous diversity. In addition to fire severity, fire frequency has an important role in forests. Holzmueller and colleagues (2009) found oak-hickory forests that had been burned as much as three times over 20 years showed an increase in oaks and hickory seedlings and increased species richness compared to unburned stands. These researchers showed species compositions did not change with burn frequency.

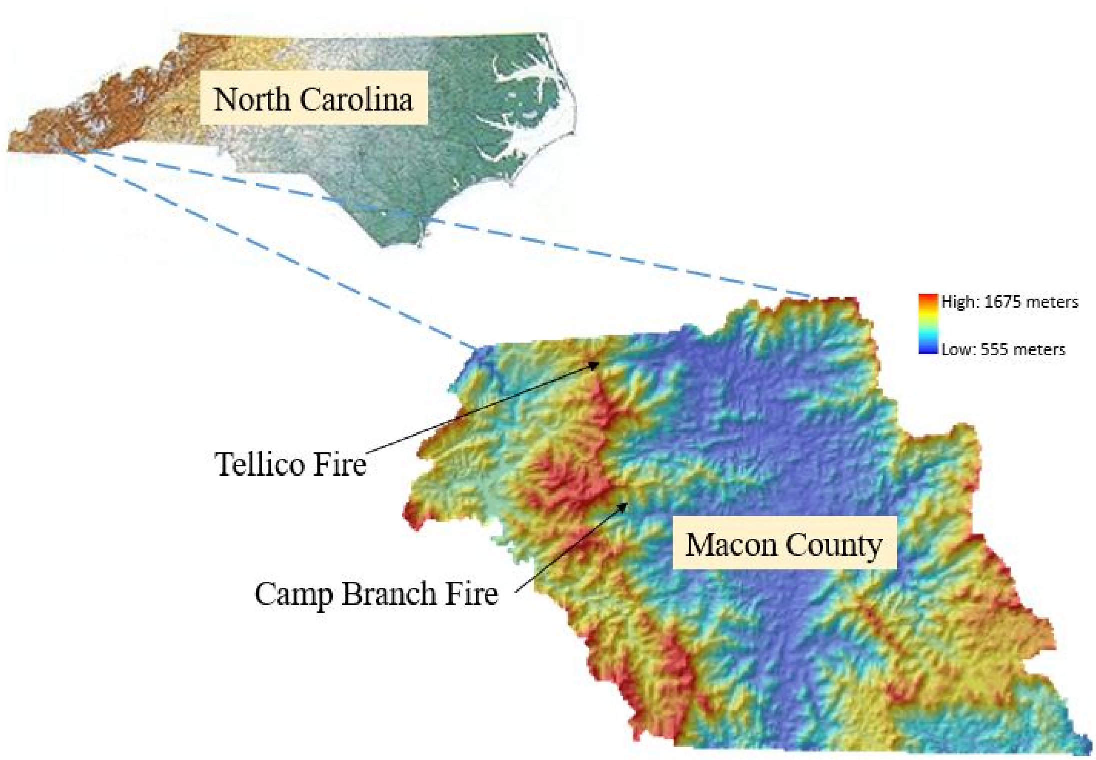

Figure 1: Physical map of study areas in Macon County, North Carolina. This map indicates the elevation of the region, as well as the two study sites.

1.4 Objectives of This Study

In the fall of 2016 during drought conditions, the southern Appalachians experienced wildfires uncharacteristic of the region (Case and Zavodsky 2018). Rather than low severity burns that might have impacts similar to most prescribed fires, the 2016 wildfire season included fires with moderate and high severity patches. Observations following the fires indicated patches of tree mortality or complete consumption of the forest floor, where all leaf litter and duff were removed, leaving only mineral soil. As this fire behavior is unusual and likely outside the knowledge based from prescribed fires in the region, the effects of such mixed-severity fires on

Appalachian hardwood forests are uncertain. To determine the effects of these fires, our study examined the vegetation response of two fire complexes with mixed-severity patches within the Nantahala National Forest in southwestern North Carolina: Camp Branch and Tellico (Fig. 1). Our study specifically measured the vegetation response to the 2016 wildfires in the growing season directly following the fires. Vegetation plots were established along sub-watersheds that were included in the larger burn complexes. As pre-fire data was not available for these sites, we paired these watersheds with nearby unburned (UB) watersheds. By examining the herbaceous groundlayer, seedling, and sapling layers of our study areas, we will be able to address the following hypotheses:

1) The increased resource availability (light, nutrients) following low (L) severity fire will benefit herbaceous species. Therefore, the percent cover and species richness of herbaceous vegetation will be greater in areas with L compared to UB areas. However, we expect these levels to decrease with increasing fire severity.
2) Mountain laurel and rhododendron will have decreasing densities of seedlings and saplings with increasing fire severity.
3) Recruitment of xeric woody species seedlings and saplings will increase with increasing fire severity, whereas mesic species will decrease in density.

2. **Methods**:

2.1 Site Description:

The Tellico and Camp Branch Fire sites are located in the Nantahala National Forest, western North Carolina (35.210174, -83.661393) (Fig. 1). The forests of these areas are mixed-hardwood, with variable composition based on topography and elevation. For example, forest communities on xeric ridges are different than those in rich coves (Simon et al. 2005). Rich Cove Forests (Montane Intermediate Subtype) were a dominant community type found in our study plots. Overstory species that typify this forests type include tulip-*poplar*, red maple, buckeye (*Aesculus flava* Aiton), white ash (*Fraxinus Americana* L.), and basswood (*Tilia Americana* L.) (Schafale 2012). Unique herbaceous species in Rich Cove Forests include *Actaea racemose*, *Caulophyllum thalictroides, Prosartes lanuginose, Aruncus dioicus, Collinsonia canadensis, and Laportea canadensis.* The research area also contained some Chestnut Oak Forests (Dry Heath Subtype), which are found on xeric slopes and ridges. These forests are dominated by chestnut oak with the addition of other oaks including scarlet oak (*Quercus coccinea* Münchh) and red oak (*Quercus rubra* L.). The sapling layer is dominated by mountain laurel, huckleberry *(Gaylussacia* spp.), and blueberry *(Vacinium s*pp.). Other species that make this community unique include the presence of American chestnut, flame azalea (*Rhododendron calendulaceum* Michx.), buffalo nut (*Pyrularia pubera*), and *Carex pensylvanica* (Schafale 2012).

At Tellico the average annual precipitation is 101 to 305 cm and the average temperature is 2 to 14ºC (Soil Survey Staff 2017). The elevation of this study site ranges from 357 to 2030 m. Camp Branch has an average annual precipitation of 94 to 305 cm and the average temperature is 2-14ºC. The elevation of Camp Branch is between 366 and 4727 m. For both Camp Branch and Tellico, the prevalent soil type is a fine-loamy, parasesquic, mesic Typic Hapludults (Evard series) (Soil Survey Staff 2017). Prior to the 1920’s, this region experienced infrequent low severity fires in early spring and late fall. The frequency and severity of these fires varied depending on abiotic factors such as topography, precipitation, and wind (Pyne 1984).

Additionally, biotic factors such as vegetation influence fuels, which, in turn, effect fire behavior.

An exceptional drought effected the southern Appalachians in late summer of 2016 (Case and Zavodsky 2018). Exceptional drought is determined by six indices, including the CPC Soil Moisture Model Percentiles, Palmer Drought Severity Index, Percent of Normal Precipitation, remotely sensed Satellite Vegetation Health Index, Standardized Precipitation Index, and U.S. Geological Survey Daily Streamflow Percentiles (Svoboda et al. 2002). Exceptional drought is classified D4, the highest on a scale from D1 to D4 from the Drought Monitor. The impacts of an exceptional drought include “exceptional and widespread crop/pasture losses”, “shortages of water in streams, reservoirs, and wells creating water emergencies”, and a “fire risk that is exceptionally dangerous” (Svoboda et al. 2002). During the exceptional drought, 26 wildfire ignitions that required management occurred in the Nantahala Ranger District alone (M. Wilkins, District Ranger, Nantahala Ranger District, personal communication). These fires burned over 100,000 hectares. The 2016 fires in the Nantahala National Forest were human caused, including both accidental ignitions and arson (M. Wilkins, District Ranger, Nantahala Ranger District, personal communication). The Forest Service actively worked to extinguish these fires, which were extinguished by precipitation in late-November. The Tellico fire burned 5,619 hectares and Camp Branch burned 1,385 hectares (Fig. 2 and 3). Both of these fires were chosen for our study because of the opportunity to examine mixed-severity effects (Katherine Elliott, USFS, personal communication).

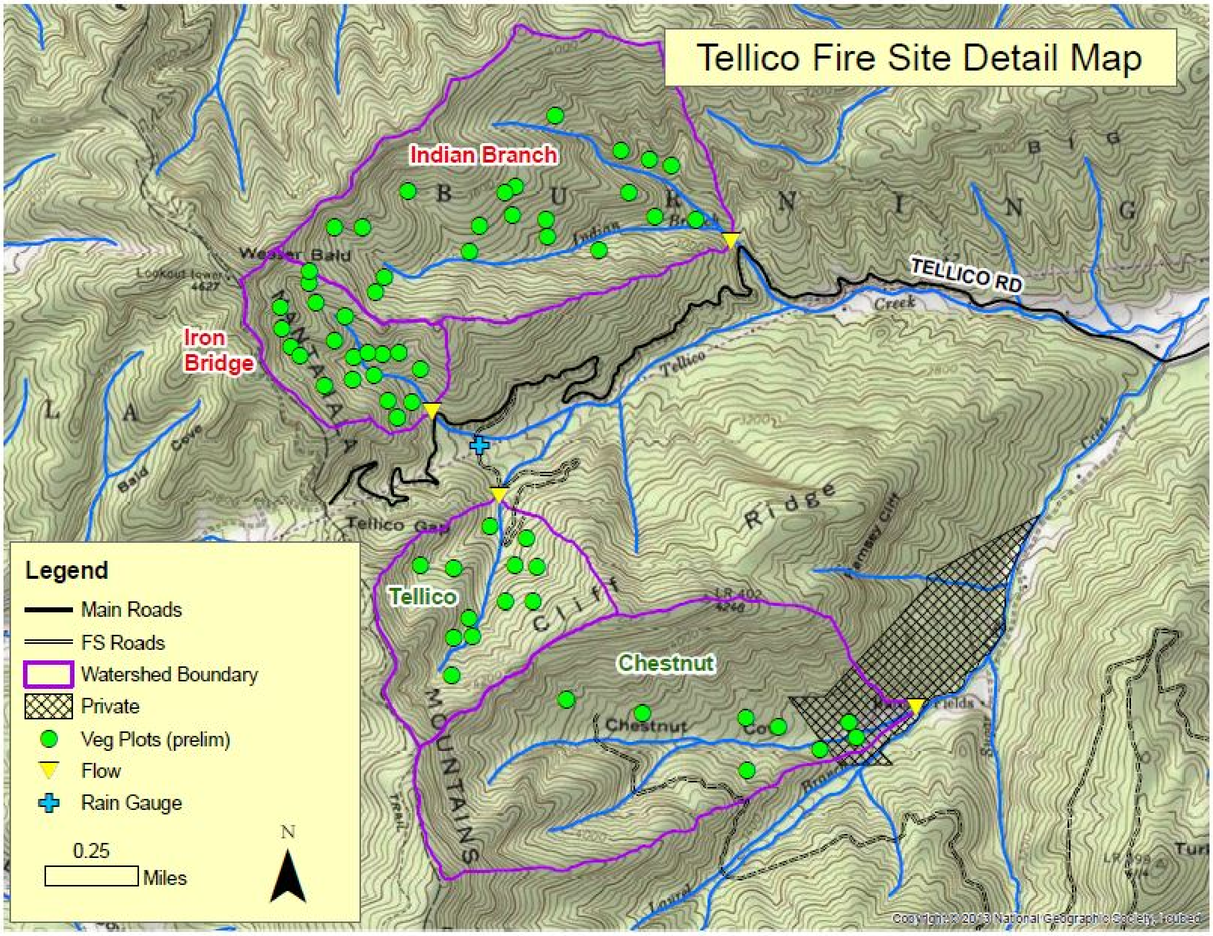

Figure 2: Topographic map of Tellico. Watersheds are outlined in purple and research plots are indicated in green. Tellico and Chestnut are UB reference watersheds, while Indian Branch and Iron Bridge are B watersheds.

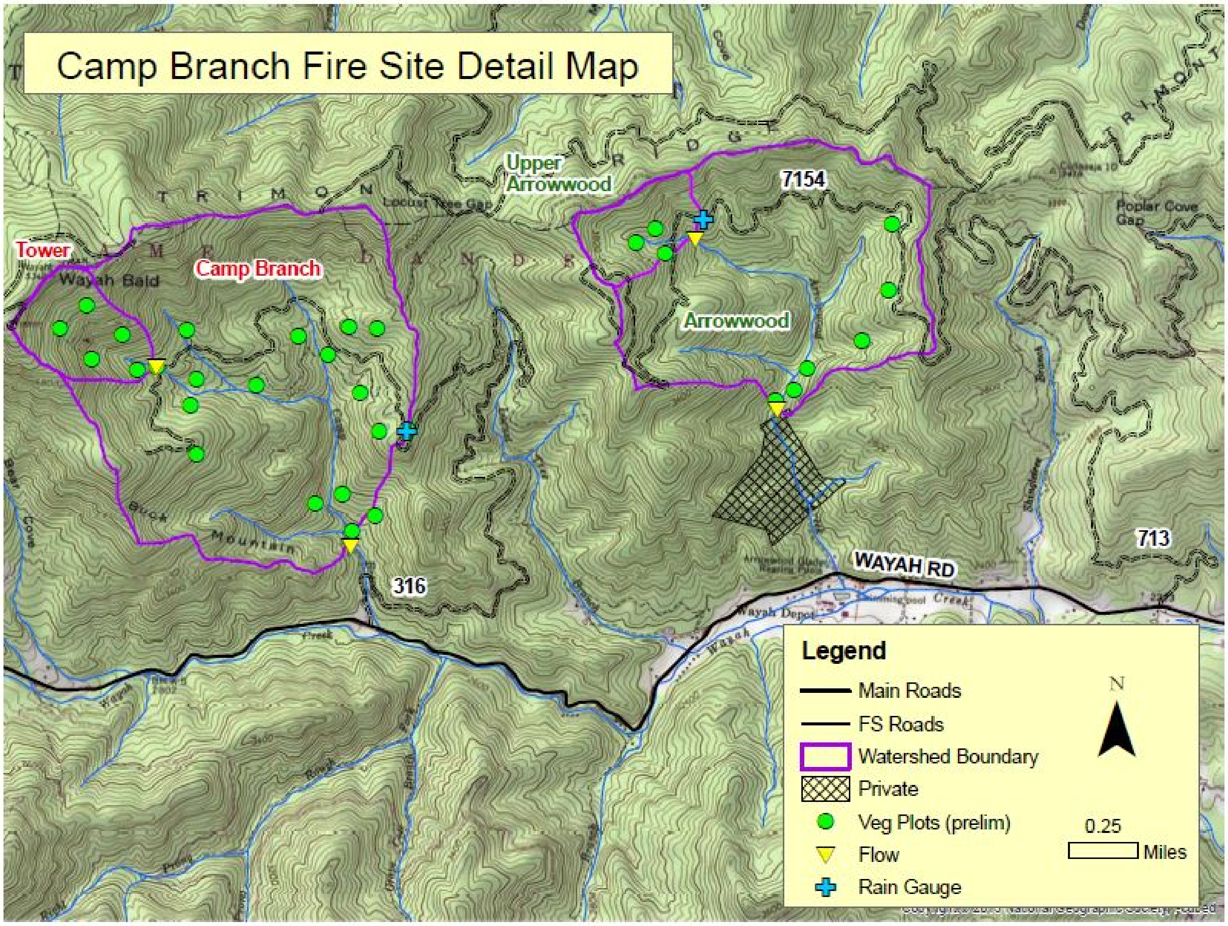

Figure 3: Topographic map of Camp Branch. Watersheds are outlined in purple and research plots are indicated in green. Arrowwood and Upper Arrowwood are UB reference watersheds, while Camp Branch and Tower are B watersheds.

3.2 Field Methods:

Sample plots were located in burned (B) and adjacent, UB watersheds for both sampled wildfires. The Tellico site included two UB watersheds and two B, while the Camp Branch site included one UB watershed and one B. Each UB watershed contained 12 sample plots, and each B contained 20 plots, totaling 96 sample plots, each 7 m in radius (Figure 4). We sampled vegetation by layers: saplings, seedlings (including basal sprouts), and herbaceous groundlayer. Saplings included woody stems at least 0.5 m tall and if over 1.37 m tall, less than 5.0 cm diameter at breast height (DBH). The data recorded from the sapling layer included species, stem count, and DBH. All herbaceous species and woody species were inventoried in two 1-m x 1-m herbaceous groundlayer quadrats. These two quadrats were located 1 m from the center of the plot and oriented parallel to the slope. In herbaceous groundlayer plots, we recorded species and percent cover for herbaceous vegetation. In these plots we also recorded species, percent cover, and stem count for woody species under 0.5 m tall. Total percent cover of groundlayer quadrats was determined for bare ground and vegetation, including herbaceous plants and included woody species. Percent cover data was recorded based on a visual assessment.

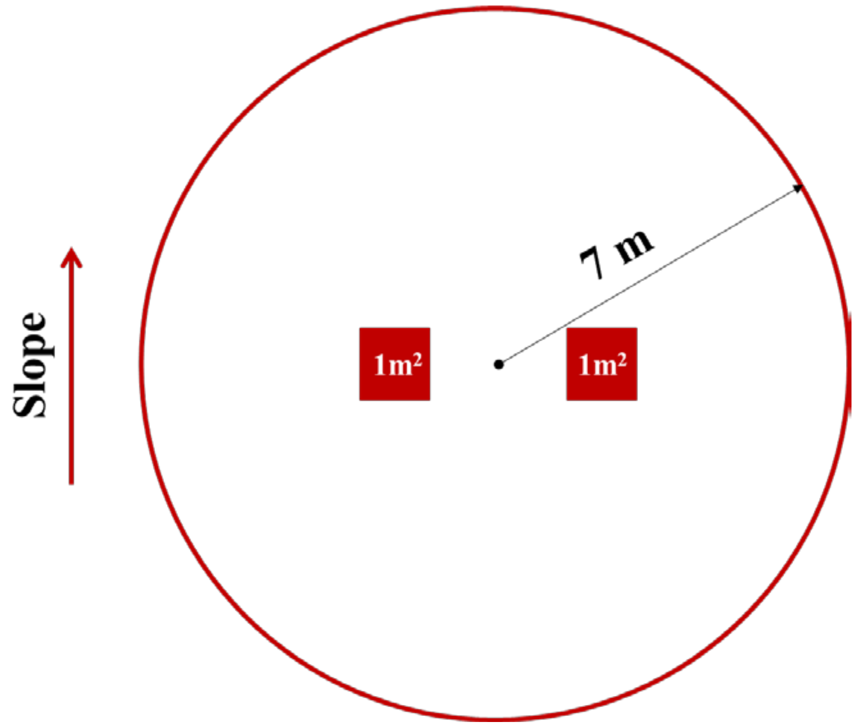

Figure 4: Diagram representing vegetation research plots used in this study. This plot is 7 m in radius. This 7 m area accounts for the sapling vegetation inventoried in this study. There are two 1 m^2 quadrats located 1 m from the center of the research plot, oriented parallel to the slope. These quadrats account for the herbaceous vegetation and seedlings inventoried.

3.3 Data Analysis:

Plot-level burn severity values were calculated based on tree mortality, organic layer depth, bole char height, tree basal area loss, and exposed mineral soil. Each of these indicators was given a rank from one to five, one being minimally impacted and five highly impacted. These ranked values were then averaged to estimate a burn severity rank for each plot (Table 1). A burn severity value of 1.2 or less was considered a low severity burn, 1.3 to 2.9 a low to moderate, and over 3.0 a moderate to high severity fire. For our analysis we combined low to moderate and moderate burn severity classes, as well as moderate to high and high burn severity classes.

We evaluated diversity of saplings, seedlings, and groundlayer vegetation with species richness. For saplings, species richness was determined for each 7-m radius research plot. The species richness of all research plots in a burn severity class was then averaged to determine the average species richness for each burn severity class. For all calculations, we used individual research plots as replicates. Stem density was used to gain a perspective of sapling survival and regeneration for all woody stems and for specific species. Stem density was determined for each species in a 7-m radius research plot then averaged over each burn severity class. Additionally, stem density was used to analyze seedling recruitment. Seedling recruitment was calculated with the same method as the sapling density, but calculated over a 1-m^2 area. Herbaceous groundlayer species richness was also used to determine differences in recruitment amongst burn severity classes. Herbaceous groundlayer species richness was calculated similar to sapling species richness, but was calculated over a 1-m^2 area. The herbaceous groundlayer was analyzed based on total percent cover of vegetation. The total percent cover that was determined in the field in individual 1-m^2 quadrats was used to calculate an average for each 7-m plot. The average total percent cover calculated for each 7-m plot was used to determine an average amongst burn severity classes. We carried out a Kruskal-Wallis test (non-parametric analysis of variance) on sapling stem density and diversity, shrub density in the sapling layer, seedling density, and herbaceous groundlayer percent cover and herbaceous groundlayer diversity using R statistical software (R Core Team 2013). Following this Kruskal-Wallis test we carried out Dunn's test of multiple comparisons using R statistical software to determine if significant differences existed among individual burn severity classes.

Burn Severity Class	Parameter Ranges				
	Tree Mortality (%)	**BA Loss (%)**	**Bole Char Height (m)**	**Oe+Oa Depth (cm)**	**Exposed Mineral Soil (%)**
Moderate-High	≥31	≥16	>0.8	≤0.8	≥16
Low-Moderate	11-30	6-15	0.2-0.8	1.4-0.9	1-15
Low	0-10	0-5	0-0.1	>1.4	0

Table 1: Burn severity index created for the wildfires that effected the southern Appalachian Mountains in fall of 2016. Burn severity classes were based on measures taken across burned plots. Measured parameters included tree mortality, basal area (BA) loss, tree bole char height, forest floor (Oe+Oa) depth, and percentage exposed mineral soil. Tree mortality and basal area loss were assessed in September 2017 (one growing season after the wildfires, and before leaf fall), all other parameters were measured January–February 2017 (less than three months after the wildfires).

4.0 Results:

4.1 Saplings:

In the first growing season following the 2016 fall wildfires, there was variation in sapling stem density based on burn severity class. Our measure of saplings includes all tree species, shrubs, and woody vines over 0.5 m tall and if over 1.37 m tall, less than 5.0 cm DBH. Overall, there was a significant difference in sapling density among UB plots and burn severity classes based on a Kruskal-Wallis test ($p < 0.0001$, df= 3). Using Dunn's test of multiple comparisons, we determined there was a significant difference among UB and all plots that were burned (L, L-M, and M-H). UB plots had the highest average sapling density (1.52±0.18 (SE) stems m^{-2}, Fig. 1). There was a small but gradual increase in sapling density among B plots with increasing burn severity (L= 0.23±0.07 stems m^{-2}, low to moderate (L-M= 0.24±0.03 stems m^{-2}, Fig. 1). Among B plots, the greatest sapling density was in medium to high (M-H) burn severity plots (0.36±0.09 stems m^{-2}, Fig. 1).

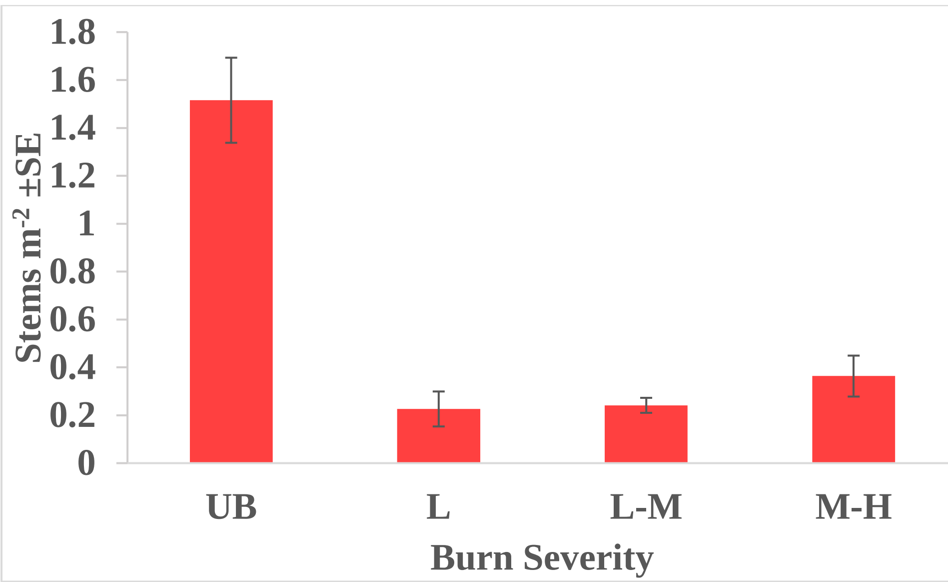

Figure 5: Average number of saplings m^{-2} by tree species within each burn severity class. The error bars represent standard error.

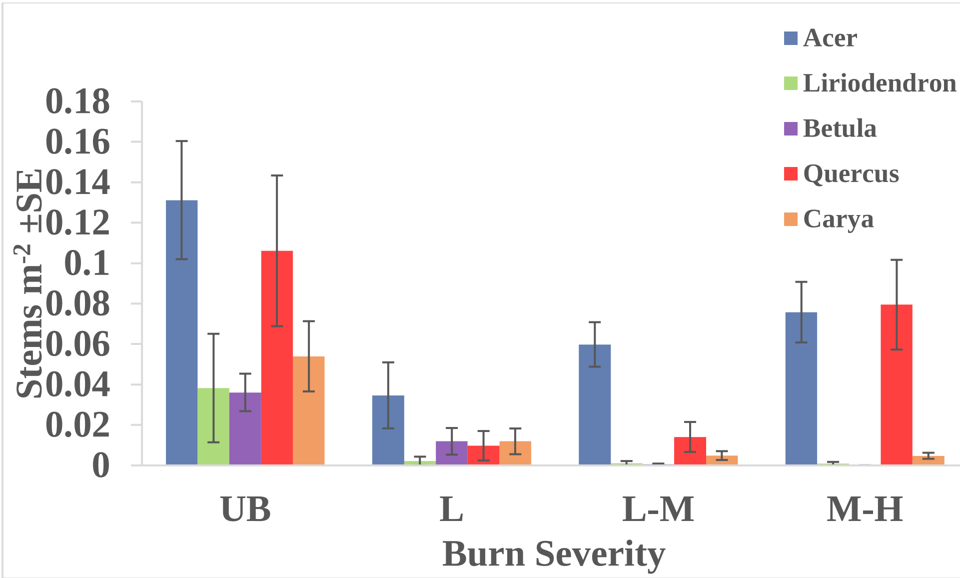

Figure 6: Sapling species density across burn severity classes. Average number of sapling species of interest m^{-2} within each burn severity class. The error bars represent standard error.

We also documented evidence that fires affect sapling community composition (Fig. 6). Oak and maple sapling densities were similar across UB (0.11±0.04 vs. 0.13±0.03 stems m^{-2}) and B plots (0.05±0.01 vs. 0.07±0.01 stems m^{-2}) (Fig. 6). Additionally, within M-H plots, oak sapling densities (0.08±0.02 stems m^{-2}) were comparable to maple saplings (0.08±0.01 stems m^{-2}) (Fig. 6). Maples exceed oaks in both L (0.03±0.02 vs. 0.01±0.01 stems m^{-2}) and L-M (0.06±0.01 vs. 0.01±0.01 stems m^{-2}) burn severity plots (Fig. 6).

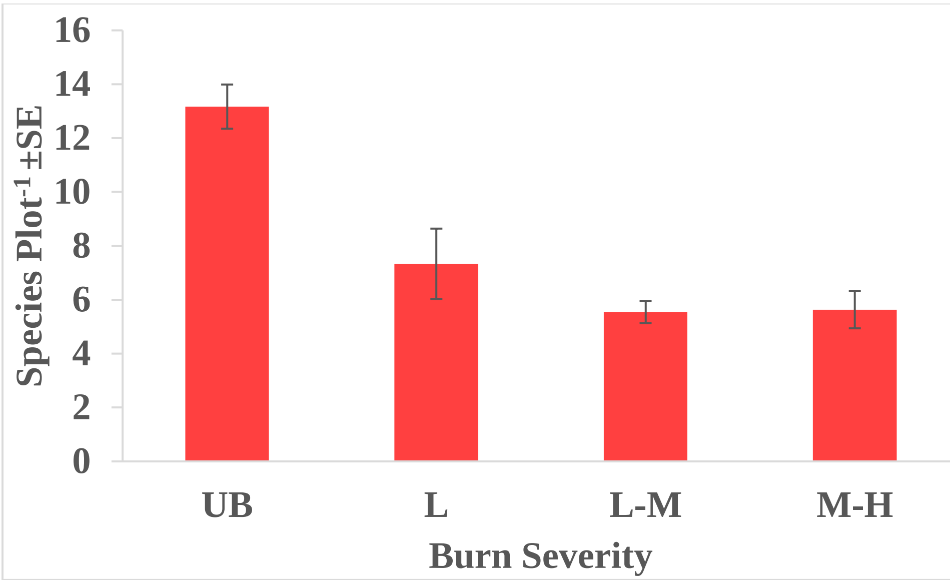

Figure 7: Average sapling species richness plot^{-1} within each burn severity class. The error bars represent standard error.

The sapling species richness was lower in B areas. We determined there was a significant difference in species richness among UB plots and all burn severity classes ($p < 0.0001$). Furthermore, there was a significant difference between UB plots and all other burn severity classes. UB watersheds had the greatest sapling species richness (13.16±0.83 species plot^{-1}, Fig. 7). Species richness declined with burn severity. Amongst B plots, L severity burned plots had the greatest species richness (7.33±1.31 species plot^{-1}, Fig. 7), and L-M and M-H were similar, with M-H having a slightly greater species richness (5.54±0.41 species plot^{-1} and 5.63±0.69 specie plot^{-1}, Fig. 7).

4.2 Shrubs:

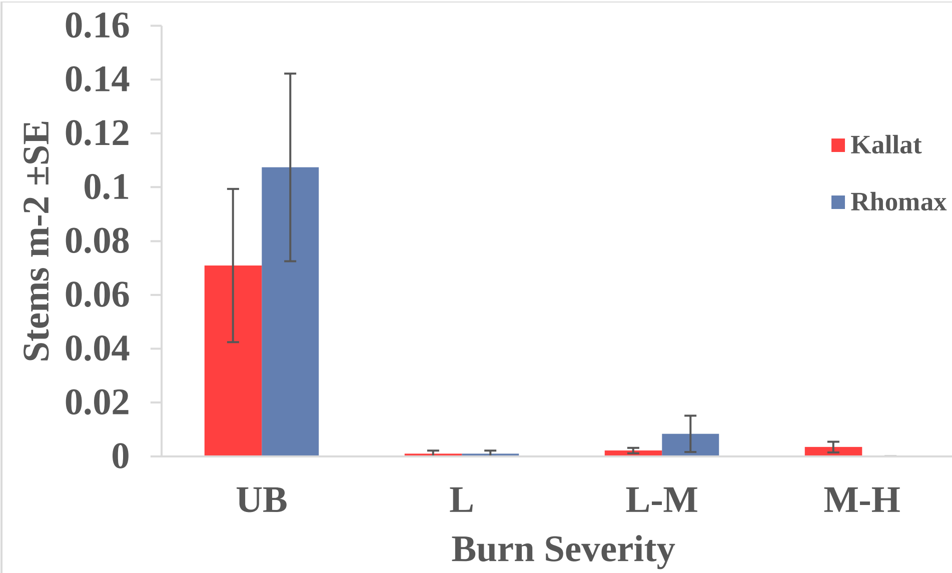

Figure 8: Average number of mountain laurel (Kallat) and rhododendron (Rhomax) in the sapling layer m^{-2} within each burn severity class. The error bars represent standard error.

There was a marked change in the density of both rhododendron and mountain laurel sprouts in the sapling layer in all burn severity classes (Fig. 8). Rhododendron stem density was significantly different among UB and all burn severity classes ($p < 0.0001$, Fig. 8). The only two classes that were not significantly different from one another were the L and L-M classes. Mountain laurel stem density was not different among burn severity classes ($p = 0.93$). In UB plots the average density of both these ericaceous shrubs was greatest. Rhododendron had an average density of 0.11±0.03 stems m^{-2} in UB plots, while mountain laurel had an average density of 0.07±0.03 stems m^{-2} (Fig. 8). In B areas the density of mountain laurel and rhododendron were low.

According to a Kruskal-Wallis test there was a significant difference in rhododendron seedlings density among burn severity classes ($p = 0.03$). However, according to the Dunn's test of multiple comparisons there was not a significant difference among individual burn severity classes. The 2016 wildfires increased mountain laurel seedling/sprout density. In contrast, we only documented rhododendron seedlings in UB plots (0.11±0.05 stems m^{-2}). Mountain laurel density did not vary among burn severity classes ($p = 0.66$). The average density of mountain laurel seedlings in UB plots was (0.21±0.10 stems m^{-2}). Mountain laurel was not recorded in L severity burned plots. However, mountain laurel densities were higher in L-M and M-H burn severity plots, having an average of 1.60±1.27 stems m^{-2} in L-M plots and 1.55±0.90 stems m^{-2} in M-H plots.

4.3 Seedlings:

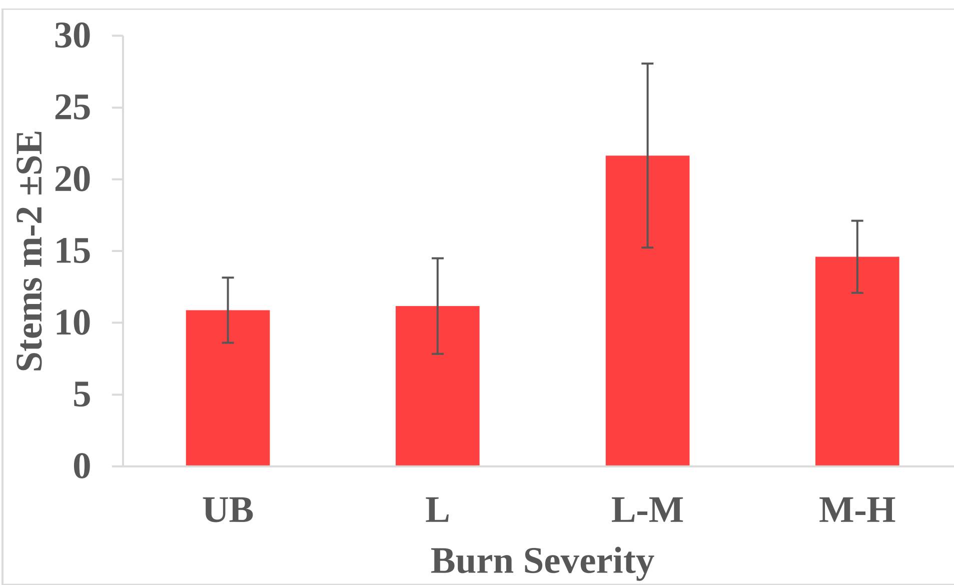

Figure 9: Average number of seedlings m^{-2} for all woody species within each burn severity class. The error bars represent standard error.

There was no difference in the density of seedlings among burn severity classes ($p = 0.39$). Seedling and basal sprout density included all tree species, shrubs, and woody vines under

0.5 m tall. The average seedling density in B watersheds was 17.08±2.88 stems m^{-2}, while UB averaged 10.88±2.27 stems m^{-2} (Fig. 9). In L burned plots seedling density was 11.17±3.33 stems m^{-2} and 21.65±6.41 stems m^{-2} in L-M severity plots (Fig. 9). The seedling density was 14.60±2.51 stems m^{-2} in M-H plots (Fig. 9).

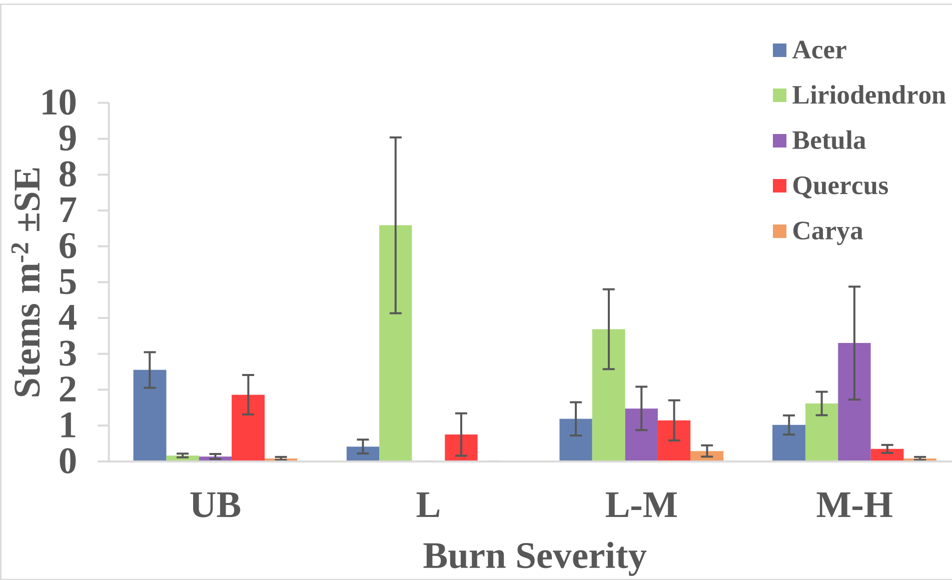

Figure 10: Average number of focal seedling species m^{-2} within each burn severity class. The error bars represent standard error.

Overall, we did not find a significant difference in seedling species richness among any of the burn severity classes (p= 0.61). Amongst the focal species in UB plots, maples and oaks occurred in the greatest densities (2.55±0.50 stems m^{-2} and 1.86±0.55 stems m^{-2}, Fig. 10). Although lower in all burn severity classes compared to UB, maple seedling densities varied with burn severity (L= 0.42±0.19 stems m^{-2}, L-M= 1.19±0.46 stems m^{-2}, M-H= 1.02±0.27 stems m^{-2}, Fig. 10). Tulip-*poplar* was recorded in low densities in UB plots (0.17±0.05 stems m^{-2}) and high densities in L plots (6.58±2.45 stems m^{-2}) (Fig. 10). Across burn severity classes, there was a decline in tulip-*poplar* seedling densities with increasing fire severity (L-M= 3.30±1.57 stems m^{-2}, M-H= 1.48±0.61 stems m^{-2}, Fig. 10). Birch was increasingly abundant in higher burn severity classes (L-M= 1.62±0.33 stems m^{-2}, M-H= 3.89±1.11 stems m^{-2}, Fig. 10). However, this species was only detected in low densities in UB plots (0.14±0.07 stems m^{-2}, Fig. 10) and was not present in L plots. Similar to maples, oak densities were lower in B plots compared to UB plots and varied by burn severity classes (L= 0.75±0.59 stems m^{-2}, L-M= 1.15±0.56 stems m^{-2}, M-H= 0.35±0.11 stems m^{-2}, Fig. 10). Hickory densities were low in all burn severity classes and was not present in L plots (UB= 0.08±0.04 stems m^{-2}, L-M= 0.29±0.16 stems m^{-2}, M-H= 0.08±0.04 stems m^{-2}, Fig. 10).

4.4 Groundlayer:

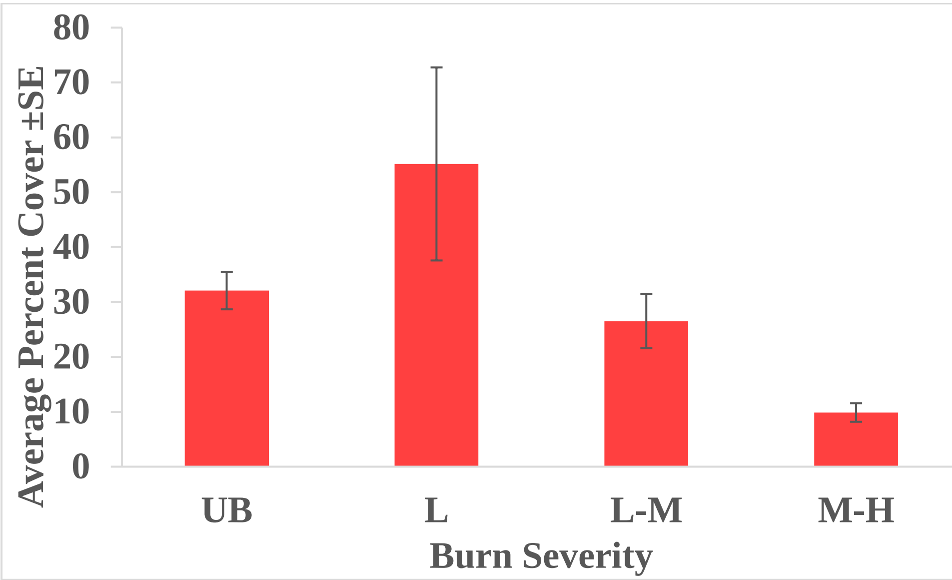

Figure 11: Average percent cover of the groundlayer m^{-2} within each burn severity class. The error bars represent standard error.

Our results indicate an effect of burn severity on the groundlayer density, which includes both woody seedlings under 0.5 m tall and all herbaceous species. We determined there was a significant difference in percent cover among UB and burn severity classes ($p<0.0001$). M-H plots had significantly less percent cover then UB and all other burn severity classes. Groundlayer cover in L burn severity plots (55±18%, Fig. 11) was the greatest of all burn severity classes, exceeding UB plots (32±3%, Fig. 11). There was a decline in average groundlayer percent cover as burn severity increased (L-M= 27±4%. M-H= 10±1%, Fig. 11).

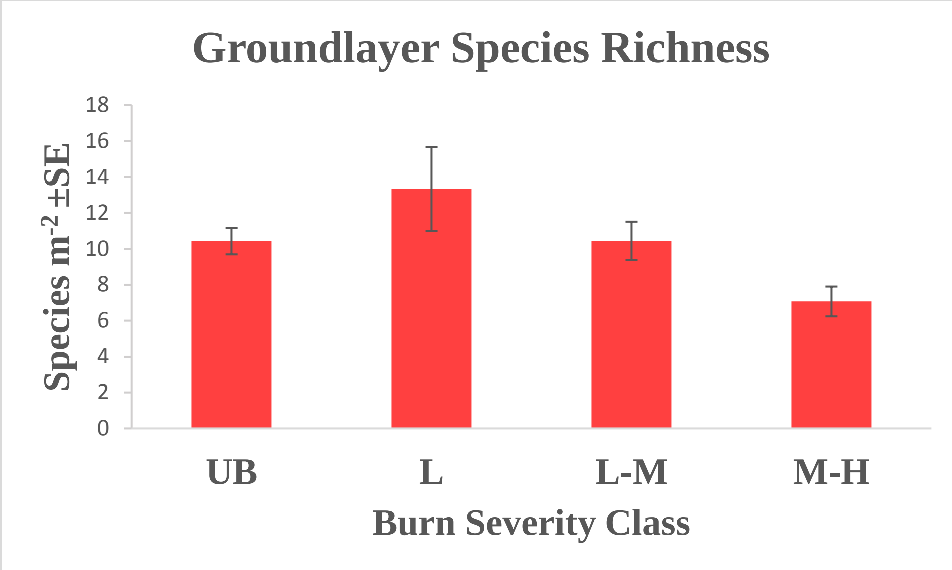

Figure 12: Average seedling species richness m^{-2} within each burn severity class. The error bars represent standard error.

The species richness of the groundlayer follows a similar trend as the percent cover of vegetation in this layer. The groundlayer species richness varied among UB and all burn severity classes ($p= 0.01$). M-H plots had significantly lower species richness compared to UB and all other burned severity classes. The richness of plants in UB areas was 10.43 species m^{-2}±0.74 (Fig.12). The highest species richness was in L severity B plots (13.33 species m^{-2}±2.33, Fig. 12). L-M burned plots were similar to UB plots in species richness (10.44 species m^{-2}±1.07, Fig. 12). M-H had the lowest species richness of all burn severity classes (7.07 species m^{-2}±0.83, Fig. 12).

5.0 Discussion:

The fires that affected the southern Appalachians in the fall of 2016 were of mixed severity with high severity patches (Katherine Elliott, USFS, personal communication).

Although these fires were unusual for the region, the effects may be indicative of future conditions if fires become more severe in the southern Appalachians. Future climate predictions indicate wildfires will be more frequent and severe in southeastern North America (Liu et al. 2013, Mitchell et al. 2014, Stanturf and Goodrick 2011). We examined the effects of two mixed-severity fire complexes to understand their effects on forest understories. Our analyses included the herbaceous ground layer, which provides important ecosystem functions and is highly diverse. We also examined the woody seedling and sapling layers which might indicate future changes in forest canopies.-Our results indicate low severity fires increase groundlayer cover and diversity, while higher severity fires initially decrease these indices. Mountain laurel and rhododendron decreased in density in the understory with fire of any severity. Finally, understory woody species did not increase in density or diversity following fire of any severity.

5.1 Saplings

Woody species richness was lower in B areas and decreased with increasing fire severity. This trend is likely the result of mortality of smaller tree species and species that are fire sensitive (Dey and Hartman 2005, Waldrop et al. 1992). Our results are contrary to Hagen and colleagues (2015), who reported woody species richness increased following fire. It’s possible that their burned sites did not include high severity patches. Furthermore, the differences in woody vegetation response may be a result of the short recovery time between the wildfires and our sampling. Hagen and colleagues (2015) sampled vegetation two to three years after fire. If the forest vegetation is given greater time to recover, a greater diversity of woody species may regenerate in B areas. Seedling recruitment of woody species is an indication of future recovery because these species may recruit to the understory and overstory, having a greater effect on regeneration. We did not identify differences in woody species composition among burn severities.

Sapling density varied by fire severity. Similar to Elliot and Vose (2010), we documented lower stem density in L and L-M burned plots compared to UB. Across plots, the abundance of most species was low, particularly in M-H severity plots. While species including maple and oaks might have produced basal sprouts, the lack of many species in the sapling layer of B plots may be impart due to a lack of basal sprouting. The sapling layer might require additional time for fire recovery. For example, Hagen and colleagues (2015) determined stems less than 5 cm DBH increased 2-3 years following a fire.

In the sapling layer, we did not find clear evidence of an increase in xeric species saplings relative to UB plots or relative to mesic species. In fact, we documented a trend of higher maple sapling density in L-M and M-H burned plots. These fire-sensitive species may have responded to fire with increased basal sprouting, regardless of severity (Abrams 1992, Brose et al. 2014, Elliott and Swank 2008, Nowacki and Abrams 2008, Nowacki and Abrams 2015). M-H severity fires likely damaged more trees leading to greater sprouting in these areas. In L-M plots, maple saplings were more abundant than oaks, yet densities were similar on M-H plots. This is somewhat consistent with Schwartz and colleague (2016), who did not find an increase in oak dominance following multiple prescribed fires in the Great Smokey Mountains National Park. In contrast, Hagen and colleagues (2015) observed fire had a positive effect on oaks. It is unclear whether the fires are measured to have altered the future forest species composition (Arthur et al. 2012). Other research indicates increasing oak dominance might in some cases require a combination of thinning and burning opposed to just burning, because this would open more overstory gaps and increase light to the groundlayer and understory (Brose et al. 2014). Oaks need more than 30 to 50% full sunlight to be recruited into the canopy (Arthur et al. 2012).

5.2 Seedling Layer

5.2.1 *Mesic Species*

There was a high level of tulip-*poplar* seedling regeneration in L plots, while birch seedlings were most in L-M and M-H plots. This regeneration is likely linked to seed dispersal from abundant overstory species. Both tulip-*poplar* and birch produce numerous seeds that are wind dispersed and could easily travel to burned areas (Burns and Honkala 1990). Others have observed increases in tulip-*poplar* seedling density immediately following cool fires that removed the leaf litter (Glasgow and Matlack 2007). Higher severity burned patches generally occurred on ridge tops where tulip-*poplar* are less abundant than in the rich coves where more of the lower severity plots were located. There were much fewer tulip-poplar seedlings in UB areas relative to any B areas indicating tulip-poplar, a shade-intolerant species, likely benefited from fire. Similarly, the increase in birch in L-M and M-H patches might be partially attributed to the abundance of birch at higher elevations near ridge tops. There were much fewer tulip-*poplar* seedlings in UB areas relative to any B areas. This indicates tulip-*poplar*, a shade-intolerant species, likely benefited from fire.

5.2.2 *Xeric species*

We did not find evidence that the mixed-severity fires increased the recruitment of xeric species seedlings. Hickory seedling densities were low across all burn severities, as well as in

UB areas, which might be due to the long-term result of mesophication and a decline of overstory hickories to produce seeds (Nowacki and Abrams 2008). Overall, oak seedling density decreased in B plots relative to UB plots in our study. Small seedlings often experience mortality following fires (Alexander et al. 2008). Fire-tolerant species including oaks do not develop fire resistance with a large root system and protected basal buds until they become larger. Further, acorns probably experience mortality in low intensity fires if located on leaf litter because the temperatures can reach 260 ºC, which causes mortality (Greenberg et al. 2012). Therefore, mortality might explain the relatively small amount of seedling recruitment in M-H severity plots where all litter and duff were consumed~~.~~ In contrast to our study, an increase in oak seedling density has been observed following higher severity fires in studies such as Elliott and colleagues (2009). Prescribed fire has also shown to increase white oak (*Quercus alba* L.) and hickory seedling densities in studies (Holzmueller et al. 2009). This difference in regeneration may have been a result of location and the overstory species composition or the fact this study focused on areas that were burned twice. Additionally, 15 to 22 years had elapsed since the last fire affected the forest in this study and the recovery recorded may have been delayed longer than the several months between the time the 2016 fires occurred and our vegetation sampling.

It is possible that in our study region, the forest is still recovering from fire effects. These fires occurred following an extreme drought, which likely increased wildfire risk (Case and Zavodsky 2018). The drought conditions that were present prior to the fire may have caused decline or mortality in mesophytic species such as maples and tulip-*poplar* independent of fire (Klos et al. 2009). Although the drought might have had an effect, it is possible that across our sites, mesophication has progressed to a level where this system has been irreversibly changed from historic conditions (Nowacki and Abrams 2008). If this is true, increasing oak and hickory abundance might require more active management intervention.

5.2.3 *Ericaceous Shrubs*

Our results indicate fire may be effective at removing both mountain laurel and rhododendron because we documented few seedlings and saplings of either mountain laurel or rhododendron in all burn severity classes relative to UB areas. Alternately, it is possible these shrubs were not common in areas that burned. Following low to moderate severity prescribed fire in the southern Appalachians, Elliott and Vose (2010) reported a decrease in the density of rhododendron. Hagen and colleagues (2015) only observed a decreased in mountain laurel densities following multiple fires. However, there was a significant increase in mountain laurel densities following a single fire. This increase was likely lower than that which would have occurred if this area was not burned. Many of the mountain laurel sprouts in our study were found in dense clonal groups. Extended monitoring will be needed to determine if these basal sprouts recruit to the understory and possibly result in increased density of these ericaceous shrubs.

5.3 Groundlayer

Prior to the implementation of fire suppression policies in the 20th century, Southern Appalachian forests are thought to have had an open overstory and understory layer with a high level of herbaceous density and species richness, maintained by regular fire (Harrod et al. 2000). Disturbances including fire provide conditions, including removal of leaf litter, increased light, and a pulse of nitrogen and other nutrients that can result in a growth response in the forest groundlayer (Elliott et al. 2011, Knoepp et al. 2009). In agreement with other studies, we documented an increase in the vegetation cover of the groundlayer of L patches (Arthur et al. 1998, Elliott et al. 1999, Elliott and Vose 2010, Hagan et al. 2015). For example, Hagen and colleagues (2015) examined the effects of once and twice burned areas in forests located in the southern Appalachian Mountains and found that herbaceous species cover increased for both burn frequencies. Similarly, Schwartz and colleagues (2016) determined an increase in herbaceous vegetation following prescribed fire. In our study many herbaceous plants likely were able to resprout from a living rhizome following a low severity fire. Factors such as whole plant mortality, seed destruction, and lower levels of ant facilitated seed dispersal may have caused a decrease in herbaceous cover of higher burn severity classes (Elliott et al. 2014, Greenberg et al. 2012, Mitchell et al. 2002).

In contrast to the L patches, as fire severity increased, groundlayer cover decreased. L-M plots had a lower groundlayer cover compared to UB plots, and M-H patches had the lowest groundlayer cover of all treatments. This decrease in cover in areas affected by higher severity fire is likely caused by a number of factors. We sampled vegetation in the growing season immediately following fires. The herbaceous vegetation may not have had time to respond in the short time in between the fires and our initial sampling. The seeds of some plants that were stored in the seedbank may have also experienced mortality. Seeds begin to experience mortality in high temperatures created by fire (Greenberg et al. 2012). Seedbank survival during high severity fire has not been studied for many of the herbaceous plants found in the southern Appalachians and is a topic in need of further research.

Similar to percent cover, L plots had greater groundlayer species richness than UB plots, which was likely caused by the fires clearing the understory of leaf litter and stimulating herbaceous growth through increased nitrogen (Elliott et al. 2011, Knoepp et al. 2009). Additionally, these low severity fires likely created more open conditions that suit the light requirements of a greater diversity of plants compared to UB areas (Elliott et al. 2011). Similar to our results, Elliot and Vose (2010) documented an increase or no change in the herbaceous species diversity after low to moderate severity prescribed fires. Additionally, Hagen and colleagues (2015) showed single and multiple natural wildfires in the mountains of North Carolina had a positive influence on herbaceous species richness. Following single and multiple prescribed fires in the Great Smokey Mountain National Park, Holzmueller and colleagues (2009) also found an increase in herbaceous species richness.

We documented a decline in groundlayer species richness with increasing burn severity, which is likely driven by factors including seed mortality (Greenberg et al. 2012). In contrast to our results, Elliott and colleagues (1999) found an increase in diversity in areas affected by moderate-severity prescribed fire one and two years following fire. The increase in herbaceous vegetation was attributed to an increase in resources such as light and nutrients following the fire. Although we did not find similar results, our research sites may reflect those of Elliott and colleagues (1999) given greater recovery time. It is possible we did not observe an increase in herbaceous diversity in higher severity B plots because the fires occurred during drought, because plant community recovery is delayed due to the late season fires, or because the fires were more severe than prescribed fires or because there were more xeric sites with less herbaceous cover.

The 2016 wildfires created patches of open forest conditions where higher severity fires occurred. These conditions may limit some forms of dispersal, while benefiting others. For example, many forest herbs rely on ants for dispersal (Elliott et al. 2014, Mitchell et al. 2002).

Aphaenogaster spp. is an ant species that is important for dispersal of forest herbaceous plants in this region. This ant species prefers mesic conditions. The distance from the interior of burned areas to suitable habitat and seeds sources may be an additional barrier for dispersal, because ants only disperse seeds several meters. Many of the areas that experienced L-M and M-H severity fires are much larger than the distance ants move seeds over the short time between the wildfires and our first sampling.

5.4 Limitations

The wildfires we are studying occurred following exceptional drought in the fall of 2016 (M. Wilkins, District Ranger, Nantahala Ranger District, personal communication). We were unable to sample vegetation prior to the fires because of the unpredictable nature of wildfires. This confounds the results from our B and UB areas, as well as among burn severities. To gain an understanding of the fires, we sampled watersheds that were geographically close to one another and affected by the same wildfire. Other confounding factors include the slope and aspect of areas where fires occurred. Higher severity fires may have been more prevalent on ridges. Additionally, fewer mesic species may grow in these areas and may be represented to less of a degree compared to xeric species. However, the system we used to determine burn severity was based on how the environment was affected by fire opposed to the type of vegetation growing in an area. This method of determining fire severity likely overcomes the tautology, that may have occurred if burn severity was based off the type of vegetation present.

5.5 Conclusion

These initial results of mixed severity fires give an indication that fire severity is important for the type of vegetation recovery following a fire. Our results indicate the herbaceous vegetation seems to respond positively to mixed-severity fires. In the first year following the fires, we did not find evidence that mixed-severity fires provided woody species that were historically important a competitive advantage. However, amongst B plots we found the highest abundance of oak sapling regeneration in M-H severity B plots. Further, these fires removed ericaceous plants such as mountain laurel and rhododendron in the forest understory. It will be important to monitor these areas as they recover over the next several years to determine if these trends persist or there is a lag effect that results in changes in species composition.

LITERATURE CITED

Abrams, M. D. 1998. The red maple paradox. Bioscience 48:355-364.

Abrams, M. D. 1992. Fire and the Development of Oak Forests. Bioscience 42:346-353.

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.

Arthur, M. A., R. D. Paratley, and B. A. Blankenship. 1998. Single and repeated fires affect survival and regeneration of woody and herbaceous species in an oak-pine forest. Journal of the Torrey Botanical Society :225-236.

Arthur, M. A., H. D. Alexander, D. C. Dey, C. J. Schweitzer, and D. L. Loftis. 2012. Refining the oak-fire hypothesis for management of oak -dominated forests of the eastern United States. Journal of Forestry 110:257-266.

Brose, P., D. Dey, and T. Waldrop. 2014. The fire—oak literature of eastern North America: synthesis and guidelines. Gen.Tech.Rep.NRS-135.Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station.98 p. 135:1-98.

Burns, R. M., and B. H. Honkala. 1990. Silvics of North America, Vol. 2–Hardwoods, Agricultural Handbook 654. US Department of Agriculture, Washington, DC :.

Case, J. L., and B. T. Zavodsky. 2018. Evolution of 2016 drought in the Southeastern United States from a Land surface modeling perspective. Results in physics 8:654-656.

Chiang, J., R. W. McEwan, D. A. Yaussy, and K. J. Brown. 2008. The effects of prescribed fire and silvicultural thinning on the aboveground carbon stocks and net primary production of overstory trees in an oak-hickory ecosystem in southern Ohio. Forest Ecology and Management 255:1584-1594.

Dale, V. H., L. A. Joyce, S. McNulty, R. P. Neilson, M. P. Ayres, M. D. Flannigan, P. J. Hanson, L. C. Irland, A. E. Lugo, and C. J. Peterson. 2001. Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. Bioscience 51:723-734.

Day, F. P., D. L. Phillips, and C. D. Monk. 1988. Forest communities and patterns. Pages 141-149 *In* Anonymous Forest hydrology and ecology at Coweeta, Springer, .

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.

Edwards, S. L., S. Demarais, B. Watkins, and B. K. Strickland. 2004. White-tailed deer forage production in managed and unmanaged pine stands and summer food plots in Mississippi. Wildlife Society Bulletin 32:739-745.

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

Elliott, K. J., and W. T. Swank. 2008. Long-term changes in forest composition and diversity following early logging (1919–1923) and the decline of American chestnut (Castanea dentata). Plant Ecology 197:155-172.

Elliott, K. J., J. M. Vose, and D. Rankin. 2014. Herbaceous species composition and richness of mesophytic cove forests in the southern Appalachians: synthesis and knowledge gaps. The Journal of the Torrey Botanical Society :39-71.

Elliott, K. J., C. A. Harper, and B. Collins. 2011. Herbaceous response to type and severity of disturbance. Pages 97-119 *In* Anonymous Sustaining Young Forest Communities, Springer, .

Elliott, K. J., J. M. Vose, and R. L. Hendrick. 2009. Long-term effects of high intensity prescribed fire on vegetation dynamics in the Wine Spring Creek Watershed, western North Carolina, USA. :.

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.

Flatley, W. T., C. W. Lafon, H. D. Grissino-Mayer, and L. B. LaForest. 2013. Fire history, related to climate and land use in three southern Appalachian landscapes in the eastern United States. Ecological Applications 23:1250-1266.

Ford, W. M., R. H. Odom, P. E. Hale, and B. R. Chapman. 2000. Stand-age, stand characteristics, and landform effects on understory herbaceous communities in southern Appalachian cove-hardwoods. Biological Conservation 93:237-246.

George, L. O., and F. A. Bazzaz. 2003. The herbaceous layer as a filter determining spatial pattern in forest tree regeneration. The herbaceous layer in forests of eastern North America :265-282.

Gilliam, F. S. 2014. The herbaceous layer in forests of eastern North America. Oxford University Press, .

Gilliam, F. S. 2007. The Ecological Significance of the Herbaceous Layer in Temperate Forest Ecosystems. BioScience 57:845-858.

Glasgow, L. S., and G. R. Matlack. 2007. Prescribed burning and understory composition in a temperate deciduous forest, Ohio, USA. Forest Ecology and Management 238:54-64.

Greenberg, C. H., T. L. Keyser, S. J. Zarnoch, K. Connor, D. M. Simon, and G. S. Warburton. 2012. Acorn viability following prescribed fire in upland hardwood forests. Forest Ecology and Management 275:79-86.

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

Harrod, J. C., M. E. Harmon, and P. S. White. 2000. Post-fire succession and 20th century reduction in fire frequency on xeric southern Appalachian sites. Journal of Vegetation Science 11:465-472.

Holzmueller, E. J., S. Jose, and M. A. Jenkins. 2009. The response of understory species composition, diversity, and seedling regeneration to repeated burning in southern Appalachian oak-hickory forests. Natural Areas Journal 29:255-262.

Klos, R. J., G. G. Wang, W. L. Bauerle, and J. R. Rieck. 2009. Drought impact on forest growth and mortality in the southeast USA: an analysis using Forest Health and Monitoring data. Ecological Applications 19:699-708.

Knoepp, J. D., K. J. Elliott, B. D. Clinton, and J. M. Vose. 2009. Effects of prescribed fire in mixed oak forests of the southern Appalachians: forest floor, soil, and soil solution nitrogen responses. The Journal of the Torrey Botanical Society :380-391.

Liu, Y., S. Goodrick, and J. Stanturf. 2013. Future U.S. wildfire potential trends projected using a dynamically downscaled climate change scenario. Forest Ecology and Management 294:120-135.

McEwan, R. W., J. M. Dyer, and N. Pederson. 2011. Multiple interacting ecosystem drivers: toward an encompassing hypothesis of oak forest dynamics across eastern North America. Ecography 34:244-256.

Mitchell, C. E., M. G. Turner, and S. M. Pearson. 2002. Effects of historical land use and forest patch size on myrmecochores and ant communities. Ecological Applications 12:1364-1377.

Mitchell, R. J., Y. Liu, J. J. O'Brien, K. J. Elliott, G. Starr, C. F. Miniat, and J. K. Hiers. 2014. Future climate and fire interactions in the southeastern region of the United States. Forest Ecology and Management 327:316-326.

Muller, R. N. 2003. Nutrient relations of the herbaceous layer in deciduous forest ecosystems. The herbaceous layer in forests of eastern North America.Oxford University Press, New York :15-37.

Nilsen, E. T., B. D. Clinton, T. T. Lei, O. K. Miller, S. W. Semones, and J. F. Walker. 2001. Does Rhododendron maximum L.(Ericaceae) reduce the availability of resources above and belowground for canopy tree seedlings? The American Midland Naturalist 145:325-343.

Nowacki, G. J., and M. D. Abrams. 2015. Is climate an important driver of post-European vegetation change in the Eastern United States? Global Change Biology 21:314-334.

Nowacki, G. J., and M. D. Abrams. 2008. The demise of fire and “mesophication” of forests in the eastern United States. Bioscience 58:123-138.

AnonymousFire and fire surrogate treatments in mixed-oak forests: Effects on herbaceous layer vegetation. In: Butler, Bret W.; Cook, Wayne, comps. The fire environment--innovations, management, and policy; conference proceedings. 26-30 March 2007; Destin, FL. Proceedings RMRS-P-46CD. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. CD-ROM. p. 475-485; 2007.

Pyne, S. J. 1984. Introduction to wildland fire. Fire management in the United States. John Wiley & Sons, .

R Core Team. 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing :.

Roman, D. T., K. A. Novick, E. R. Brzostek, D. Dragoni, F. Rahman, and R. P. Phillips. 2015. The role of isohydric and anisohydric species in determining ecosystem-scale response to severe drought. Oecologia 179:641-654.

Schafale, M. P. 2012. Guide to the Natural Communities of North Carolina Fourth Approximation. NC Department of Natural Resources, Natural Heritage Program :.

Schwartz, N. B., D. L. Urban, P. S. White, A. Moody, and R. N. Klein. 2016. Vegetation dynamics vary across topographic and fire severity gradients following prescribed burning in Great Smoky Mountains National Park. Forest Ecology and Management 365:1-11.

Simon, S. A., T. K. Collins, G. L. Kauffman, W. H. McNab, and C. J. Ulrey. 2005. Ecological zones in the Southern Appalachians: First approximation. US Department of Agriculture, Forest Service, Southern Research Station, .

Stambaugh, M. C., R. P. Guyette, J. M. Marschall, and D. C. Dey. 2016. Scale dependence of oak woodland historical fire intervals: Contrasting the Barrens of Tennessee and Cross Timbers of Oklahoma, USA. Fire Ecology.12 (2): 65-84. 12:65-84.

Stanturf, J. A., and S. L. Goodrick. 2011. Chapter 17. Fire. Southern forest futures project.Edited by DW Wear and JG Greis.USDA Forest Service Southern Region, USDA Forest Service Southern Research Station, Southern Group of State Foresters.Available from http://www.srs.fs.usda.gov/futures/[accessed 23 August 2011] :.

Svoboda, M., D. LeComte, M. Hayes, R. Heim, K. Gleason, J. Angel, B. Rippey, R. Tinker, M. Palecki, and D. Stooksbury. 2002. The drought monitor. Bulletin of the American Meteorological Society 83:1181-1190.

Vose, J. M., and K. J. Elliott. 2016. Oak, fire, and global change in the eastern USA: what might the future hold? Fire Ecology 12:.

Waldrop, T. A., D. L. White, and S. M. Jones. 1992. Fire regimes for pine-grassland communities in the southeastern United States. Forest Ecology and Management 47:195-210.

Wurzburger, N., and R. L. Hendrick. 2007. Rhododendron thickets alter N cycling and soil extracellular enzyme activities in southern Appalachian hardwood forests. Pedobiologia 50:563-576.