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THE ROLE OF FIRE AND A FIRE-FREE INTERVAL IN THE RESTORATION OF UPLAND OAK COMMUNITIES ON THE CUMBERLAND PLATEAU, KENTUCKY

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THESIS

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

By

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ABSTRACT OF THESIS

THE ROLE OF FIRE AND A FIRE-FREE INTERVAL IN THE RESTORATION OF UPLAND OAK COMMUNITIES ON THE CUMBERLAND PLATEAU, KENTUCKY

The decline of upland oak (*Quercus* spp.) communities in our eastern forests has been attributed to the loss of periodic disturbance after decades of fire suppression. As land managers have begun to reintroduce fire, effects on oak regeneration and species composition have varied widely, making it apparent that our understanding of how fire can aid in oak forest management needs refinement. Restoring upland oak communities requires decreasing stand density and opening of the canopy to release moderately shade-intolerant oaks in the understory. This necessitates an extended fire-free interval to allow these oaks to be recruited into larger size classes and develop resistance to future fires. The ability of prescribed fire alone to create these structural changes is uncertain due to the low intensity of prescribed burns which for the most part do not kill larger diameter trees. In this work, I examined the utility of a fire-free interval following repeated fire alone as a management tool, as well as the combined effects of fire and mechanical removal in the form of midstory mastication. Where forest structure is significantly reduced by fire or mechanical removal, restoration of oak communities is complicated by both prolific sprouting and ingrowth of competitor species and the introduction of invasive species. The results of this study suggest that, in the absence of mechanical removals, reductions in stem density necessary to restore conditions for oak regeneration might be limited to sites that experience higher fire severity and/or drier landscape positions. Additionally, the rapid response of competing non-oak stems such as maple (*Acer* spp.), yellow-poplar (*Liriodendron tulipifera*) and sassafras (*Sassafras albidum*) during the fire-free interval and the increasingly severe invasion of Japanese stiltgrass (*Microstegium vimineum*) following disturbance are serious hindrances to successful restoration of upland oak ecosystems. Despite these management concerns, results of the research reported in this thesis indicate that restoring disturbance regimes slows the process of mesophication, improves size and stature of oak regeneration, and increases community diversity across the landscape.

KEYWORDS: Fire-free interval, prescribed fire, *Quercus* spp., mesophication, *Microstegium vimineum*, disturbance

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CHAPTER 1 THE ROLE OF FIRE DISTURBANCE IN THE MAINTENANCE OF OAK-DOMINATED ECOSYSTEMS

The Central Hardwood Forest Region, of which the Cumberland Plateau is part, is the largest contiguous, temperate hardwood forest in the world (Smith, 1995). The physiography, highly variable topography, and history of anthropogenic and natural disturbances characteristic of the region have created a complex and heterogeneous landscape. Upland oak (*Quercus*) forests are an important element of this landscape and contribute to the diversity of both species composition and habitat structure. In particular, the genus *Quercus* is of tremendous value to both humans and wildlife. The nutrient rich mast of oak species is a fundamental food source for many bird and mammal species in eastern hardwood systems (Martin et al., 1961; McShea et al., 2007). Oaks provide habitat for a variety of invertebrates, which in turn support a diverse population of birds, reptiles, amphibians and small mammals (Rodewald & Abrams, 2002; Valencia-Cuevas & Tovar-Sánchez, 2015). The large size and longevity of oak species provides stable nesting habitat, and decay-resistant snags create valuable nesting for cavity-dwelling species (Rodewald & Abrams, 2002; Runde & Capen, 1987). Avian communities in Pennsylvania demonstrated a marked preference for oak habitat suggesting that the loss of mature oak systems in eastern forests would result in reduced species richness and abundance of native birds (Rodewald & Abrams, 2002).

Upland oak forests have been on the landscape since the last glacial maximum and evidence suggests that periodic fire likely facilitated their development and persistence across multiple temporal and spatial scales. Several characteristics of oak species imply fire adaptation, including hypogeal germination, moderately high light requirements, development of thick bark with age, ability to compartmentalize wounds effectively, and the suitability of fire-created seedbeds for acorn germination (Elliott et al., 2004; McEwan et al., 2011). In addition, oak species have dormant buds near the root collar, which is often buried in the soil and protected from the heat of fire. Oak seedlings preferentially allocate carbohydrates into root growth developing more biomass belowground, in root systems, than in aboveground shoot growth. This allows oak species to re-capture growing space more quickly from fire induced-top-kill than other mesic competitors (Dey & Fan, 2009; Dey & Schweitzer, 2015; McEwan et al., 2011). In total, these physical and physiological adaptations are thought to allow oak species to survive periodic fire more readily than more fire-sensitive and mesic species.

Paleoecology studies in the Central Hardwood Forest Region provide evidence that Indigenous Peoples were using and managing forests for thousands of years, including with fire. Beginning in the late Holocene (ca. 3,000 BP), changes in local climate, increasing Indigenous populations, and a cultural shift towards more permanent settlements and agriculture coincided with an increase in charcoal accumulated in pond and soil sediment. This increase in charcoal indicates frequent surface fires which are thought to have encouraged the dominance of fire-adapted species, including oak and pine (*Pinus* spp.), on ridges and upper slopes. (Fowler & Konopik, 2007; Watts, 1980; Delcourt et al., 1998). Fires were likely periodic, low-intensity surface fires and most frequently ignited in the spring or fall. Efforts to determine historic and pre-historic fire return intervals in eastern forests have yielded highly variable estimates influenced by human population levels, changes in culture, and drought years. Fire scar studies across the Central Hardwood Forest Region found mean fire return intervals that ranged from three to 19 years pre-European settlement (1650 – 1850) (Guyette et al., 2006).

While early European settlers initially continued the Indigenous Peoples burning tradition, unsustainable industrialized logging in the late 19th and early 20th centuries led to large increases in woody surface fuels in the Central Hardwood Forest Region. The accumulation of highly flammable slash and logging refuse contributed to catastrophic, stand-replacing fires in the mid-1900s, which prompted an era of active fire suppression at the federal level that lasted until the late 1900s (Abrams, 1996; Abrams, 1992; Brose et al., 2001). During this same period, the loss of the American chestnut (*Castanea dentata*) from an introduced blight fungus (*Cryphonectria parasitica*) initiated widespread changes in forest composition where chestnut was once dominant. Additionally, the loss of the passenger pigeon (*Extopistes migratorius*) from over hunting removed a potentially important disturbance event as their large flocks created substantial canopy damage and gaps that could have benefited oak reproduction. Overhunting of white-tailed deer (*Odocoileus virginianus*) also led to drastic population declines in the late 19th century; however, populations rebounded during the 20th century to above pre-settlement levels and increased browse pressure preferentially on oak species. Lastly, analysis of multi-decadal drought frequencies over the past 500 years show that the most recent century is marked by a notable increase in moisture availability (McEwan et al., 2011). However, whether the increase in precipitation seen in recent decades played an important role in compositional change in eastern forests has been challenged in recent years (Hanberry et al., 2018). The removal and alteration of several forms of disturbance, extinction of keystone species and changing climate have each likely contributed to a compositional shift away from oaks in our contemporary mixed hardwood forests.

Despite the challenges of disentangling the combined influences listed above, it has been hypothesized that fire suppression played a central role in initiating a widespread shift in species composition away from oaks and towards more mesic and shade-tolerant species. Decades without fire on the landscape allowed mesic and otherwise competitive species such as maples, sourwood (*Oxydendrum arboreum*), black gum (*Nyssa sylvatica*), American beech (*Fagus grandifolia),* and sweet birch (*Betula lenta*) to establish in the understory (Abrams & Downs, 1990; Blankenship & Arthur, 2006; Brose et al., 2007; Nowacki & Abrams, 2008). This species shift has been referred to as mesophication, a process whereby mesic and shade-tolerant species increasingly dominate a landscape by creating an environment that favors their growing conditions while preventing the establishment of more xeric, shade-intolerant species (Nowacki & Abrams, 2008). Red maple (*Acer rubrum*) is most often cited as a driver of mesophication and has become a dominant component across a wide gradient of habitat types throughout much of the eastern US (Abrams, 1998).

A direct consequence of mesophication is a reduction in light in the understory. Oak species are moderately shade-intolerant and do not compete well in the closed-canopy environment of maple-dominated forests. This provides an advantage for shade-tolerant species to establish while reducing the ability of oaks to regenerate in the understory (Brown & Parker, 1994; Crow, 1992; Dillaway et al., 2007). Litter additions from oak versus non-oak species also lead to important physical and chemical changes in the understory (Babl et al., 2020; Alexander & Arthur, 2014). Maple leaves have a higher packing ratio than oak leaves; as they accumulate on the forest floor, air pockets are minimized, moisture is trapped, and flammability is reduced. Oak leaves, on the other hand, have a very low packing ratio and maintain a curl when dry, trap air pockets and provide a more flammable surface fuel (Kreye et al., 2014) Additionally, species differences in bole texture and canopy architecture can affect the local hydrology of the forest (Alexander & Arthur 2010). Denser, deeper canopies of mesic species like maple and American beech increase shade and create a cooler and moister microclimate in the understory (Alexander & Arthur, 2010; Babl et al., 2020). The canopy and bole structure of maples also leads to increased stem flow, further increasing moisture levels at the base of maple trees (Alexander & Arthur 2010). In total, these physical and chemical changes to the local environment constitute a positive feedback loop that supports continued maple dominance and prevents subsequent surface fire, discouraging the long-term presence of oaks on the site (Hanberry et al., 2012). The combination of these environmental changes and altered disturbance regimes has led to a widespread failure of oaks regenerating in eastern forests (Nowacki & Abrams, 2008).

The culmination of current knowledge on this relationship is encapsulated by the oak-fire hypothesis, which states that (1) periodic fire has been an integral disturbance in the mixed-oak forests of eastern North America for millennia; (2) oaks have several physical and physiological characteristics that should allow them to survive at higher rates than their competitors in a periodic fire regime; (3) the lack of fire in the latter 20th century is a major reason for the current widespread oak regeneration problem; and (4) reintroducing fire via prescribed burning could open the canopy through mortality of smaller midstory stems and promote oak regeneration (Arthur et al., 2012; Brose & Van Lear, 2004; Brose et al., 2012).

Based on this understanding of the role of fire in the development of oak-dominated forests, land managers are increasingly using prescribed fire to maintain and restore oak communities. However, several ecologists have called for more careful consideration of how fire can meet our specific land management goals (Arthur et al., 2012). Results of studies across eastern forests have produced mixed results indicating that the relationship between fire and oak regeneration is not as simple as once thought (Arthur et al., 2015; Keyser et al., 2017; Brose et al., 2013). Specifically, there is a growing consensus that prescribed fire alone might be unable to reverse mesophication and promote oak reproduction on all but the most xeric of sites (Hutchinson et al., 2012; Franklin et al., 2003; Blankenship & Arthur, 2006). Low intensity prescribed fire characteristic of fire management in the Central Hardwood Forest Region does not reliably kill large diameter stems, precluding necessary increases in light to the understory. Combined use of partial timber harvests (thinning, shelterwood, group selection) and prescribed burning have shown promising results where financially and logistically possible (Hutchinson et al., 2005; Dey et al., 2010). Where such combined treatments are untenable, canopy gap formations from secondary natural disturbances (wind throw, ice storms, droughts) could similarly provide the structural change needed to promote upland oak communities (Hutchinson et al., 2005).

Despite the lack of clarity regarding the effectiveness and specific elements of management using prescribed fire (e.g., fire severity, frequency, seasonality, canopy gap formation), in the absence of active management, widespread homogenization and ongoing mesophication will continue. The loss of upland oak communities will undoubtedly lead to substantive losses in species and habitat diversity across the heterogeneous landscape of eastern forests. Further, as flammability of these forests continues to decline with increasing dominance of mesic species in the understory, the window of time that fire can be effectively used as a management tool is dwindling. For this reason, it is of upmost importance that we continue to hone and refine management approaches to restore upland oak systems and retain landscape heterogeneity. The University of Kentucky Forest Ecology laboratory, the USDA Forest Service Bent Creek Experimental Forest, and the Daniel Boone National Forest have conducted collaborative research in eastern Kentucky for more than two decades aimed at developing a better understanding of how prescribed fire can aid oak restoration and the maintenance of community diversity. Numerous publications resulting from this collaboration investigate the function of repeated fire in altering stand structure and promoting oak seedling regeneration (Arthur et al., 2015; Keyser et al., 2017; Royse et al., 2010; Green et al., 2010; Alexander et al., 2008; Blankenship & Arthur, 2006); the role of mesophication in altering flammability, forest hydrology, and chemistry (Alexander & Arthur, 2010; Alexander & Arthur, 2014); the role of wildfire in promoting oak communities and preserving landscape heterogeneity (Black et al., 2018); the impact of prescribed burning on fuel beds and light environment (Arthur et al., 2017; Loucks et al., 2008; Chiang et al., 2005); and mechanical removals for oak woodland restoration (Black et al., 2019), among many others not listed here.

This thesis represents the continued efforts of this collaboration, and specifically investigated the role of a fire-free interval in a long-term repeated fire study (Chapter 2), and the application of prescribed fire following midstory mastication in an oak woodland restoration project (Chapter 3). The results of these studies will further our understanding of the impacts of reintroducing disturbance in a highly variable landscape with the goal of preserving our valuable oak forest resource and conserving diversity of species, habitat, and community type.

CHAPTER 2 THE ROLE OF A FIRE-FREE INTERVAL IN MANAGING UPLAND OAK FORESTS ON THE CUMBERLAND PLATEAU, KENTUCKY.

ABSTRACT

After decades of fire suppression, oaks are failing to recruit into larger size classes in the understories of eastern forests. Land managers have grappled with this challenge and the realization that restoring upland oak communities requires decreasing stem density and opening of the canopy to release the moderately shade-intolerant oaks in the understory. Repeated prescribed fire is one of the tools that managers are turning to, followed by an extended fire-free interval to allow time for oak seedlings to recruit into larger size classes and develop resistance to future fires. Previous results from this study area showed that repeated prescribed fire significantly reduced sapling (2-10 cm DBH) and midstory (10-20 cm DBH) stem density and basal area. With this study, I explored the capacity for oaks in the understory to be recruited into the sapling layer during a fire-free interval. Recruitment of oak saplings was found only in burn treatments and only in intermediate and sub-xeric landscape positions. Basal area was the primary driver for oak recruitment on these plots, with no oak saplings found in plots with basal area greater than 20 m^2ha^{-1}. This suggests that in the absence of mechanical thinning and harvest, higher severity fires might be needed to kill larger diameter trees and create more open-canopy forests. The dominance of maple (*Acer*) in the midstory and sapling layers was reduced by repeated fire on drier landscape positions, thereby slowing the process of mesophication. This is especially noteworthy due to the continuing increase in dominance of maple species in the Fire Excluded treatment, indicating a divergence of species composition between burned and unburned forests. Despite these trends, as of 2018, non-oak species continued to dominate the sapling and midstory layers in all treatments. Prolific sprouting and ingrowth of maple species and competing shade-intolerant species such as yellow-poplar (*Liriodendron tulipifera*) and sassafras (*Sassafras albidum*) during the fire-free interval are hindrances to successful restoration of upland oak ecosystems with repeated prescribed fire.

INTRODUCTION

Upland oak ecosystems across much of the eastern United States are experiencing oak recruitment failure in part due to decades of fire suppression. Most upland oak species in eastern forests have low to intermediate shade-tolerance and require periodic disturbance to increase light levels at the forest floor to develop (Abrams, 1992). In addition to moderately high light requirements, several other attributes of oak species imply fire adaptation including hypogeal germination, development of thick bark with age, ability to compartmentalize wounds effectively, and the suitability of fire-mediated seedbeds for acorn germination (Abrams, 1992 ; Elliott et al., 2004).

Oak species have been a dominant component of eastern forests since the late Holocene (ca 3,000 BP). Paleoecological studies show that the increase in oak dominance on the landscape coincides with an increase in charcoal accumulation found in pond sediments and soil, indicating recurring surface fires (Fowler & Konopik, 2007; Delcourt & Delcourt, 1998; Delcourt et al., 1998; Watts, 1980). These periodic fires acted as an ecological filter for thinner barked, shade-tolerant species like maples (*Acer* spp.*)* and American beech (*Fagus grandifolia*) and created more open structured forests dominated by oak (Abrams, 1992; Brose et al., 2006). This was especially true on drier and nutrient poor sites where oak species were more competetive and where fire more readily occurred (Dey & Fan, 2009). Fire suppression, beginning in the mid 1900s, allowed natural succession to progress. This resulted in canopy closure, stand densification, and shaded understories that promoted a compositional shift towards shade-tolerant and mesophytic species (Abrams & Nowacki, 2008). This process has been termed "mesophication" and over the decades where fire has been absent on the landscape has created a positive feedback loop where mesic species increasingly dominate by creating an environment with growing condition that favor mesic species while limiting the establishment of more xeric, shade-intolerant species (Abrams & Nowacki, 2008).

This understanding of the ecology of fire and oak has increasingly led land managers to use fire as a management tool to restore oak communities in eastern deciduous forests (Arthur et al., 2012). The ultimate goal of returning periodic fire to the landscape is to reverse the process of mesophication by restoring the more open structure of presettlement oak forests that allowed oaks in the understory to be recruited into the canopy (Nowacki & Abrams, 2008; Brose et al., 2006; Brose et al., 2013). The general consensus is that a single prescribed fire is insufficient to alter forest structure or to set back succession (Knapp et al., 2015). Repeated prescribed fires have been more effective in reducing stand density and promoting oak, but have not been shown to be entirely successful in reducing the competitive status of non-oak species (Blankenship & Arthur, 2006; Hutchinson et al., 2012a). Prolific sprouting following top-kill in many species like maples and sassafras (*Sassafras albidum*) can rapidly outcompete oak stems in the understory and increase overall stem density (Hutchinson et al., 2005, Henderson et al., 1984). However, some evidence suggests that increasing numbers of prescribed fires reduces this sprouting response over time (Arthur et al., 2015). Additionally, it has been suggested that prescribed fire alone might be insufficient to restore open structure on all but the driest of sites (Hutchinson et al., 2012b; Franklin et al., 2003; Blankenship & Arthur, 2006) because prescribed fires characteristic of the Central Hardwood Forest Region are unable to reliably kill large diameter stems. This has prompted several studies examining the combined use of partial removals (thinning, shelterwood, group selection) with prescribed burning and has shown promising results (Hutchinson et al., 2005; Dey et al., 2010). Where such combined treatments are not financially or logistically possible, canopy gap formations from secondary natural disturbances (wind throw, ice storms, droughts) could similarly provide the structural change needed to promote upland oak communities (Hutchinson et al., 2005).

Where the creation of more open structure is achieved, a period without fire is needed to allow oaks in the understory to grow large enough to survive subsequent fires (Arthur et al., 2012; Fan et al., 2012; Knapp et al., 2017). Prescribed fires will typically cause dieback or top-kill hardwood stems less than 13 centimeters in diameter (Dey & Fan, 2009; Knapp et al., 2015). Faster-growing oak species on high quality sites with adequate light conditions might be able to reach a diameter and bark thickness that would confer fire resistance in as little as 10 years, whereas slower growing species on poorer quality sites and denser canopies may take longer than 30 years (Dey & Schweitzer, 2015). Site quality and light environment also impact the growth of competing stems in the understory, adding an additional level of complexity when considering the relative competitive status of oaks during a fire-free interval (Iverson et al., 2017). Another consideration is the timing of the fire-free interval. Seedling establishment is largely dependent on oak mast years, where the acorn crop can overwhelm predation. Without the resulting influx of oak seedlings, open space created by repeated prescribed fire will be rapidly used by other species (Brose et al., 2012). Thus, the timing and duration of a fire-free interval is site-specific and depends on the presence and species composition of small seedlings, light availability, and site quality.

The results of repeated prescribed fires in the study area from 2002 to 2010 were published in Arthur et al. (2015) and showed that repeated burning significantly reduced midstory and sapling stem density and basal area of competing mesic stems, but also for oak stems. The study also showed that trees continued to experience mortality and declines in crown vigor several years following prescribed fire, which raised concern over the overall impact of repeated fire. This is especially troubling if seed-producing oaks in the overstory show continued decline. Additionally, the study provided evidence that repeated prescribed fire is effective at slowing the expansion of maple dominance. This is important not only for the competitive status of oak, but also for preserving the diversity of habitat types across the landscape. The future forest composition, then, is dependent on which cohort of species grows into this open space created by repeated fire during a fire-free interval.

With this study I examined a fire-free interval following repeated fire to determine the changes to forest structure and species composition. Continuing mortality, tree growth, and recruitment were expected to alter stand structure differentially depending on burn treatment, fire severity, and landscape position. The more open structure measured in 2010 after burning (Arthur et al., 2015) was expected to promote recruitment of understory stems into the sapling layer (2-10 cm diameter at breast height (DBH)), but what species were recruited and in what density? Were there important differences among treatments and across landscape positions? Of particular interest was whether oak sapling recruitment was facilitated by the reduction in stand density and the decline of maple dominance following repeated fire. Evidence of oaks moving into this larger size class, approaching the diameter that would confer protection from subsequent fire (~13 cm DBH), would indicate restoration of oak recruitment. Oak recruitment is defined in this study as the growth of understory oak stems into the sapling layer, with the longer-term expectation being continued growth into the midstory and eventually the canopy. More broadly, this study provides a better understanding of how the reintroduction of a fire influences a diverse and heterogeneous landscape.

METHODS

Site Description

The study area is located in the Cumberland Plateau physiographic region of eastern Kentucky within the Cumberland District of the Daniel Boone National Forest. The region is characterized by temperate, humid climate and abundant rainfall. Annual air temperature averages 12.8° C, with mean daily temperatures of 0.5° C in January and 24° C in July (NOAA, 2019). The study area exists in a highly dissected landscape with variable terrain. Elevation ranges from 260 to 360 meters (m) and slopes range from 0% to 75%. Soil characteristics vary widely with topography, but the most prevalent series is Brownsville-Berks Channery silt loam, which are described as strongly to moderately acidic, well-drained soils characteristic of highly dissected uplands (Soil Survey Staff, NRCS USDA).

Geographic information system (GIS) data were obtained via ESRI ArcGIS version 10.6.1 to quantify landscape heterogeneity (ESRI, 2011). A heat load index (HLI) was calculated for the study area from a five-foot digital elevation model (DEM) from Kentucky's Elevation Data & Aerial Photography Program. The heat load index was accessed from an open source toolbox available in ArcMap 10.6.1 which considered the combined influence of aspect, annual solar radiation, and slope (Evans, 2014; McCune & Keon, 2002).

Species composition at the start of the study (2002) in the overstory (≥ 20 cm DBH) was dominated by chestnut oak (*Quercus montana;* 23% of overstory stem density (stems ha^{-1})), white oak (*Q. alba;* 18%), hickory (*Carya spp.;* 12%) and yellow-poplar

(*Liriodendron tulipifera*; 10%). The midstory (10-20 cm DBH) was dominated mainly by sugar maple (*Acer saccharum*; 28% of midstory stem density) and red maple (*Acer rubrum*; 16%). The sapling (2-10 cm DBH) layer was comprised mainly of sassafras *(Sassafras albidum*; 15% of sapling stem density), red maple (15%), sourwood (*Oxydendrum arboreum*; 14%), and yellow-poplar (14%). *Quercus* species comprised ~62% of overstory stem density, ~24% of midstory stem density and only ~2% of sapling stem density.

Experimental Design

Three study sites (Buck Creek, Chestnut Cliffs, and Wolf Pen) located in Bath and Menifee counties were established in 2002, ranging in size from 200-295 hectares (ha). Within each study site, three treatments were applied at random; a Fire Excluded control, a Less Frequent burn treatment (burned 2 times between 2003 and 2009), and a Frequent burn treatment (burned 5 times between 2003 and 2011; Figure 2.1).

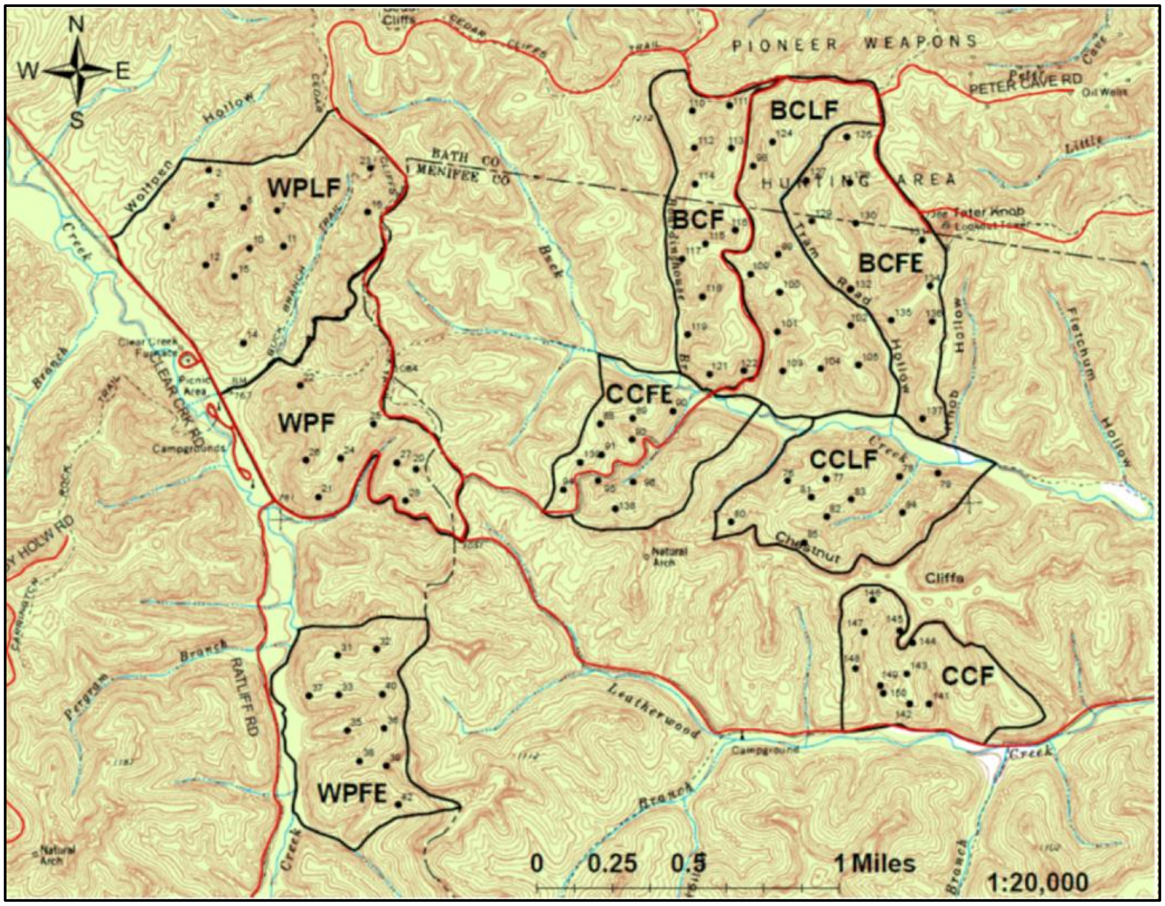


Figure 2.1 Map of project study area in the Cumberland district of the Daniel Boone National Forest, KY. Red lines represent roads, the dotted black line represents county lines, and the thick, solid black lines represent the boundaries of the nine experimental units. In unit labels, BC refers to site 'Buck Creek', CC refers to 'Chestnut Cliffs', and WP refers to 'Wolf Pen'. Treatment labels follow site labels, FE, LF, and F, referring to 'Fire Excluded', 'Less Frequent', and 'Frequent', respectively. Plots are represented by numbered dots. Contour lines are 20 feet, and all major drains are labeled along with major landforms.

Ninety-three permanent plots were established across these treatments, 33 in the Buck Creek study site, 30 in the Chestnut Cliffs study sites and 30 in the Wolf Pen study sites. Plots were chosen within treatment units using a stratified random design (Loucks et al., 2008). Plots are 10 meters by 40 meters (0.04 ha) arrayed with the long side parallel to the topographic contour (Figure 2.2).

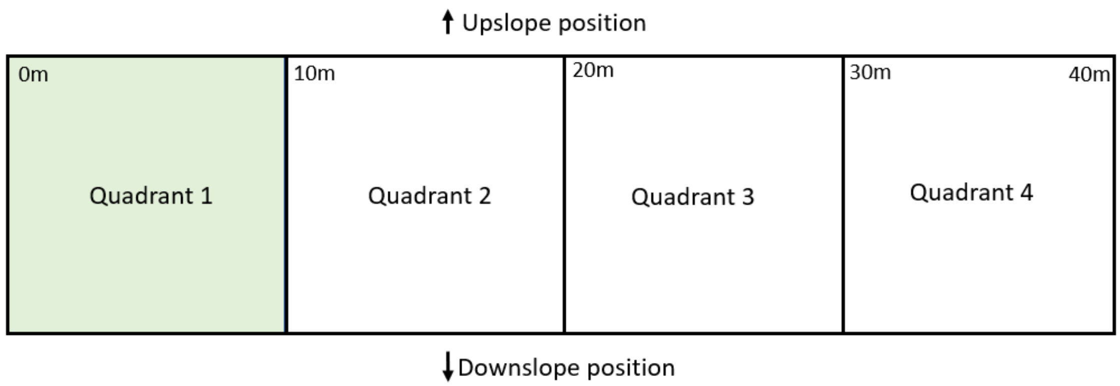


Figure 2.2 Diagram of plot layout showing orientation on the slope and important measurement locations. All plots are located in the Daniel Boone National Forest, KY. At each intersection shown in the diagram are rebar stakes that monument the site. Trees nearest to the four corners are spray painted blue for navigation. Quadrant 1 is filled in and shows where on each plot saplings were measured to calculate sapling recruitment.

To account for landscape heterogeneity, each plot was assigned a landscape position (LP) value based on tree species composition. The landscape position index ranges from 0 (xeric) to 4 (mesic) and is the weighted average landscape position value assigned to each species present in the stand (McNab et al. 2007). Values in this study area ranged from 1.83 to 3.17 and were used as a continuous or categorical variable, where plots with landscape position index values ≥ 2.7 were categorized as sub-mesic, ≤ 2.19 sub-xeric and plots with landscape position index values between 2.2 and 2.69 intermediate. Category breaks were established at the onset of the study in 2002.

Fire prescription

USDA Forest Service personnel from the Cumberland Ranger District conducted all prescribed fires in the dormant season following pre-established prescription parameters (USDA Forest Service, 2011). Fires were ignited by hand using drip torches or via aerial ignition. When data were collected for this study, plots within the Less Frequent fire treatment had not experienced fire for 9 years, and in the Frequent treatment, 7 years. Forest Service personnel aimed for timing fires during bud swell of maple species, just prior to leaf expansion. The large spatial extent of the study area combined with its highly dissected and steep terrain prevented direct measurements of flame height or spread rates. Fire intensity was therefore estimated using fire sensitive Tempilac® paints as described in Loucks et al. (2008). Prescribed fires were of low to moderate intensity, with minimum average temperatures ranging from 79° C to 644° C (Arthur et al., 2015). A composite burn index (CBI) was estimated from data collected after the first fire in 2003 to quantify fire severity, with a maximum possible value of three. Maximum char height and percent consumption of light fuels were used to calculate the CBI, which in this study ranged from zero on unburned plots to 1.7 on plots that experienced higher severity fire.

Data Collection

Forest structure data were collected in 2002 to establish pre-burn conditions, and subsequently in 2003, 2004, 2006, 2008, 2009, 2010, 2011 and 2018. Results from stand structure data collected through 2010 are published in Arthur et al. (2015). Each tree greater than or equal to 10 cm DBH within each 0.04 ha plot was identified to species and fitted with a unique aluminum tree tag in 2002 to allow for repeated measures through time. Tags were replaced and recorded as needed in subsequent measurement years. Stems 2-10 cm in diameter were measured and tagged as described above within the first 10 m by 10 m quadrant on the left of each plot facing uphill. Saplings growing into this size class were tagged and recorded to account for sapling recruitment. Trees missing from the plot during re-measurement were assumed to be dead. When possible, aluminum tags were located on logs and in decaying wood matter to be more certain that mortality occurred. Each tagged tree had data recorded on DBH and crown vigor during May through August of each measurement year. Crown vigor categories were estimated based on the percent of the canopy experiencing dieback: 0 = standing dead, 1= >50% dieback with some live canopy, 2 = 25-50% dieback, and 3 = < 25% dieback. Data collected in 2018 were compared with data collected following the last prescribed fire. Therefore, 2010 data were used in the Fire Excluded control and in the Less Frequent burn treatment, and 2011 data were used in the Frequent treatment. To avoid complication when discussing the fire-free interval, the start of the fire-free interval will be referenced as 2010 throughout this study.

Statistical Analysis

The experiment was analyzed with plot as the experimental unit (n = 93) and the site-treatment combination as a random effect (n=9). Stems were split into three size classes: 2-10 cm DBH (saplings), 10-20 cm DBH (midstory) and ≥ 20 cm DBH (overstory) based on expected size-dependent fire impacts and were established at the start of the study (Arthur et al., 2015). Species groups were selected based on species presence and abundance across all plots. Species groups considered in the analysis were: 'Oak' (*Q. alba, Q. coccinea, Q. falcata, Q. montana, Q. muhlenbergii, Q. rubra, Q. stellata, Q. velutina*), 'Maple' (*A. rubrum, A. saccharum*), yellow-poplar, sassafras, and sourwood. All other species accounted for less than 10% of stem density in any size class. Landscape position was incorporated as a categorical variable in combination with treatment and year to explore differences in stand structure and species abundance. Landscape position was included as a continuous covariate in the generalized linear mixed effects models exploring mortality and sapling recruitment. Statistical analysis was performed using R Studio version 3.5.1 ("Feather Spray").

Stand structure summary analyses were conducted for the species groups listed above to better understand trends in species abundance through the fire-free interval, among treatments, and across landscape positions. Importance values were also calculated to determine overall changes in species importance in the midstory and sapling layers. Importance values were calculated by combining the relative frequency (number of plots encountered), relative abundance (stems per hectare) and relative dominance (basal area) of each species group using package BiodiversityR (Package for Community Ecology and Suitability Analysis; Kindt & Coe, 2005) with potential values ranging from 0 to 300.

For analysis of mortality rate during the fire-free interval, a binary response of 'dead' or 'alive' was computed for each tree measured in 2018 based on comparison with data collected at the start of the fire-free interval. Trees dead in 2010 were removed from this analysis. Changes in crown vigor were calculated for each tree based on its measured crown vigor category in 2010 and in 2018, with possible categories being +2, +1, 0, -1, -2 and -3. Using an information-theoretic approach (Burnham & Anderson, 2002), Bayesian generalized linear models (binomial distribution) from package blme (Bayesian Estimation in Generalized Linear Models; Dorie, 2014) predicting mortality of stems ≥ 10 cm DBH during the fire-free interval were compared and ranked based on Akaike's Information Criterion (AIC; Burnham & Anderson, 2002) with R packages AICcmodavg (Model Selection and Multimodel Inference Based on (Q)AIC(c); Mazerolle, 2019) and MuMIn (Multi-model inference; Barton, 2009). The equation for AIC is $n\ln(RSS/n)+2k$, where n is the sample size, ln is the natural log, RSS is the residual sums of squares, and k is the number of parameters in the model. AIC scores incorporate goodness of fit (RSS) as well as the number of paramters (k). This helps prevent overfitting of the model, such that lower AIC scores indicate a better fit with the fewest number of parameters (Anderson et al., 1998). Sapling recruitment was calculated as a plot count of stems which grew into the sapling size class from 2010 to 2018. Using the same information-theoretic approach described above, generalized linear mixed-effects models (negative binomial distribution, linear parameterization (Hardin & Hilbe, 2012) created using package glmmTMB (generalized linear mixed models using Template Model Builder; Brooks et al., 2017) predicting sapling recruitment were compared and ranked. A negative binomial distribution was used to account for overdispersion (non-oak sapling recruitment) and zero-inflation (oak sapling recruitment) in the count data (Lindén & Mäntyniemi, 2011). A correction for small sample size (creating an AICc score) was used by adding $2k(k+1)/(n-k-1)$ to the AIC score following the recommendations of Anderson et al., (1998).

Models predicting recruitment into the sapling layer and mortality incorporated categorical data (fire treatment) and continuous data (landscape position, heat load index, composite burn index, and basal area) as well as fixed effects (all categorical and continuous variables listed) and a random effect (site and treatment interaction). From all subsets of the global model, supported models were those that had model likelihood values of 0.125 or greater or a change in AIC(c) value less than four from the top model (whichever cut off was reached first; Burnham & Anderson, 2002). Only supported models were presented for this study and the total model set can be found in Appendix A-2. Within these supported models, only models that included informative parameters (covariates not already included in the top model) and which conserved parsimony (did not add more degrees of freedom than the top model) were included in the final model set and subsequent analysis.

Within the information-theoretic framework, it is generally not accepted to include hypothesis tests such as p-values. Additionally, the AIC/AICc ranking does not provide R^2 values to indicate goodness of fit for models (Burnham & Anderson, 2002). Instead, the models within the final model set were considered more holistically and a Z-statistic was calculated to consider individual variables within models. A Z-statistic (absolute value of the estimated variable coefficient divided by the standard error) was computed for each variable in each top model to better understand the strengths of the relationships to mortality and sapling recruitment.

Predicted mortality and sapling recruitment were calculated by averaging the responses from each model based on their model weights using function modavgPred (compute Model-Averaged Predictions) in R package AICcmodavg (Mazerolle, 2019). To visualize the individual relationship to the response variable (mortality and sapling recruitment), each model variable was graphed with all other covariates held at their mean value. The direction and slope of each regression line reveals the general relationship between the model variable and the response variable (mortality and sapling recruitment) within the context of all top selected models.

RESULTS

Mortality and crown dieback

Annual mortality across all treatments in the overstory ranged from 1 to 1.6% during the fire-free interval. Of note, mortality of maple species in the Less Frequent treatment (3.2%) was more than double that of oak species (1.5%), resulting in over a quarter (25.9%) of overstory maple species experiencing mortality during the fire-free interval (Table 2.1) Total average annual mortality in the Less Frequent treatment (1.6%) was also higher than would be expected for background mortality rates in the Daniel Boone National Forest (~0.8% annually; USDA Forest Service, Forest Inventory and Analysis). Midstory mortality was higher than in the overstory, ranging from 2 to 3.1% annually. Midstory mortality of oak species over the fire-free interval in the Frequent treatment (3.2% annually) was higher than the total midstory average (2.7% annually) and of maple species (2.1% annually), and higher than what is considered background mortality rates (~1.2; USDA Forest Service, Forest Inventory and Analysis). Mortality was highest in the sapling size class with average rates across treatments ranging from 3.2 to 4.9% annually. In the Less Frequent treatment, mortality of maple saplings was particularly high at 6.2% annually. Overall, mortality rates were highest in the Less Frequent regardless of size class. Within the Less Frequent treatment, mortality rates of maple species were higher than for oak species regardless of size class. Note that there were no oak saplings present in the study area in 2010, thus no mortality rates were calculated for this species group and size class (Table 2.1).

Table 2.1 Average mortality (%) of overstory (≥ 20 cm DBH), midstory (10-20 cm DBH), and sapling (2-10 cm DBH) stems in all prescribed fire treatments during the fire-free interval (2010-2018) in the Daniel Boone National Forest, KY. Average percentages listed for all species, and for maple (*Acer*) and oak (*Quercus*) separately. Average annual mortality rate in parentheses (over 8 years for the Fire Excluded and Less Frequent treatments, and 7 years for Frequent treatment). NA values indicate no stems present in 2010.

Size-Class	Treatment	All spp	*Acer*	*Quercus*
Overstory	Fire-Excluded	9.6 (1.2)	10 (1.3)	8.3 (1.0)
	Less Frequent	12.4 (1.6)	25.9 (3.2)	11.7 (1.5)
	Frequent	8.4 (1.2)	6.7 (1.0)	9.2 (1.3)
Midstory	Fire-Excluded	16.8 (2.1)	12.8 (1.6)	11.3 (1.4)
	Less Frequent	25.2 (3.1)	21.5 (2.7)	19.4 (2.4)
	Frequent	18.6 (2.7)	14.6 (2.1)	22.2 (3.2)
Saplings	Fire-Excluded	25.7 (3.2)	23.4 (2.9)	*NA*
	Less Frequent	39.1 (4.9)	43.6 (6.2)	*NA*
	Frequent	25.6 (3.7)	20.8 (3.0)	*NA*

Probability of mortality of stems ≥ 10 cm DBH during the fire-free interval was predicted by treatment and fire severity (CBI). Only the top ranked model was included in analysis based on AIC and likelihood values (Table 2.2). (See Appendix A2, Table A2-1 for complete model list). Fire severity was the most important predictor variable and had a moderately strong positive relationship with mortality rate ($|Z| = 2.25$), followed by the Frequent treatment which had a weaker positive relationship ($|Z| = 1.75$) and the Less Frequent treatment, which had a weak positive relationship ($|Z| = 0.1$) (Table 2.3; Figure 2.3).

Table 2.2 Model selection table for mortality rate of stems ≥ 10 cm DBH in the Daniel Boone National Forest, KY. All subsets of the global model were ranked using AIC (Aikake's Information Criterion). Variables included in the global model include a composite burn index (CBI), heat load index (HLI), landscape position (LP), and burn treatment (Trt). All models with Likelihood values of > 0.125 or models with delta AIC of < 4 were considered for analysis. Top models are indicated by an * by the model number. Global Model: bglm (Mortality~ Trt + CBI + HLI + LP + (1|SiteTrt), family= binomial)

Model	CBI	HLI	LP	Trt	df	ΔAIC	AICWt	Likelihood
10*	0.27			+	5	0.00	0.41	1
2	0.18				3	1.52	0.19	0.47
12	0.27	0.06		+	6	1.98	0.15	0.37
14	0.26		-0.02	+	6	3.51	0.07	0.17

Table 2.3 Relative importance of model variables of top model set for total sapling recruitment in the Daniel Boone National Forest, KY. Variables include a composite burn index (CBI) and the two burn treatments, Frequent (F) and Less Frequent (LF). Variable estimates (β), standard errors (SE) and Z-statistic listed in order of variable importance for each included model.

Model	Variable	β	SE	|Z|
10	CBI	0.27	0.12	2.25
	Trt:F	0.56	0.32	1.75
	Trt:LF	0.03	0.3	0.1

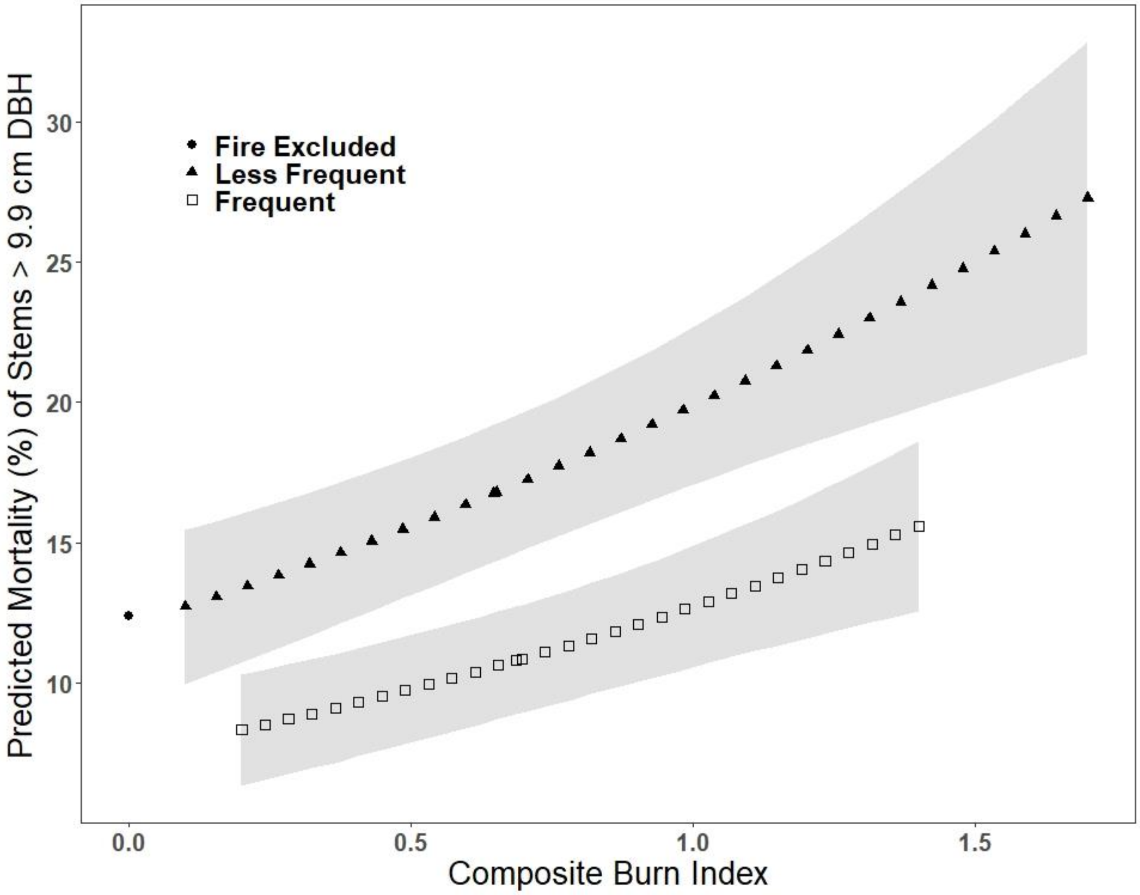


Figure 2.3 Response prediction graph for mortality of stems ≥ 10 cm DBH in the Daniel Boone National Forest during a fire-free interval (2010-2018) as predicted by top selected models. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011.

The majority of trees (50-86%) across size classes and treatments remained in the same crown vigor category from 2010 to 2018 (Table 2.4). Of those trees that declined in crown vigor during the fire-free interval, most experienced a decline of 3 crown vigor categories, which is synonymous with mortality. Far fewer trees experienced a crown decline of -2 and 81% of trees in this category also died during the fire-free interval (data not shown). Similarly, few trees experienced a crown change of -1, with a notable exception for midstory oaks in the Less Frequent treatment, 19% of which showed a decline in crown vigor by one class during the fire-free interval. Twenty-four percent of trees that declined by one crown vigor class died during the fire-free interval. Very few trees recovered crown vigor during the fire-free interval. Of note, 11% of midstory oaks in the Frequent treatment increased by one crown vigor category, and in the overstory, oaks recovered in higher proportions than maples in the Fire Excluded and Less Frequent treatments (Table 2.4).

Table 2.4 Percent change in crown vigor category from 2010 to 2018 for saplings (2–10 cm DBH), midstory (10–20 cm DBH) and overstory (≥ 20 cm DBH) trees in all prescribed fire treatments in the Daniel Boone National Forest, KY. Where 0 indicates no change, negative numbers indicate decline in crown vigor, and positive numbers indicate improvement in crown vigor. Percent change in crown vigor is shown for all species, then for maple (*Acer* spp.) and for oak (*Quercus* spp.).

		-3			-2			-1			0			1			2		
Size-Class	Treatment	All spp	Acer	Quercus	All spp	Acer	Quercus	All spp	Acer	Quercus	All spp	Acer	Quercus	All spp	Acer	Quercus	All spp	Acer	Quercus
Saplings	Fire-Excluded	16.83	15.96	NA	8.42	7.45	0.00	5.94	6.38	NA	64.85	68.09	NA	2.97	2.13	NA	0.99	0.00	0.00
	Less Frequent	29.69	30.77	NA	6.25	10.26	0.00	10.94	7.69	NA	50.00	48.72	NA	3.13	2.56	NA	0.00	0.00	0.00
	Frequent	20.51	20.83	NA	7.69	0.00	0.00	2.56	0.00	NA	56.41	70.83	NA	12.82	8.33	NA	0.00	0.00	0.00
Midstory	Fire-Excluded	13.36	11.20	7.55	3.44	1.60	3.77	4.58	4.80	1.89	73.66	79.20	81.13	4.58	2.40	5.66	0.38	0.80	0.00
	Less Frequent	13.21	9.23	12.90	8.81	7.69	9.68	11.32	7.69	19.35	59.75	69.23	58.06	5.66	4.62	0.00	1.26	1.54	0.00
	Frequent	11.29	10.91	22.22	6.45	3.64	0.00	5.65	3.64	5.56	70.97	80.00	61.11	4.84	1.82	11.11	0.81	0.00	0.00
Overstory	Fire-Excluded	7.35	10.00	6.37	1.10	0.00	1.27	2.94	0.00	3.82	86.03	90.00	84.08	2.57	0.00	4.46	0.00	0.00	0.00
	Less Frequent	9.22	18.52	8.76	2.76	7.41	2.19	7.37	3.70	8.76	76.04	70.37	73.72	4.61	0.00	6.57	0.00	0.00	0.00
	Frequent	5.88	6.67	7.19	2.52	0.00	3.27	6.30	6.67	5.23	80.25	80.00	79.08	5.04	6.67	5.23	0.00	0.00	0.00

Overstory stand structure

Previous results from this study area showed that overstory stems were not significantly affected by repeated fire (Figure 2.4; Arthur et al., 2015). Likewise, during the fire-free interval stem density and basal area of overstory (>20 cm DBH) trees were relatively unaffected by treatments or landscape position (Figure 2.4). (See Appendix A2, Table A2-2 for stem densities of trees in 2002, 2010 and 2018 among treatments).

Midstory stand structure

Repeated prescribed fire from 2002 to 2010 significantly decreased stem density and basal area in the midstory (10-20 cm DBH), most notably on intermediate and sub-xeric landscape positions (Figure 2.4; Arthur et al., 2015). In 2018, after the fire-free interval, midstory stem density in the sub-xeric and intermediate landscape positions remained lower in both burn treatments compared to the Fire Excluded control, with a more pronounced difference in the sub-xeric landscape position (Figure 2.3). Additionally, stem density in the Less Frequent treatment continued to decrease in the sub-xeric position from 2010 to 2018 (Figure 2.4). In the sub-mesic landscape position, stem density in both burn treatments was not reduced in 2010, nor in 2018 after the fire-free interval (Figure 2.4). Midstory basal area also remained lower through the fire-free interval on the two drier landscape positions and not in the sub-mesic position (Figure 2.4

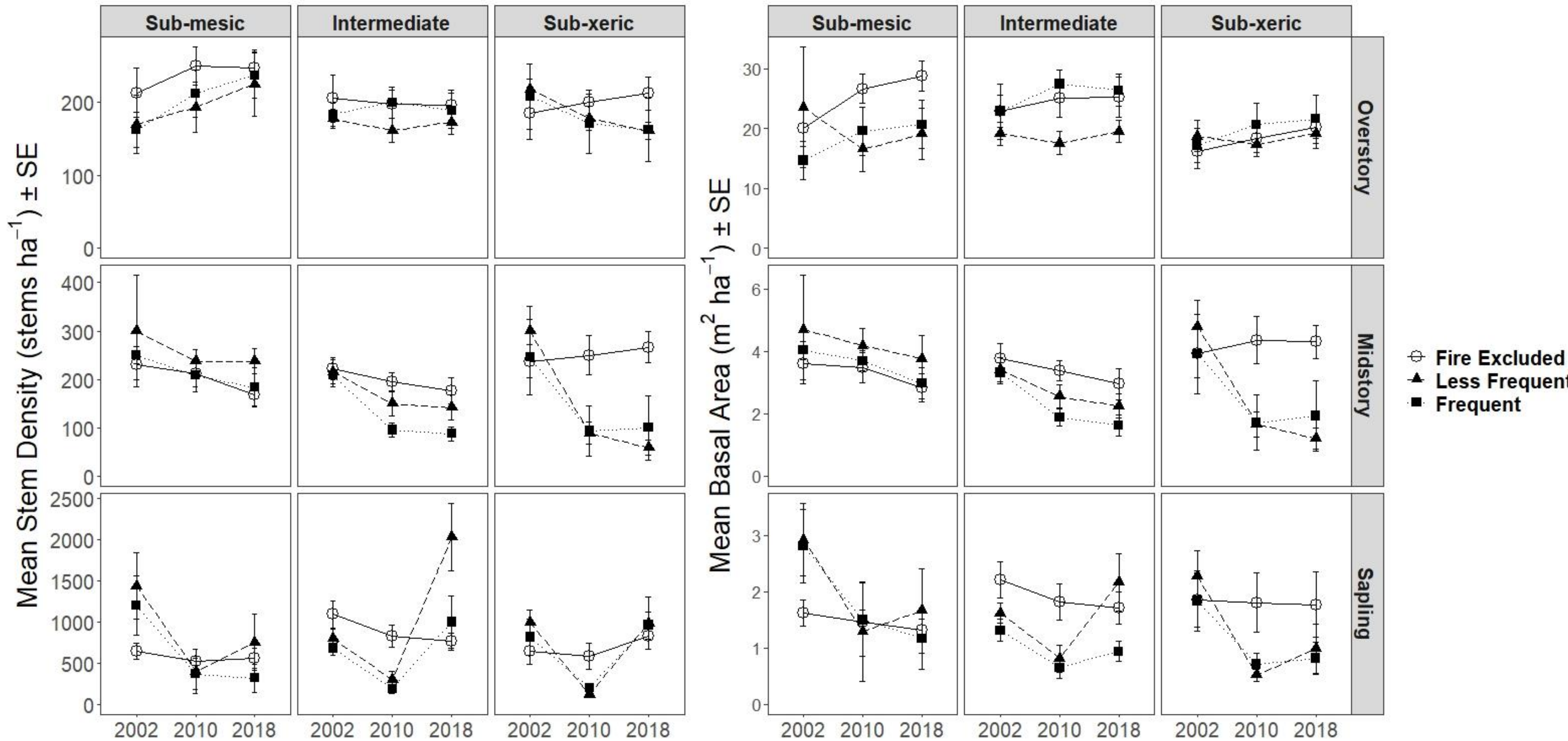


Figure 2.4 Density (stems ha^{-1}) and basal area (m^2 ha^{-1}) of overstory (≥ 20 cm DBH), midstory (10-20 cm DBH) and sapling (2-10 cm DBH) stems within each landscape position in the Daniel Boone National Forest, Kentucky. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011. Error bars represent ± one standard error

Maples, oaks and yellow-poplar were dominant species in the midstory, comprising 65% of total midstory stem density (Appendix A2, Table A2-2). Oaks in the sub-xeric landscape position recovered to pre-burn density during the fire-free interval following a reduction during the period of repeated prescribed fire. In the intermediate landscape position, midstory oak stem density did not vary greatly among years or treatments (Figure 2.5). In the sub-mesic landscape position, midstory stem density of oak was much lower in both burn treatments prior to prescribed fire in 2002 and remained lower in the burn treatments throughout the study. Midstory stem density of maples declined in both the Frequent and Less Frequent treatments in the sub-xeric landscape position and continued declining in the Less Frequent treatment during the fire-free interval. Conversely, maple stem density increased in the in the Fire Excluded treatment in the sub-xeric position throughout the study duration (Figure 2.5). Moderate declines of midstory maple stem density were seen in the intermediate landscape position in the Frequent treatment during the fire free interval. Opposite of oak, midstory stem density of maples in the Fire Excluded treatment was lower than both burn treatments in the sub-mesic landscape position in 2002 prior to buring and remained lower throughout the study. Yellow-poplar was present on only the sub-mesic and intermediate landscape positions and was recruited at high densities in the Less Frequent treatment in the intermediate landscape position (Figure 2.5).

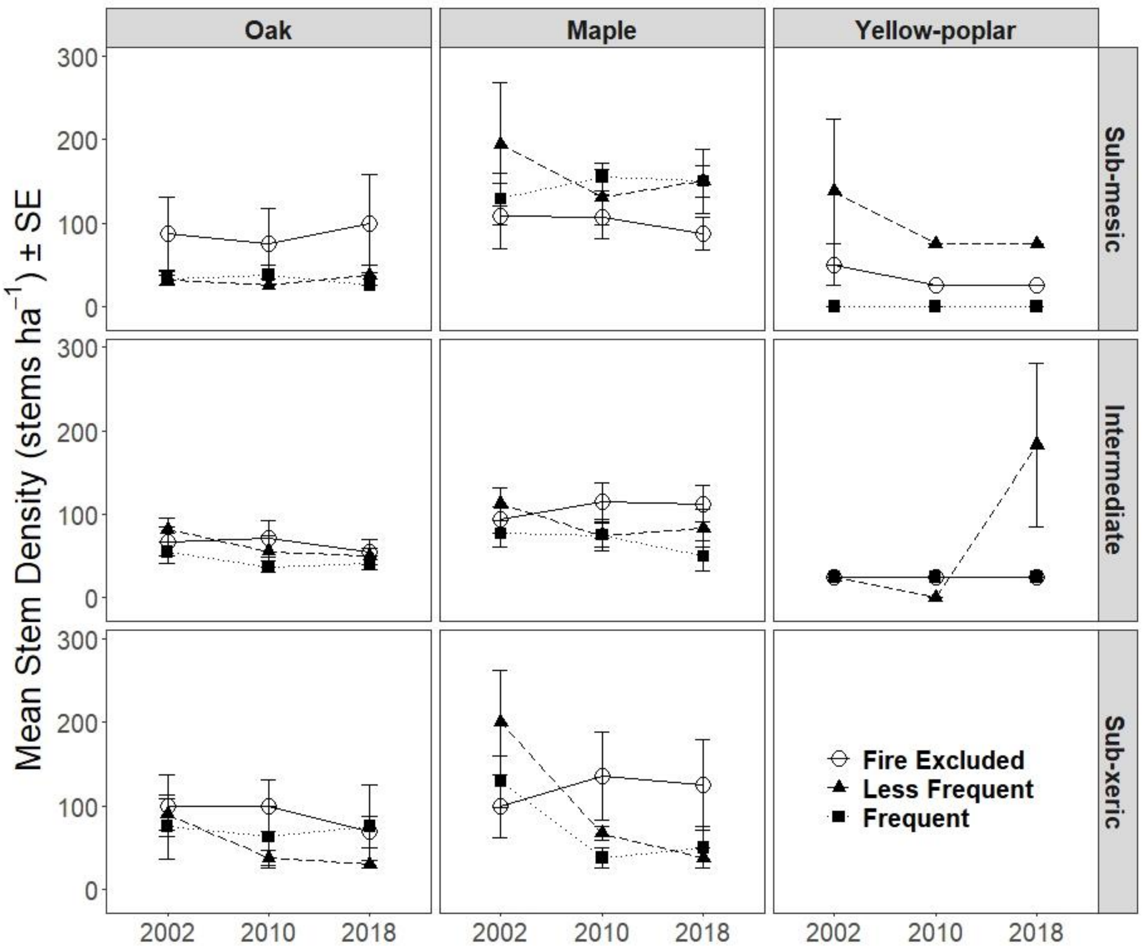
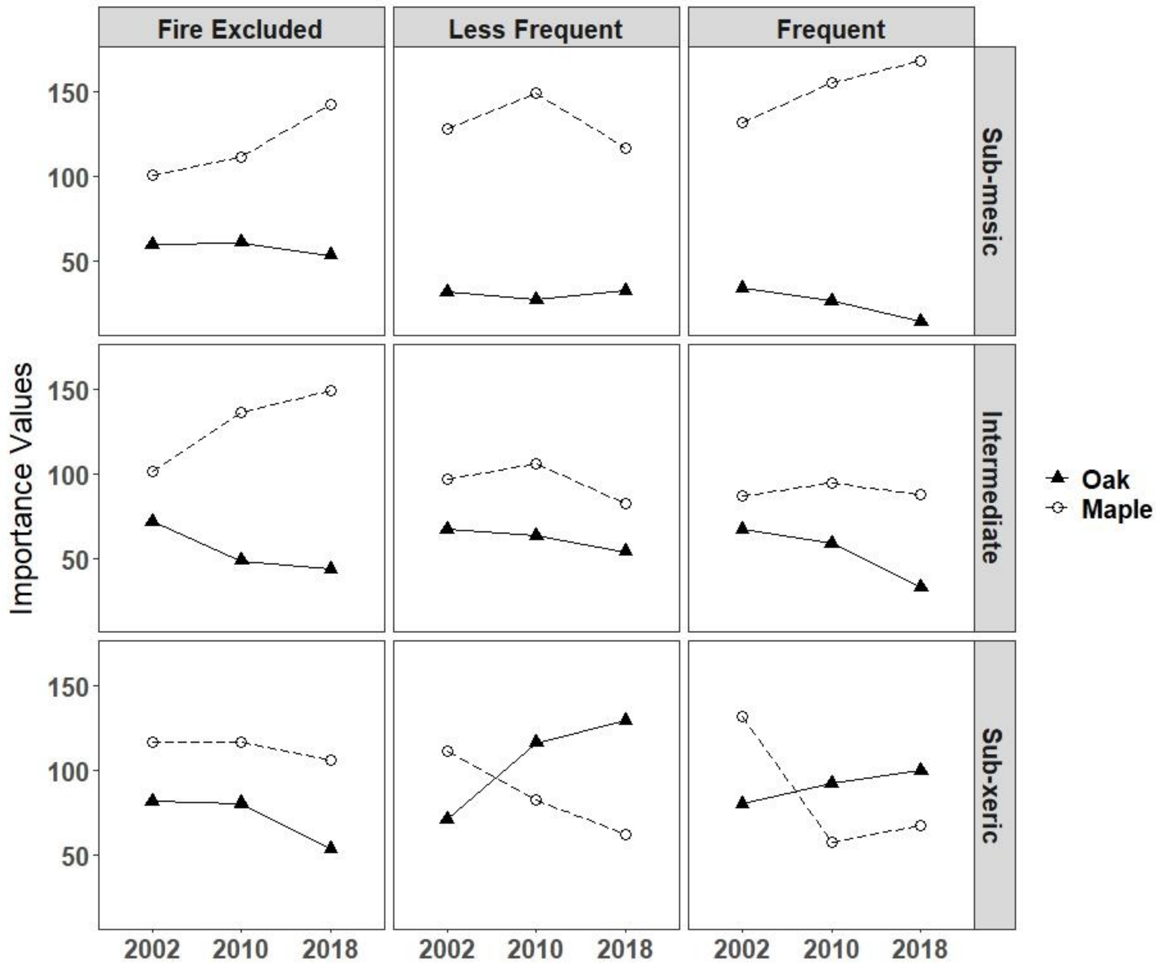


Figure 2.5 Density (stems ha^{1}) of oak (*Quercus*), maple *(Acer*) and yellow-poplar (*Liriodendron tulipifera*) in the midstory (10–20 cm DBH) within each landscape position in the Daniel Boone National Forest, Kentucky. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011. Error bars represent ± one standard error.

Despite these changes to midstory species abundance, maple stems in the midstory generally maintained the highest importance values throughout the study (Figure 2.6); the only exceptions to this were in burned sites in sub-xeric and intermediate landscape positions, where the relative importance of oak increased as the relative importance of maple declined (Figure 2.6). (See Appendix A2, Table A2-3 for importance values of species in 2002, 2010 and 2018 among all treatments and landscape positions).

Figure 2.6 Importance values of oak (*Quercus*) and maple (*Acer*) species in the midstory



(10-20 cm DBH) within each landscape position in the Daniel Boone National Forest, Kentucky. Values represent the sum of the relative frequency, abundance and dominance of the species group and can range from 0 to 300. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011.

Sapling layer

Repeated prescribed fire significantly decreased stem density in the sapling (2-10 cm DBH) size class from 2002 to 2010, most notably on intermediate and sub-xeric landscape positions (Figure 2.4; Arthur et al., 2015). During the fire-free interval, sapling stem density increased substantially. In the sub-xeric and sub-mesic landscape positions, sapling stem density returned to similar levels as that in the Fire Excluded control, whereas in the intermediate position, sapling density in the Less Frequent treatment was markedly higher than in the Fire Excluded and Frequent burn treatments (Figure 2.4).

Oak sapling recruitment during the fire-free interval occurred in both burn treatments in the intermediate and sub-xeric landscape positions (Figure 2.7). In 2018, zero oak saplings were found in the Fire Excluded treatment or in the sub-mesic position in 2018 (Figure 2.7). Prior to repeated fire (2002), oak saplings were found in all three treatments in low density: 100 stems ha^{-1} in the Fire Excluded treatment, 183 stems ha^{-1} in the Less Frequent treatment, and 133 stems ha^{-1} in the Frequent treatment, and found in all landscape positions. In 2018, stem densities of recruited oak saplings were substantially higher: 475 stems ha^{-1} in the Less Frequent Treatment and 375 stems ha^{-1} in the Frequent treatment (Figure 2.7). Overall, oaks accounted for 7% of total sapling ingrowth during the fire-free interval and the mean DBH of oak saplings was three centimeters in diameter (data not shown).

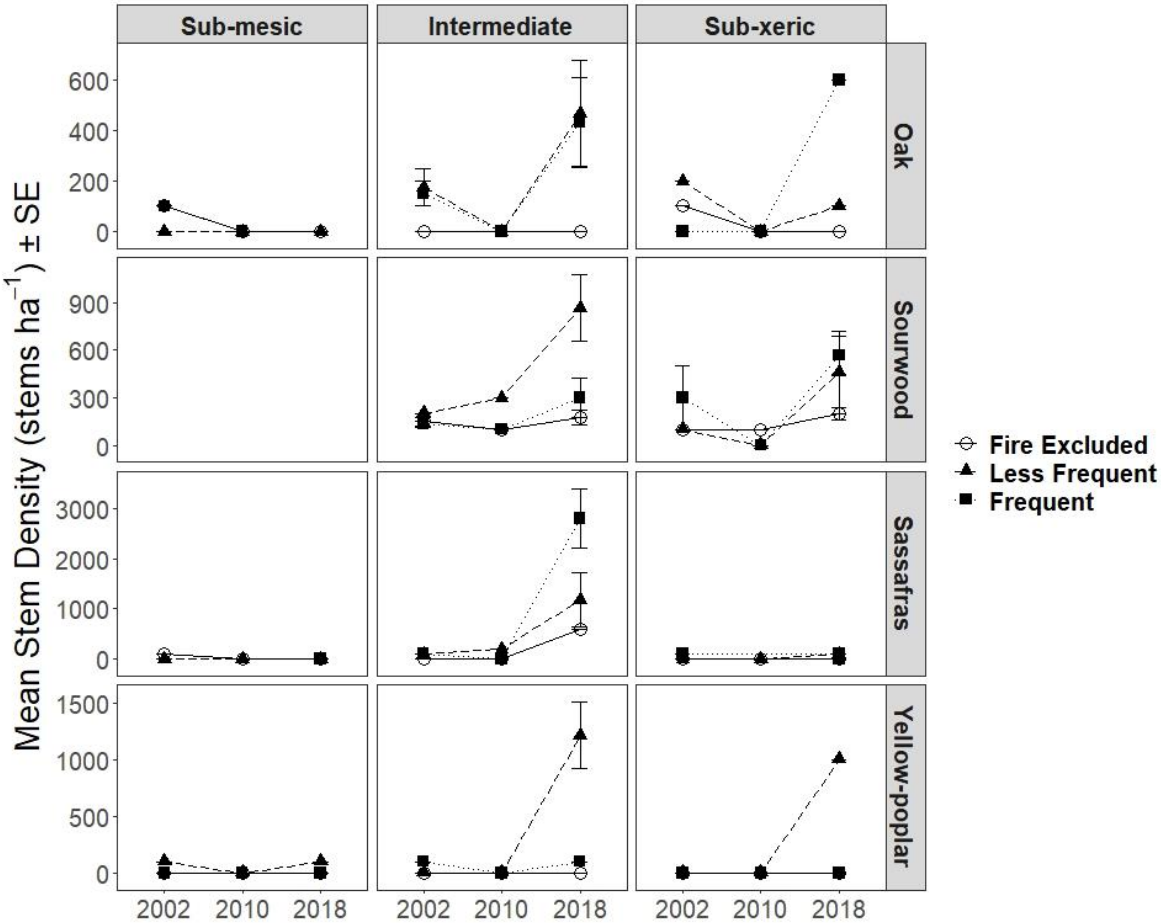


Figure 2.7 Density (stems ha[1]) of oak (*Quercus*), maple *(Acer*), sourwood (*Oxydendrum arboreum*), sassafras (*Sassafras albidum*), and yellow-poplar (*Liriodendron tulipifera*) saplings (2-10 cm DBH) within each landscape position in the Daniel Boone National Forest, Kentucky. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011. Error bars represent ± one standard error.

Total basal area (of stems ≥ 2 cm DBH) in 2018 was included in all models for oak sapling recruitment (Table 2.5). Out of the 11 supported models that were considered for analysis, the top four selected models (based on rules of parsimony and informative parameters) also included landscape position (LP), heat load index (HLI) and burn treatment (Trt) as important variables predicting oak sapling recruitment (Table 2.5). (See Appendix A2, Table A2-4 for complete model list).

Table 2.5 Model selection table for oak (*Quercus*) sapling recruitment in the Daniel Boone National Forest, Kentucky. All subsets of the global model were ranked using AICc (Aikake's Information Criterion). Variables included in the top model are basal area (BA), heat load index (HLI), landscape position (LP), composite burn index (CBI), and burn treatment (Trt). All models with Likelihood values of >0.125 or models with delta AICc of <4 were included in the analysis and included in the table below. Top models are indicated by an * by the model number.
Global Model: glmmTMB(Oak sapling recruitment ~ Trt + BA + HLI + LP + CBI (1|SiteTrt), family= nbinom1)

Model	BA	HLI	LP	CBI	Trt	df	ΔAICc	Likelihood	AICcWt
1*	-2.33	0.64				5	0	1	0.19
2*	-2.74		-0.65			5	0.109	0.947	0.18
3	-2.72	0.55	-0.54			6	0.242	0.886	0.168
4	-2.26					4	1.592	0.451	0.086
5	-2.31	0.61			+	6	2.425	0.297	0.056
6	-2.69		-0.63		+	6	2.498	0.287	0.054
7	-2.68		-0.64	0.09		6	2.499	0.287	0.054
8	-2.76	0.57	-0.54	-0.06		7	2.757	0.252	0.048
9	-2.7	0.55	-0.54		+	7	2.783	0.249	0.047
10*	-2.25				+	5	3.048	0.218	0.041
11*	-2.11			0.24		5	3.52	0.172	0.033

Basal area was the strongest explanatory variable in all four selected models as indicated by the Z values ranging from 3.84 to 4.69 (Table 2.6). Oak sapling recruitment was strongly and negatively related to basal area, heat load index had a moderate positive relationship with oak sapling recruitment ($|Z|$= 1.94), landscape position had a moderately negative relationship with oak sapling recruitment ($|Z|$= 1.76), the Less Frequent treatment had a moderate positive relationship ($|Z|$= 0.92), and fire severity had a weak positive relationship ($|Z|$=0.69) (Table 2.5; Figure 2.8). No oak saplings were found on plots with greater than 20 m^2ha^{-1} basal area, and plots that supported the greatest densities of oak saplings had less than 10 m^2ha^{-1} basal area (Figure 2.8).

Table 2.6 Relative importance of model variables of top model set for oak sapling recruitment in the Daniel Boone National Forest, KY. Variables included are basal area (BA), heat load index (HLI), landscape position (LP), and the Less Frequent burn treatment (Trt:LF). Variable estimates (β), standard errors (SE) and Z-statistic listed in order of variable importance for each included model.

Model	Variable	β	SE	|Z|
1	BA	-2.3	0.49	4.69
	HLI	0.64	0.33	1.94
2	BA	-2.74	0.67	4.09
	LP	-0.65	0.37	1.76
10	BA	-2.25	0.51	4.41
	Trt:LF	0.61	0.66	0.92
11	BA	-2.11	0.55	3.84
	CBI	0.24	0.35	0.69

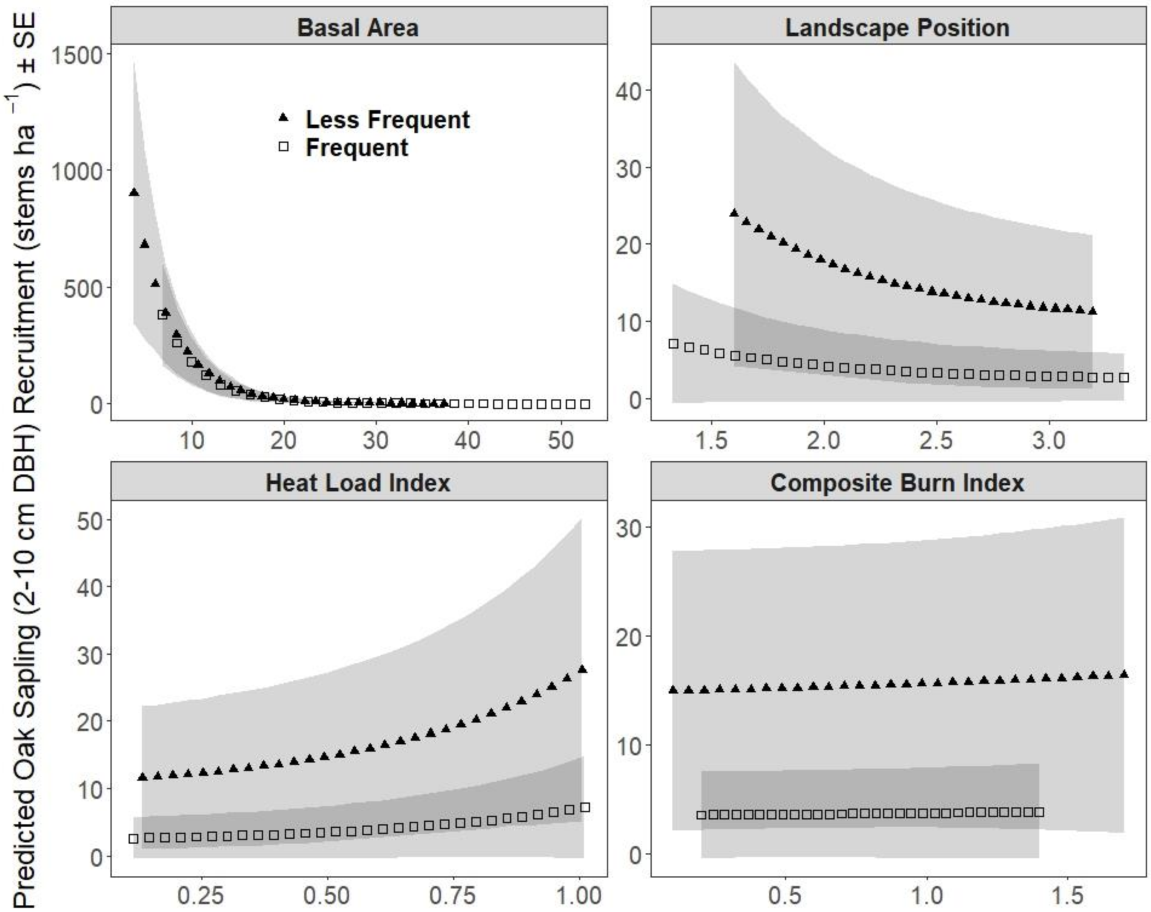


Figure 2.8 Response prediction graph of oak sapling recruitment (stems ha^{-1}) in the Daniel Boone National Forest as predicted by top selected models. To isolate the influence of each predictor variable, all other covariates are held at their mean value. This restricted the range of oak sapling recruitment for landscape position, heat load index and composite burn index to values predicted at the mean basal area. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011.

Ingrowth into the sapling layer was dominated by non-oak species. The shade-tolerant mesophytic maples and sourwood made up 16.9% and 16.8% of sapling ingrowth, respectively. Sapling recruitment of shade intolerant yellow-poplar and sassafras were 19.4% and 17.5%, respectively, compared to 7% for oaks. All other species accounted for less than five percent of sapling ingrowth, including eastern redbud (*Cercis canadensis*; 4.9%), downy serviceberry (*Amelanchier arborea*; 2.5%), black locust (*Robinia pseudoacacia*; 3.6%), blackgum (*Nyssa sylvatica*; 2.6%) and American beech (2.3%).

Repeated burning reduced maple sapling stem density in all landscape positions, but most noticeably in the sub-mesic position. However, by 2018 maples were relatively evenly spread across treatments and landscape positions due to sapling recruitment during the fire-free interval (Figure 2.7). Sourwood, sassafras, and yellow-poplar had dense but spatially variable sapling ingrowth during the fire-free interval. Specifically, all three species were absent or nearly absent from the sub-mesic landscape position and sassafras and yellow-poplar were nearly absent from the Fire Excluded treatment regardless of landscape position (Figure 2.7). Sassafras sapling ingrowth was dense in the intermediate landscape position with average density of 2,800 stems ha^{-1} in the Frequent treatment and 1,180 stems ha^{-1} in the Less Frequent treatment. Yellow-poplar was recruited in high numbers in the intermediate as well as the sub-xeric landscape positions, but only in the Less Frequent treatment (Figure 2.7). Average stem density of yellow-poplar in the Less Frequent treatment were 1,200 stems ha^{-1} (intermediate position) and 1,000 stems ha^{-1} (sub-xeric position).

Non-oak recruitment into the sapling stratum was driven primarily by landscape position (LP) and fire severity (CBI), both included in the top two selected models (Table 2.7). (See Appendix A2, Table A2-5 for complete model list). Landscape position was the most important predictor variable (|Z|= 3.43, 3.71), followed by fire severity (|Z|= 3, 3.69)(Table 2.8).

Table 2.7 Model selection table for non-oak sapling recruitment in the Daniel Boone National Forest, KY. All subsets of the global model were ranked using AICc (Aikake's Information Criterion). Variables included in the global model are basal area (BA), composite burn index (CBI), heat load index (HLI), landscape position (LP), and burn treatment (Trt). All models with Likelihood values of >0.125 or models with delta AICc of <4 were included in the analysis. Top models are indicated by an * by the model number.***Global Model: glmmTMB(Sapling recruitment ~ Trt + BA + HLI + MI + (1|SiteTrt), family= nbinom1)***

Model	BA	CBI	HLI	MI	Trt	df	ΔAICc	AICcWt	Likelihood
1*	-0.32	0.48		-0.48	+	8	0	0.331	1
2		0.61		-0.52	+	7	1.035	0.197	0.596
3	-0.32	0.45	0.11	-0.48	+	9	1.741	0.139	0.419
4	-0.4	0.33		-0.5		6	2.526	0.094	0.283
5*		0.59	0.11	-0.52	+	8	2.729	0.085	0.256

Table 2.8 Relative importance of model variables of top model set for non-oak sapling recruitment in the Daniel Boone National Forest, KY. Variables included landscape position (LP), composite burn index (CBI), basal area (BA), and the Frequent (Trt:F) and Less Frequent (Trt:LF) burn treatments. Variable estimates (β), standard errors (SE) and Z-statistic listed in order of variable importance for each included model.

Model	Variable	β	SE	|Z|
1	LP	-0.48	0.14	3.43
	CBI	0.48	0.16	3
	BA	-0.32	0.18	1.78
	Trt:F	-0.76	0.44	1.73
	Trt:LF	0.15	0.38	0.39
5	LP	-0.52	0.14	3.71
	CBI	0.59	0.16	3.69
	Trt:F	-0.88	0.45	1.96
	HLI	0.11	0.14	0.79
	Trt:LF	0.2	0.39	0.51

Other variables in selected models included basal area, heat load index, and treatment. Basal area had a negative relationship with recruitment ($|Z|= 1.78$) and heat load index had a weak positive relationship with recruitment ($|Z|= 0.79$). The Frequent treatment had a negative relationship with sapling recruitment ($|Z|= 1.73, 1.96$) while the Less Frequent treatment had a weak positive relationship ($|Z|= 0.39, 0.51$) (Table 2.8; Figure 2.9). Although the covariates landscape position, basal area and heat load index were included in the top model set, the response prediction graph shows the varying degrees to which the variables were related to non-oak sapling recruitment (Figure 2.9).

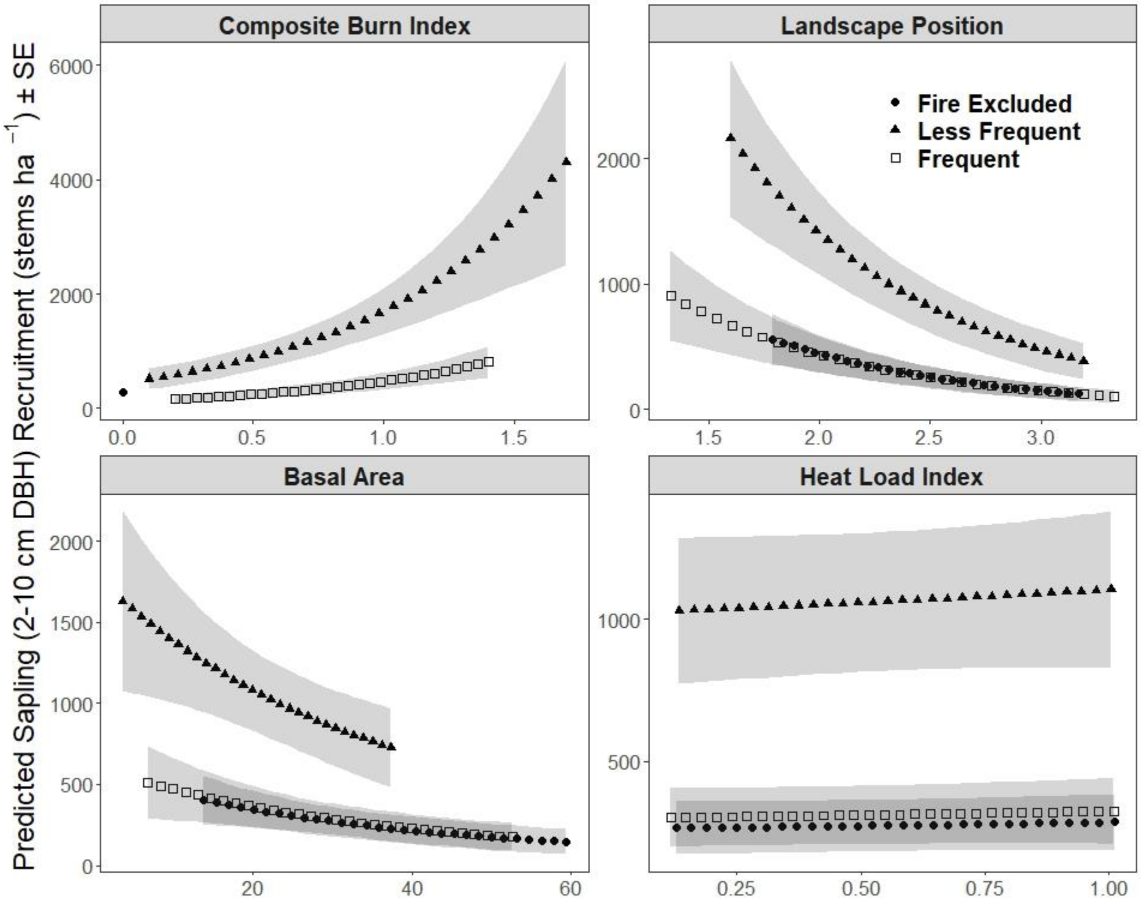


Figure 2.9 Response prediction graph of non-oak sapling recruitment (stems ha^{-1}) in the Daniel Boone National Forest as predicted by top selected models. To isolate the influence of each predictor variable, all other covariates are held at their mean value. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011.

Species abundance following the fire-free interval showed the dominance of maple generally decreasing in the burn treatments (Figure 2.10). In the Fire Excluded treatment, maples dominated the sapling layer. In the Less Frequent burn treatment, maple importance value declined from 2010 to 2018 as the relative importance of other species increased. Maple is no longer the most important species, in both burn treatments in the intermediate and sub-xeric landscape positions, having been replaced in importance by yellow-poplar and sourwood, respectively (Figure 2.10). Oak saplings also appear in the burn treatments and their importance values increased during the fire-free interval. In the Frequent burn treatment in the intermediate landscape position, the importance value of oak (41.1) exceeded that of maple (37.6) (Figure 2.10). (See Appendix A2, Table A2-6 for all importance values in 2010 and 2018 for saplings among treatments and landscape positions).

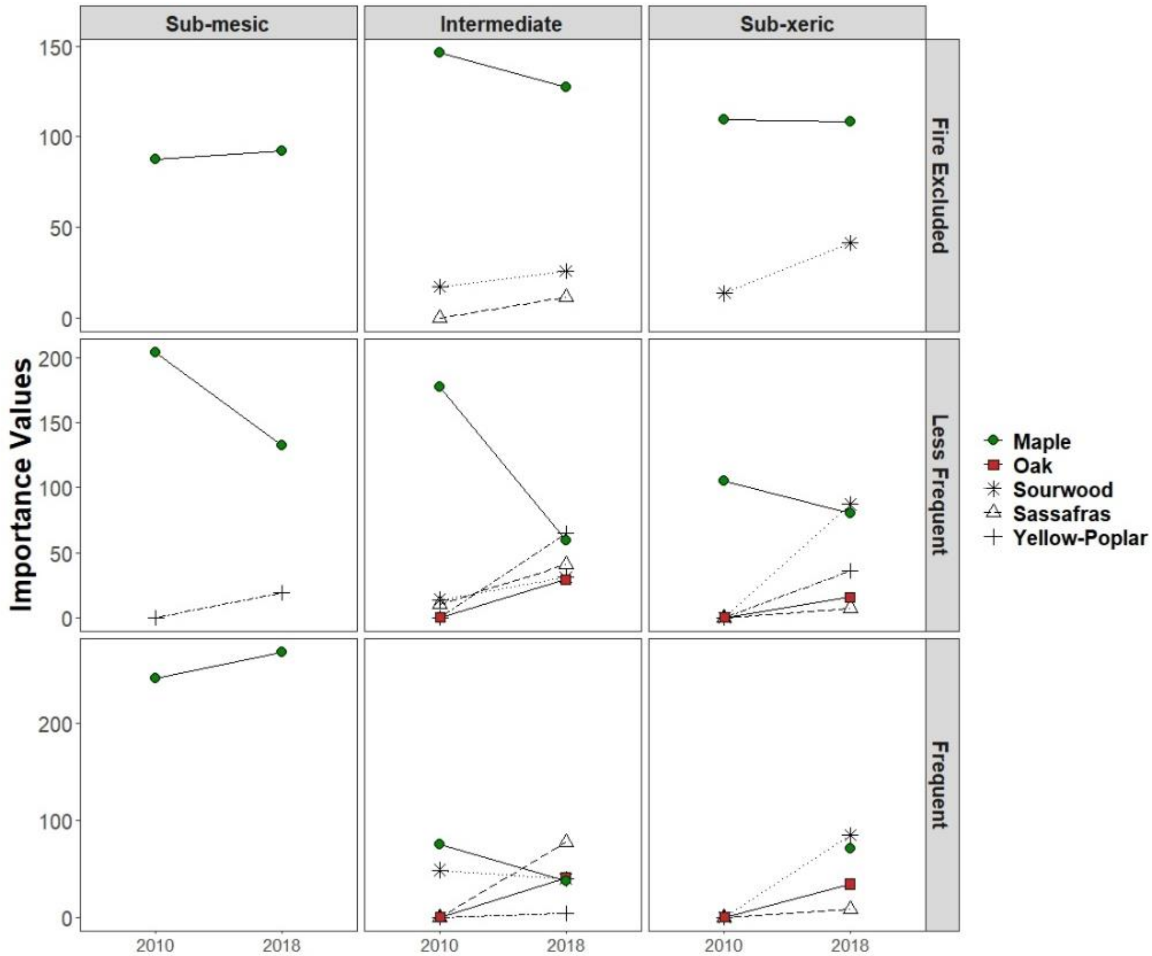


Figure 2.10. Importance values of oak (*Quercus*), maple (*Acer*), sourwood (*Oxydendron arboreum*), Sassafras (*Sassafras albidum*) and yellow-poplar (*Liriodendron tulipifera*) species in the sapling layer (2-10 cm DBH) within each landscape position in the Daniel Boone National Forest, Kentucky. Values represent the sum of relative frequency, abundance and dominance of the species group and can range from 0 to 300. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011.

DISCUSSION

Land managers are increasingly returning prescribed fire to eastern forests to reverse trends of mesophication, promote the continued presence of ecologically and economically important upland oak communities and, ultimately, to conserve landscape heterogeneity (Nowacki & Abrams, 2008; Brose et al., 2006; Brose et al., 2013). The results of this study, along with previous results from the study area, provide evidence that repeated fire followed by a fire-free interval has begun to slow the mesophication process and improve oak recruitment, primarily on drier sites (Arthur et al., 2015). In particular, understanding the consequences of a fire-free interval for oak recruitment provides a more complete understanding of prescribed fire as a management tool for upland oak forests. The fire-free interval examined in this study was shown to be necessary to allow oak sapling recruitment; however, recruitment was dominated by non-oak species and presents an ongoing management hurdle.

Mortality and crown vigor

Continued mortality of maple in the Less Frequent and Frequent treatments during the fire-free interval aligns with management objectives to create more open structure to facilitate oak recruitment. Over a quarter of overstory maples died in the Less Frequent treatment along with over 20% of midstory and 43% of sapling maple stems. Although concurrent recruitment into these size classes during the fire-free interval offsets the total change in maple stem density, the elevated mortality rate of maple species relative to oak should lead to less mesic litter addition and maple canopy influence that perpetuate the mesophication process, especially mature, seed producing maples in the overstory (Alexander & Arthur, 2010; Kreye et al., 2014; Babl et al., 2020).

Overall, trees in the Less Frequent treatment experienced markedly higher mortality than both the Fire Excluded and Frequent treatments in the overstory, midstory (with the exception of oaks in the Frequent treatment), and in the sapling layer. This result was unexpected due to the more recent 2011 fire in the Frequent treatment combined with the fires in 2006 and 2008 which we would have predicted to produce delayed mortality into the fire-free interval (Arthur et al., 2015). Fire temperature data from the initial 2003 burn showed higher temperatures for many of the Less Frequent plots (data not shown), in part due to an aerial ignition which resulted in higher than anticipated fire intensity. Although fire temperatures were not included in the CBI estimate (which measured fire severity), that the range of CBI values was higher in the Less Frequent treatment supports that plots experienced more pronounced fire effects. Furthermore, results following repeated fire showed that char height, a surrogate measure for fire intensity, was negatively correlated with crown vigor and that crown vigor declines were somewhat higher in the Less Frequent treatment (Arthur et al., 2015). A National Fire and Fire Surrogate (NFFS) study conducted in the southern Appalachians showed that the probability that a tree would experience mortality following fire was influenced by species (bark thickness) as well as pre-burn stress, as measured by crown vigor (Yaussy & Waldrop, 2010). This supports the notion that depressed crown vigor of trees within the Less Frequent treatment, especially thinner barked species like red maple, predisposed them to mortality following the 2009 burn and well into the fire-free interval.

Mortality of 22.2% of midstory oaks in the Frequent treatment presents a significant management concern. Stems in the midstory are poised to be recruited into the canopy when gaps are formed, thus substantive losses of oaks in this size class reduce the likelihood of canopy replacement by oaks. Additionally, a key tenet of the oak-fire hypothesis is that oaks survive prescribed fire at a higher rate than mesic species, in part due to thicker bark and ability to compartmentalize wounds, which this result challenges. Negative impacts to midstory oaks have been documented in other repeated fire studies, presenting a clear concern for management with repeated prescribed fire (Blankenship & Arthur, 2006; Matlack, 2013).

The results from this study showed that continued mortality, above average background mortality, continued for at least several years into the fire-free interval and was driven mainly by treatment and fire severity. Although periodic fire is not the only mortality agent affecting trees in this study area, that fire severity is still a relatively strong predictor of mortality so many years following the initial fire suggests a long-lasting legacy of burn effects. This long period of response to repeated fire makes it difficult to determine a fire frequency and return interval to meet management goals (Arthur et al., 2015). The results of this study indicate that an even longer time frame might be needed to understand the full effects of returning repeated fire back to a diverse and heterogenous landscape.

Mortality cannot be entirely linked to residual damage from repeated fire or to initial fire severity due to the abundance of other co-stressors which likely contributed to mortality throughout the study duration. Within this study area, moderate to severe drought, coupled with several defoliating insects and an ice storm in 2003, all contributed to increased mortality (Arthur et al., 2015). During the fire-free interval, precipitation records indicate mid-range to moderately moist conditions for all months of 2011 to

2018, suggesting that drought was not a continuing co-stressor (NOAA, 2019, www.ncdc.noaa.gov/temp-and-precip/drought/historical-palmers.php). However, similar to effects of repeated fire, stress from reoccurring drought could have predisposed trees throughout the study area to mortality during the fire-free interval. A study in the southern Appalachians found that drought may precipitate oak decline and contribute to trees being vulnerable to wind throw, both major contributors to oak mortality in the Central Hardwood Forest Region (Greenberg et al., 2011). Similarly, Hutchinson et al. (2005) documented high mortality of white oak (*Q.alba*) on unburned plots and suspected summer drought as a causal factor. Severe ice storms can also affect large areas and predispose damaged trees to mortality from defoliation and prescribed fire years after initial structural damage (Bragg et al., 2003; Boerner et al., 1988). Other stressors that cannot be extricated from mortality rates include outbreaks of the two-lined chestnut borer (*Agrilus bilineatus*), and increases in root rot (*Armillaria mellea*), likely due to several wet spring seasons in a row (USDA Forest Service, 2013; USDA Forest Service, 2014).

Changes in crown vigor echo concerns about continuing tree declines during the fire-free interval, but also show that some stems experienced moderate recovery, and the vast majority of trees within the study area experienced no change in crown vigor from 2010 to 2018. Portions of the -1, -2 and all of the -3 changes in crown vigor categories are synonymous with tree mortality; however, 19% of trees with a crown vigor change of -1 and 76% of trees with a crown vigor change of -2 were alive in 2018. Whether they continue to decline or recover in the future will have further impacts to forest structure and composition. In particular, 19.4% of midstory oaks in the Less Frequent fire treatment experienced a crown vigor decline of one category, which could be problematic if continued decline leads to mortality and these trees are lost to canopy recruitment.

Stand structure and oak sapling recruitment

Previous results showed the effectiveness of repeated fire in reducing sapling and midstory stem density and basal area (Arthur et al., 2015), a finding corroborated by other studies in the region (Blankenship & Arthur, 2006; Hutchinson et al., 2012a). In the midstory, stem reduction was far more pronounced in both fire treatments in the sub-xeric landscape position than in either the intermediate or sub-mesic position and in the Frequent treatment within the intermediate landscape position, supporting previous knowledge of fire behavior and burn severity in drier, more flammable landscape positions. Sapling stem density was reduced substantially in all landscape positions, as repeated fire readily kills trees of this size class (Arthur et al., 2015; Keyser et al., 2018). Reduction in stem density is thought to benefit oak reproduction by improving light availability in the understory and by decreasing competition (Abrams, 1992; Nowacki & Abrams, 2009; Brose et al., 2006). Thus, at the start of the fire-free interval (2010), the stage was set, so to speak, for oak sapling recruitment.

Oak sapling recruitment was absent on the landscape following repeated prescribed fire and was subsequently successfully recruited during the fire-free interval. This highlights the necessity of a fire-free interval following repeated prescribed fire to allow small oak seedlings in the understory to recover and grow into the sapling layer. The need for a fire-free interval has been previously noted, (Arthur et al., 2012; Fan et al., 2012; Knapp et al., 2015; Knapp et al., 2017), including another study on the Cumberland Plateau that similarly found that a fire-free interval promoted oak sapling recruitment (Poynter, 2017). Oak sapling recruitment was similarly facilitated by a fire-free interval in this study and was found only in the burned treatments and on intermediate and sub-xeric landscape positions. The absence of oak recruitment in the sub-mesic landscape position is supported by results of several other studies in the region which document improved competitive status of oak recruitment on drier landscape positions (Iverson et al., 2008; Hutchinson et al., 2012a; Iverson et al., 2017). Additionally, repeated fire substantially increased stem density of oak saplings in the study area compared to pre-burn density, improving the potential for future oak canopy replacement.

Oak sapling recruitment was driven primarily by basal area. In plots where oak sapling recruitment occurred (n= 11), basal area averaged 12 m^2ha^{-1}, notably lower than the average basal area in either burn treatment in both the intermediate or the sub-xeric landscape position (21.6 m^2ha^{-1} – 28.3 m^2ha^{-1}; data not shown). These results are supported by a study in the xeric Ozark highlands of Missouri which found that stand density must be kept low (< 14 m^2ha^{-1}) to promote sustainable density of oak recruitment. The sites in the Missouri study were far drier than those in this study and therefore presented far less competition from non-oak hardwoods. That the basal area requirement was comparable to this study in more mesic conditions underscores the importance of open structure (Larsen et al., 1997). Five out of the 11 plots that supported oak recruitment during the fire-free interval had low basal area in 2002 (< 20 m^2ha^{-1}), prior to repeated burning, but did not have any oaks in the sapling stratum. Conversely, six of the 11 plots experienced substantial mortality and resultant reductions in basal area. Reductions in basal area of up to 27.4 m^2ha^{-1} were recorded between 2002 and 2018. The appearance of oak sapling recruitment in plots which previously had a more open structure (low basal area), suggests that repeated fire reduced density of seedlings and/or small saplings that were not included in the basal area measurements (< 2 cm DBH). On many burned plots there was still dense ingrowth of small saplings (< 2 cm DBH) in 2018 that contributed to a low light environment in the understory and potentially overtopped slower growing oak saplings (personal observations). While the relationship between oak sapling recruitment and heat load index (HLI), landscape position (LP) and fire severity (CBI) were not as strong as the relationship to basal area, their inclusion in the top model set provides further evidence that oak sapling recruitment occurred primarily on dry, exposed sites where conditions were already relatively open or where fire was more effective in reducing stem density and basal area. Competing stems were more effectively reduced where more intense fire occurred on drier sites in a study in North Carolina, demonstrating how fire interacts with the landscape (Elliot et al., 1999).

Dominance of non-oak species in sapling recruitment

Despite encouraging results of increased oak sapling recruitment in the burn treatments, sapling recruitment was dominated by non-oak species. In particular, red maple, sourwood, sassafras and yellow-poplar dominated the sapling recruitment, comprising 70% of total sapling ingrowth. Non-oak sapling recruitment was driven by fire severity (CBI), and reached the highest density on intermediate and sub-xeric landscape positions. This is supported by previous findings that sprouting was significantly higher following repeated fire in the intermediate and sub-xeric landscape positions (Arthur et al., 2015). Drier sites promoting non-oak sapling recruitment support a hypothesis from a study in southern Ohio that saplings growing on more xeric sites, regardless of species, would allocate a greater proportion of biomass to root production compared to mesic sites, and would therefore respond more favorably when top-killed by repeated fire (Hutchinson et al., 2012a). Interestingly, results of this study partially contradict findings of another study in the vicinity which found that fire severity (CBI) following a wildfire was strongly correlated with oak sapling recruitment, but not for sapling recruitment of red maple, sourwood or blackgum (Black et al., 2018). Important differences exist between the studies which might help explain the variable response of sapling recruitment to fire severity. Estimated CBI values following the wildfire ranged from zero to three, much higher than what was found in this study (0-1.7), and plots within the affected wildfire area were all considered xeric upland sites (as measured by a topographic wetness index; Black et al., 2018). Thus, it may be the case that higher severity fire may be needed to see the positive relationship between fire severity and oak sapling recruitment emerge clearly on drier landscape positions.

Maples and sourwood were the most dominant shade-tolerant species recruited into the sapling layer, accounting for 17% and 16.8% of total sapling recruitment, respectively. Maple species in the sapling strata across the study plots were split evenly between red maple (48%) and sugar maple (52%) and despite significant stem density reductions following repeated fire, were the dominant component in the sapling layer in all but the sub-xeric landscape position as of 2010 (Arthur et al., 2015).

Red maple is a ubiquitous ‘super-generalist’ that is most frequently cited as contributing to the ongoing mesophication in eastern forests (Abrams, 1998; Nowacki & Abrams, 2008). However, sugar maple has also been shown to be increasing in importance across the region (Knapp & Pallardy, 2018). The results of this study contribute to the ever-growing pool of evidence that controlling the increasing dominance of maple with repeated fire is complicated; studies have varied in showing change in maple density following prescribed fire both increasing (Brose, 2010; Blankenship & Arthur., 2006), and decreasing (Arthur et al., 2015; Hutchinson et al., 2012a; Signell et al., 2005). Sourwood is a shade-tolerant upland species which was essentially absent on the sub-mesic landscape position. Sourwood in the understory is readily top-killed by low-intensity fire, but readily stump sprouts (Keyser and Zarnoch, 2014). Similar increases in stem density of sourwood following fire in were found in Sumter National Forest in South Carolina, where sprouting prevented establishment of more desirable species (Coladonato, 1992). Both maple and sourwood are shade-tolerant, and therefore should be expected to persist and remain significant competitors with oak in the understory.

Sassafras accounted for 19.4% of sapling recruitment and formed dense thickets almost exclusively within the intermediate landscape position. Positive responses of sassafras to single and repeated fire have been shown in numerous studies in the region (Albrecht & McCarthy, 2006; Alexander et al., 2008; Hutchinson et al., 2012b; Keyser et al., 2017). Relatively shade intolerant and fast growing, sassafras responds rapidly to increases in light and also prolifically root sprouts in response to high severity fire (Henderson, 1982; Dey & Hartman, 2005). Yellow-poplar was nearly absent in the sapling layer in 2002 and 2010, despite comprising 10% of the overstory. Dense yellow-poplar sapling recruitment was seen during the fire-free interval, with mean stem densities of over 1,000 stems ha^{-1}. In southern Ohio, Albrecht & McCarthy (2006) documented a similar change in yellow-poplar species abundance from nearly absent to a dominant component in the sapling layer following fire, likely due to consumption of leaf litter which promotes small-seeded species like yellow-poplar combined with reduction in stem density, as seen in this study, which favors yellow-poplar's moderately high light requirement. Yellow-poplar sapling recruitment was restricted almost entirely to the Less Frequent treatment, potentially because five prescribed burns in the Frequent treatment were sufficient to exhaust the seedbank where the two fires in the Less Frequent fire were not (Brose, 1999). Sassafras and yellow-poplar are unlikely to persist in such high density into the midstory or overstory due to competition and relative shade-intolerance. However, their initial pulse of growth following fire can outpace oak development and presents a substantial hurdle in determining timing and frequency of fire (Albrecht & McCarthy, 2006).

In general, the highest densities of sapling recruitment were in the Less Frequent treatment. In 2010, density of basal sprouts in the Less Frequent treatment was two times higher than in the Frequent treatment, and six times higher than in the Fire Excluded treatment (Arthur et al., 2015). This could potentially be due to the extended fire return interval between the two fires in the Less Frequent treatment. Following the 2003 burn, five growing seasons transpired before the second burn (2009). This would have allowed top-killed stems a relatively long period of time to allocate carbon into their root systems. Following the fire in 2009, then, sprouting was vigorous and likely allowed a greater amount of sapling recruitment compared to both the Frequent and Fire Excluded treatment.

Change in species abundance through study duration

Prior to repeated fire, midstory maples had the highest relative importance across all treatments and landscape positions. Repeated prescribed fire on sub-xeric landscape positions, however, resulted in an increase in the relative importance of oaks, so that by 2010, as well as in 2018, oaks had the highest relative importance value. This finding suggests that repeated fire on upland sites can 'prime' the midstory by increasing the abundance, frequency and size of oaks so that when large canopy gaps occur, oak species may be recruited successfully into the canopy.

Seventy-eight percent of stems in the sapling layer in 2018 were recruited during the fire free period and represent the next cohort of species that could potentially be recruited into the midstory. In both 2010 and 2018, the relative importance value of maples was higher than any other species in the Fire Excluded treatment across landscape positions. Conversely, the relative importance of maples in the burn units and on the drier landscape positions decreased through the fire-free interval (with the exception of maples in the Frequent treatment in the sub-xeric landscape position, where there were zero maple saplings in 2010). These trends are similar to that found in the midstory and suggest that repeated fire on upland sites can effectively reduce the relative importance value of maple. However, despite these trends of decreasing maple dominance, the substantive increase of sassafras, yellow-poplar and sourwood was also reflected in increasing importance throughout the fire-free interval.

Management implications

Increasing maple dominance in the absence of prescribed fire may very well limit the utility of prescribed fire moving forward as forests become less and less flammable (Abrams & Downs, 1990; Abrams & Nowacki, 2008; Arthur et al., 2012). This suggests that the window of time in which fire will be an effective restoration tool is shrinking. It is critical now, then, to identify sites that either supported upland oak communities prior to fire suppression or that could potentially support upland oak communities in the face of changing climate and reintroduce prescribed fire.

For oaks to grow into the canopy, an extended fire-free interval is required for top-killed oak seedlings to develop and be recruited into larger size classes (Arthur et al., 2012; Fan et al., 2012; Knap et al., 2015; Knapp et al., 2017). However, an extended period without subsequent burning will allow non-oak competitors to also move into size classes that are not readily controlled with prescribed fire (Harmon, 1984; Franklin et al., 2003). This trade-off between using repeated fire to control non-oak species and the moderately long fire-free interval needed for oaks to grow into larger size classes has been noted in previous work from this study area (Keyser et al., 2017) and suggests that more precise interim control methods (mechanical or chemical) may be needed to target removal of non-oak species while letting the oak saplings grow. Otherwise, shorter fire return intervals might be required to further reduce hardwood sprouting, especially just prior to a fire-free interval, to prevent dense ingrowth of non-oak species into the sapling layer and subsequent growth into larger size classes. Fire return intervals of 2-3 years have been suggested to balance the need for dampening hardwood sprouting as well as to maintain adequate surface fuel build up to sustain fire (Taft, 1997; Brewer et al., 2015; Oakman et al., 2019). Site specific differences in species composition (litter additions), climate, and co-occurring disturbance regimes (drought, ice storms) will have to be considered to identify the correct balance.

It appears that on sites that have experienced an increase in stem density and basal area associated with mesophication, either higher severity fire or other natural gap formation in the canopy will be required to facilitate oak reproduction in the absence of mechanical removals (Hutchinson et al., 2012b; Black et al., 2018). Results from this study emphasize the importance of reducing basal area to facilitate oak recruitment and suggest that a substantial amount of overstory mortality is required to restore more open structure. Evidence abounds that low intensity prescribed fire is not sufficient to alter overstory stand structure and therefore, higher severity fire might be needed. Results following a wildfire on the Cumberland Plateau in Kentucky showed that moderate to high intensity fire was effective at reducing basal area and promoting regeneration of fire-adapted species like oak and pine (Black et al., 2018). The benefit of managing naturally occurring wildfire has been increasingly recognized in western forests for promoting landscape heterogeneity and increasing resilience of forest communities to future wildfire (Barros et al., 2018). However, intense wildfires are far less frequent in eastern hardwood forests, which may require incorporation of higher severity prescribed fire to achieve these ecological benefits. At the same time, high fire severity also increases sprouting of non-oak species and can facilitate invasion by non-native species, such that follow up treatments in the form of repeated fire or herbicide will still be needed to restore fire-maintained habitats (Black et al., 2018). Additionally, planning for higher severity fire introduces a variety of planning and safety concerns that, depending on resources and social climate, might not be feasible.

It is difficult to plan management for canopy gaps created by stochastic events like windthrow, ice storms or wildfire. However, to manage for disturbance-maintained communities across large and heterogeneous landscapes, preparing the understory with repeated fire in advance of second order disturbance events has been proposed as an effective management strategy (Hutchinson et al., 2005; Dr. Mary Arthur personal communications). Moving forward, flexible, and adaptive management to properly time repeated fire as well as fire-free intervals is required for long-term restoration.

The reintroduction of repeated fire to the Central Hardwood Forest Region to promote upland oak communities can work to conserve landscape heterogeneity and to increase resilience to increased stress presented by global climate change (Vose & Elliot, 2016). My results show that fire effects are modulated by changes in the landscape. Sites that are more xeric support higher severity fire which allows reduction in stem density and basal area that facilitate upland oak reproduction. On mesic landscape positions, prescribed fire does not result in large structural changes thereby also preserving those habitat types. Many climate change models predict increased temperatures and periodic drought in the eastern US moving into the 21^{st} century (IPCC 2014; Melillo et al., 2014). This will favor more drought-adapted tree species, especially on those sites that are already drier landscape positions (Klos et al., 2009). Thus, managing for inclusion of xeric, drought-adapted communities can help prime the landscape for anticipated changes in climate, and increase landscape heterogeneity to help mitigate potential losses due to novel assemblages of co-stressors.

CHAPTER 3 UNDERSTORY RESPONSE TO PRESCRIBED FIRE FOLLOWING MIDSTORY REMOVAL IN AN OAK WOODLAND RESTORATION IN EASTERN KENTUCKY.

ABSTRACT

Oak (*Quercus* spp.) woodland habitat has been nearly lost throughout the Central Hardwood Forest Region due in part to decades of fire suppression. Woodland habitat is typified by a discontinuous canopy and an open midstory that support a diverse community of forbs and graminoids. This study examined whether a single prescribed fire applied to masticated and un-masticated upland hardwood forest stands promoted the restoration of oak woodland habitat in the Daniel Boone National Forest. Forty 0.04-hectare (ha; 0.1 acre) study plots were established on two upland hardwood ridges in 2015 to monitor restoration success. A mastication (type of mechanical thinning) treatment was applied in 2016 to 20 of the 40 plots and reduced stem density and basal area of the midstory (stems ≤ 12.7 cm DBH; 5 in DBH). A prescribed burn was then conducted on a subset of masticated and un-masticated plots in 2018 (a total of 20 plots) to assess the effects of fire and mastication on species diversity and composition. In the summer of 2019, data were collected on herbaceous vegetation cover, shrub cover, tree seedling density and species diversity. Data were compared before (2017) and after (2019) the application of the prescribed fire, and among the four treatments present in 2019: an undisturbed control (C), mastication (M), burn (B), and the combined mastication-burn (MB). Of particular interest was the invasion of Japanese stiltgrass (*Microstegium vimineum*), an invasive grass that proliferated after the mastication treatment. Results of this study show that a single prescribed burn began to improve the competitive status of oak seedlings relative to red maple (*Acer rubrum*), altered percent cover of key herbaceous lifeforms, and significantly increased species diversity and floristic quality compared to the control treatment. Repeated prescribed fire will be needed to combat sprouting of tree seedlings and woody shrubs that threaten the further establishment of native grasses and forbs characteristic of oak woodlands. Prescribed fire also significantly increased the cover of Japanese stiltgrass, complicating results from the study and potentially obfuscating the efficacy of prescribed fire and mastication for oak woodland restoration in the absence of Japanese stiltgrass.

INTRODUCTION

Oak woodland communities have declined dramatically across the eastern US since early European settlement (Guyette et al., 2002). Oak-dominated communities were historically located along ridgetops in eastern Kentucky, where Indigenous Peoples used fire to manage forests for hunting and later for agriculture. Paleoecology studies have found increases in pollen from xerophytic and fire-adapted species and charcoal beginning approximately 3,000 BP. The concurrent increase of charcoal and pollen from fire-adapted species indicates an increase in frequent surface fire which promoted the dominance of oaks (Fowler & Konopik, 2007; Delcourt & Delcourt, 1998; Delcourt et al., 1998). The devlopment of oak woodlands were likely restricted to ridges with limited soil moisture and thin soils in the southern Appalachians, but the exact spatial and temporal extent of oak woodlands throughout the Central Hardwood Forest Region is unknown (Clark & Schweitzer, 2016; Matlack, 2013). Despite this lack of clarity of the precise distribution and extent of oak woodlands prehistorically and historically, oak woodlands are a desired future condition for many federal and state agencies due to the high level of biodiversity associated with woodland habitat and maintain community diversity at a landscape scale (USDA Forest Service, 2015; Vander Yacht, 2013).

Oak woodlands differ from oak forest in several key ways. Woodland habitat is typified by a discontinuous canopy cover between 30-80% and a sparse or absent midstory. Discontinuous canopy allows a much wider gradient of light availability in the understory compared to forests, supporting a diverse community of forbs and graminoids (Vander Yacht et al., 2017; Kinkead et al., 2013). The majority of species found in woodland ecosystems are in the herbaceous layer, which is often more diverse than in either prairie or forest communities (Taft, 2009; Zenner et al., 2006). Conservation of species diversity and management for resilience against oak decline have been cited as main reasons for the creation of oak woodland habitat in the southern Appalachians (Clark & Scweitzer, 2016; Taft, 2009; Dey & Kabrick, 2015). Additionally, goals for an oak woodland restoration project in the Daniel Boone National Forest included proactively managing for the encroaching invasion by gypsy moth (*Lymantria dispar*), which is expected to exacerbate stress for upland oak trees already affected by poor site quality and density related competition (Schweitzer et al., 2014).

Due in part to decades of fire suppression, both structural and compositional changes are required to create oak woodland habitat. Multiple interacting factors, including changes to historic disturbance regimes, loss of keystone and foundational species, and changes in regional climate, have resulted in the 'mesophication' of eastern forests (McEwan et al., 2013; Nowacki & Abrams, 2008).Mesophication refers to the shift from open structured communities dominated by oak and hickory (*Carya* spp.) to more shade-tolerant, mesophytic species like maple (*Acer* spp.), resulting in denser, more shaded forests (Nowacki & Abrams, 2008). Upland oak species are moderately shade-intolerant, and as a result, seedlings of these oak species regenerating in the understory of closed forests remain suppressed and are not recruited into larger size classes. This has led to a widespread failure of advanced oak regeneration across the eastern United States (Abrams, 1992; Abrams & Downs, 1990; Nowacki & Abrams, 2008).

Returning periodic prescribed fire to the landscape in the absence of other treatments has had mixed results in restoring structure and species composition of upland oak communities (Brose et al., 2006; Arthur et al., 2012). Prescribed fire has been successfully shown to reduce midstory stem density (Arthur et al., 2015), promote oak regeneration and recruitment into larger size classes (Iverson et al., 2017; Waldrop et al., 2016), and restore herbaceous diversity indicative of pre-settlement oak woodland (Hutchinson et al., 2005; Peterson & Reich, 2008). However, results have also indicated that prescribed fire characteristic of the Central Hardwoods region is largely ineffective in killing larger diameter stems necessary to adequately increase light to the forest floor (Blankenship & Arthur, 2006; Hutchinson et al., 2012). Additionally, the positive response of many non-oak competitors via sprouting following top-kill has proven to be a considerable management concern (Blankenship & Arthur, 2006; Arthur et al., 2015).

Silvicultural treatments (thinning, shelterwood, group selection)have had moderate success in oak woodland restorations, due to the ability to remove trees in the overstory and by preferentially removing non-oak species(Kinkead et al., 2013; Albrecht & McCarthy, 2006). Oak woodland restorations that have combined thinning and prescribed fire have consistently been more successful than either method individually (Barefoot et al., 2019; Iverson et al., 2017; Vander Yacht et al., 2017). This method has been referred to as 'release burning' and is comprised of initial mechanical removals to restore structure followed by prescribed fire to stimulate grass and forb growth and to combat woody encroachment (Brose, 2014). Mastication is a form of mechanical removal, or thinning, commonly used in the western United States to reduce vertical fuel continuity to prevent canopy fires (Kreye et al., 2014). In eastern woodlands, that same reduction in vertical structure can be implemented as a restoration tool (Kane, 2010). Mastication involves a rotating blade or drum that chips and shreds small diameter, non-commercial trees. This woody material is typically left on the forest floor and creates a dense, artificial bed of woody fuel, which can have complicated impacts on fire behavior, seedling germination and understory species composition (Kane, 2009; Kreye et al., 2014).

The introduction of invasive species is a restoration concern with any disturbance-based management. Of particular concern in the eastern US is the invasive grass, Japanese stiltgrass (*Microstegium vimineum*). Japanese stiltgrass is an annual C_4 species native to Southeast Asia that was originally used as packing for imported Chinese porcelain (Culpepper et al., 2018). Japanese stiltgrass is a prolific producer of seeds that persist in seedbanks for up to five years and has become a substantial management problem throughout eastern North America due to its ability to invade a wide range of habitats (Barden, 1987; Gibson et al., 2002). Unfortunately, Japanese stiltgrass is strongly encouraged by prescribed fire as well as disturbances to soil and canopy, further complicating the restoration of oak woodlands in forests with Japanese stiltgrass (Wagner & Fratterigo, 2015).

Black et al. (2019) investigated the impacts of midstory mastication on tree seedling density, shrub and herbaceous cover and the prevalence of Japanese stiltgrass invasion in advance of prescribed fire treatments. Mastication significantly reduced stem density and basal area of the midstory and led to significant decreases in canopy cover. This consequent increase in light to the forest floor prompted a significant increase in the cover of native graminoids and forbs, providing encouraging results for oak woodland restoration. Woody encroachment by sprouting can quickly recreate canopy cover, inhibiting establishment and growth of the herbaceous layer (McGuire et al., 2001; Waldrop et al., 2016). Black et al. (2019) noted that prolific sprouting , as well as invasion of Japanese stiltgrass, remained hurdles to oak woodland restoration two years following mastication. They hypothesized that future prescribed fire would be necessary to maintain understory openness by top-killing dense tree seedling sprouts and other woody plants. However, there was also the concern that fire would exacerbate the Japanese stiltgrass invasion.

This study examined the effects of a prescribed burn on masticated and non-masticated sites to better understand the effectiveness of combined mechanical and fire treatments for oak woodland restoration. Specifically, this study examined how the addition of a prescribed fire affected tree seedling density and composition, herbaceous cover, invasion of Japanese stiltgrass, and overall changes to floristic quality. We hypothesized that a prescribed fire would reduce overall stem density of seedlings and increase dominance of fire-adapted oak species. We also hypothesized that prescribed fire would shift herbaceous cover from woody species to grasses and forbs associated with fire-maintained woodlands. Lastly, we hypothesized that prescribed fire would increase species diversity but would also increase the dominance of Japanese stiltgrass.

METHODS

Site Description

The study area is located in the Cumberland Plateau physiographic region of eastern Kentucky within the Daniel Boone National Forest. The region is characterized by temperate, humid climate and abundant rainfall. Annual air temperature averages 12.8° C (55° F), with mean daily temperatures of 0.5° C (33° F) in January and 24° C (75° F) in July (NOAA, 2019). The study is comprised of two study sites, Buffalo Branch (38°12'38" N, -83°21'27" W) and Spartman (38°15'35" N, -83°22'24" W) north of Morehead, Kentucky. The two study sites are located on broad, level ridges on rolling hills characteristic of the area. Soils are Berks, Gilpin, Latham and Tilsit series silt loams indicative of upland ridges and upper slopes. These series are well drained for the most part and range from moderately to strongly acidic. Portions across the study area are characterized by reduced depth to water table (20- 60 cm; 8-24") and restrictive fragipan layers (36- 86 cm; 14-34") that may lead to ponding (Soil Survey Staff, NRCS USDA; personal observation).

The study area experienced a salvage harvest in 2012-2013 following damage from a 2003 ice storm. Pre-harvest basal area at both sites averaged 27.5 m^2ha^{-1} (120 ft^2ac^{-1}). Basal area following the salvage harvest was 14.9 m^2ha^{-1} (65 ft^2ac^{-1}) at Buffalo Branch, and 16.6 m^2ha^{-1} at Spartman (72.3 ft^2ac^{-1})(Black, 2017). Post harvest, the dominant overstory species were white oak (*Quercus alba*), scarlet oak (*Q. coccinea*), chestnut oak (*Q. montana*), and red maple (*Acer rubrum*). Other species in the overstory and midstory layers included hickory (*Carya spp.*), yellow-poplar (*Liriodendron tulipifera*), black oak (*Q. velutina*), blackgum (*Nyssa sylvatica*), sourwood (*Oxydendrum* *arboreum*), and sassafras (*Sassafras albidum*). Prior to this study, neither site had experienced fire in >30 years.

Experimental Design

Forty experimental 0.04-hectare (0.1 acre) units (plots) were established in 2015, evenly split between the two study sites (Buffalo Branch and Spartman). Half of the plots on each ridge were subjected to a midstory removal via mastication, which created 20 control plots and 20 masticated plots. The mastication treatment was implemented by US Forest Service personnel in spring 2016, targeting stems ≤ 12.7 cm DBH (≤ 5 inch DBH) using a tracked excavator style Bobcat®, model E85, with a masticating head mounted onto a hydraulic boom. Stems were masticated if they fit within the target size class or were slightly larger (< 20 cm DBH (< 7.9 in. DBH)) and of an undesirable species, such as red maple. Some stems were not masticated if an obstacle prevented access to the tree, or if it was part of a multi-stemmed clump, where mastication of the sprouts could damage the potentially merchantable larger tree. The newly masticated materials were left undisturbed after the treatment was complete. Results from this mastication treatment were published in Black et al., 2019.

Midstory mastication was followed by prescribed fire to create a 2 X 2 factorial study to investigate the individual and combined influence of mastication and fire in an oak woodland restoration project. The burn treatment was implemented by DBNF personnel on April 18, 2018, following established prescription parameters (USDA Forest Service, 2015). Fires were hand-ignited with drip torches and temperatures were measured using pyrometers painted with Tempilac paints as described in Loucks et al., 2008.

Therefore, the experimental design in 2019 included four treatments: untreated control (C), mastication only(M), mastication-burn (MB), and burn only (B), resulting in 10 replications (plots) within each treatment. To distinguish and show comparisons between data collected in 2017 before burning, and data collected after burning (2019), we used a different naming convention to indicate the original treatment (masticated and un-masticated control), and the further subdivision of plots that were subsequently burned in 2018. , The two treatments in 2017 (control and mastication) will be referred to in this study as: C1 (control with no fire), C2 (control with planned fire), M1 (mastication with no fire), and M2 (mastication with planned fire).

Plot layout was designed to allow machine access for mastication treatment, and also to minimize variability in slope or landscape position. Plots were 0.04 ha (0.1 ac) (11.28 m radius; 37 ft) surrounded by an 18.3 m (60 ft) buffer zone between plots to accommodate skid trails for machine transport. Plot design and measurements were based on USDA Forest Service protocols (USDA Forest Service, 2015) and modified slightly where necessary to meet specific project objectives.

Data collection

Canopy closure was estimated with a convex spherical densiometer held at approximately 1.5 m (5 feet) from the ground. Measurements were taken at the center of each plot, and at each of four subplots located six meters from plot center in each cardinal direction (0°, 90°, 180°, 270°) at each plot (n= 160). Vegetation data were collected following the Forest Service Fire Monitoring Plan protocol (USDA Forest Service, 2015). Measurements of the understory community examined all herbaceous species and tree seedlings less than 0.6 m (2 ft) in height. Cover data were collected as estimated percentages in 1 x 1-meter (3 x 3 ft) quadrats at each subplot in the summer of 2019 (n= 160). Within each quadrat, vascular plant species were identified to genus or species and cover estimates were recorded separately by functional groups: tree seedlings (<0.6 m height; <2 ft), non-woody vines, forbs, and graminoids (grasses and sedges). Species identification and Latin nomenclature were derived from Jones, 2005. Functional groups were further subdivided into native and non-native for analysis. Cover estimates were all made by one person to reduce bias in assigning cover values.

Shrub cover data were collected in a nested sub-plot with a radius of 3 m (9.8 ft), located at the same four locations per plot as the understory quadrat. Within each sub-plot, all woody shrubs and woody vines were identified to species when possible, otherwise to genus, and estimated as a percent cover. Cover estimates were all made by one person to reduce bias in assigning cover values.

Tree seedling stem density was calculated (stems per area) in a nested sub-plot of 1.5 meters (5 ft), located at the same four cardinal direction locations as described above. Every tree seedling was counted and summed per location based on species and height. Height categories were defined as small seedlings, less than 0.6 m (< 2 ft) in height, and large seedlings, greater than or equal to 0.6 m (≥ 2 ft) in height and less than 5.08 cm (2 in) in diameter.

Species richness data were calculated in a meander survey (Goff, 1982) of approximately 10 minutes at each plot to record species not accounted for in the plot-based sampling described above. Plants encountered were identified to genus or species. Specimens that were only identified to genus were considered separate from those identified to species. For example, *Viola sororia* and *Viola sagitatta* were considered individually and separately from the genus *Viola*, which could have included *Viola pubescens*, *Viola hirsutula* or a number of other species within this genus. These data were combined with overstory, shrub, and understory quadrat data to estimate total species presence per plot. Each species was assigned a coefficient of conservatism (C) value which describes the ecological tolerance of a species and the habitats in which it is found (Swink & Wilhelm, 1994). Species with a C value of zero are those species with a wide range of ecological tolerance, including aggressive invaders and common ruderal species. Species with a C value of 10 are restricted to very narrow habitat types and are often intolerant of human disturbance. C values are ascribed to each species within a specific geographic region by a team of experts in the field. Currently, there are no C value assignments for Kentucky flora so values from Ohio were used for this study. Authors of the floristic quality assessment index for the state of Ohio define the extent of the geographical region for which the C values were calculated to include the Allegheny and Interior Plateaus of Kentucky (Andreas et al., 2004).

Lastly, a floristic quality index (FQI) was calculated using the following formula;

$$FQI = \overline{C}\sqrt{S}$$

where $\overline{C}$ is the average C value per plot and S is the average species richness per plot. The inclusion of these two metrics provides a weighted measure of species diversity that accounts for the identities of the individual species present in the study area.

Statistical analysis

Statistical analyses were performed using R (R Version 3.5.1, R Core team, 2018). Linear and generalized linear models were fit using package lme4 to examine changes in canopy closure, total and relative seedling stem density, and herbaceous cover (Bates et al., 2014). Treatment and site were included as fixed effects in all models and year was included in models comparing pre-burn (2017) and post-burn (2019) data for repeated measures analysis. Multiple comparisons were accounted for using functions glht and emmeans from package multcomp in R (Hothorn et al., 2008).Count data (stem density) were fit using a Poisson distribution and assessed for overdispersion by calculating the overdispersion parameter ĉ. If ĉ was significantly greater than one, data were fit using a negative binomial distribution. Residual diagnostics were completed using package 'gvlma' and visual checking of residual plots (Pena & Slate, 2006). Response variables were square-root-transformed as needed to meet model assumptions of normality and homogeneity variances. Species richness, coefficient of conservatism, and floristic quality index were compared among treatments in 2019 using the Kruskal-Wallis test to perform a one-way ANOVA with an alpha value of 0.05.

RESULTS

Canopy Closure

Results from a previous study on these study plots showed that canopy cover was significantly reduced by midstory mastication compared to the control in 2017 ($p = 0.001$) (Black et al., 2019). Remeasurements taken in 2019 found no additional significant changes to canopy cover in treatments that were burned in 2018 (C2 – B, M2 – MB) or in the unburned treatments (C1 – C, M1 – M) (Table 3.1).

Mean canopy cover in 2019 was lowest in the masticate-burn treatment, followed by the masticate treatment. Canopy cover in the burn and control treatments remained comparatively high in the absence of midstory removal. Looking at canopy cover in 2019 among treatments, canopy cover remained lower in both the masticate treatments (C–M, B–M; $p = 0.003$, $p = 0.051$) and the masticate-burn treatment (C–MB, B–MB; $p = 0.0003$, $p = 0.007$) and was not significantly affected by fire alone (C–B, $p = 0.66$) (Table 3.2).

Table 3.1 Mean canopy cover % (SE) in 2017 and 2019 in the Daniel Boone National Forest. C1–C are control plots that remained unburned, C2–B are the subset of control plots burned in 2018, M1–M are masticated plots unburned, and M2–MB are the subset of plots that were masticated and burned. Standard error in parenthesis. Significance determined at α = 0.05.

Contrast	2017	2019	P-value
C1 – C	89.2 (2.5)	90.9 (2.3)	1.00
C2 – B	86.4 (3.1)	85.4 (2.5)	1.00
M1 – M	79 (3.3)	72.5 (3.4)	0.85
M2 – MB	74.9 (4.2)	68.6 (4.8)	0.88

Table 3.2 Comparison of mean canopy cover (%) among treatments in 2019 in the Daniel Boone National Forest. Significance determined at α = 0.05

Contrast	P-value
Control – Burn	0.66
Control – Masticate	**0.003**
Control – Masticate-Burn	**0.0003**
Burn – Masticate	**0.051**
Burn – Masticate-Burn	**0.007**
Masticate – Masticate-Burn	0.85

Seedling stem density

A significant increase in stem density of small seedlings (56%, $p < 0.001$) and a significant decrease in density of large seedlings (64%, $p<0.001$) across all species was found following mastication in 2016 (Black et al., 2019). Between 2016 and 2017, small seedling stem density declined 16% over all treatments ($p=0.001$) and large seedlings density was unchanged (Black et al., 2019). Decreasing stem density of small seedlings continued in this study across treatments from 2017 to 2019 (Table 3.3). However, this decline was only statistically significant in the masticate-burn treatment.

Table 3.3 Change (%) in stem density (stems ha^{-1}) of small seedlings (<0.6 m in height) from 2017 to 2019 in the Daniel Boone National Forest. C1–C are control plots that remained unburned, C2–B are the subset of control plots burned in 2018, M1–M are masticated plots unburned, and M2–MB are the subset of plots that were masticated and burned. SE given in parenthesis. Significance determined at $\alpha = 0.05$

Treatment	2017 stem density	2019 stem density	Percent change	P-value
C1 – C2	242 (18.2)	179 (14.5)	-26%	0.07
C2 – B	220 (21.1)	168 (19.7)	-23%	0.13
M1 – M	238 (21.4)	197 (20.7)	-17%	0.46
M2 – MB	264 (20.8)	166 (14.1)	-37.1%	**0.001**

Large seedling stem density increased from 2017-2019 in the control, masticate and masticate-burn treatments, but the increase was only significant in the masticate treatment (+128%, p = 0.013; Table 3.4).

Table 3.4 Change (%) in stem density (stems ha^{-1}) of large seedlings (≥0.6 m in height) from 2017 to 2019 in the Daniel Boone National Forest. C1–C are control plots that remained unburned, C2–B are the subset of control plots burned in 2018, M1–M are masticated plots unburned, and M2–MB are the subset of plots that were masticated and burned. SE given in parenthesis. Significance determined at $\alpha = 0.05$

Treatment	2017 stem density	2019 stem density	Percent change	P-value
C1 – C	39.7 (6)	45.8 (6.4)	+39.6%	0.7
C2 – B	50.8 (4.3)	31.9 (6.5)	-37%	0.44
M1 – M	23.5 (6.4)	53.6 (13)	+128%	**0.013**
M2 – MB	24.9 (5.9)	35 (7.5)	+50.9%	0.66

Relative stem density of key species groups

Despite significant changes in total stem density of both small and large seedlings from 2017 to 2019 (Table 3.3, Table 3.4), there were no significant changes to the relative stem density (RSD) of any species in the paired treatments prior to and after the prescribed fire (C1 – C, C2 – B, M1 – M, M2 – MB). (See Appendix A3, Table A3-1 for complete list of p-values.) However, mean RSD of species varied significantly among the four treatments in 2019 (Figure 3.1). Mean RSD of large oak seedlings was significantly higher in the masticate-burn treatment (35%) compared to the control (20.4%, $p = 0.02$). Mean RSD of small oak seedlings was also highest in the masticate-burn treatment (32.6%), significantly higher than the masticate treatment ($p = 0.003$), but not the control or burn treatments ($p = 0.82$, $p = 0.72$). Small red maple seedling RSD was higher in the control (49.2%) than in the burn treatment ($p = 0.003$), but similar to the other two treatments. In the mastication treatment, density of large yellow-poplar seedlings was markedly higher in the masticated treatment compared to the control ($p = 0.02$), the burn treatment ($p = 0.004$), and the masticate-burn treatment ($p = 0.002$). Similarly, mean RSD of small yellow-poplar seedlings was significantly higher than in the control ($p = 0.0007$), burn treatment ($p = 0.034$), and masticate-burn treatment ($p = 0.0004$; Figure 3.1). Notable in the burn treatment is the significantly higher RSD of large and small sassafras seedlings. Mean RSD of large sassafras seedlings in the burn treatment was significantly higher than in the masticate ($p=0.04$) and in the masticate-burn treatments ($p=0.05$). Small sassafras seedling RSD was significantly higher than the masticate-burn treatment as well ($p=0.049$).

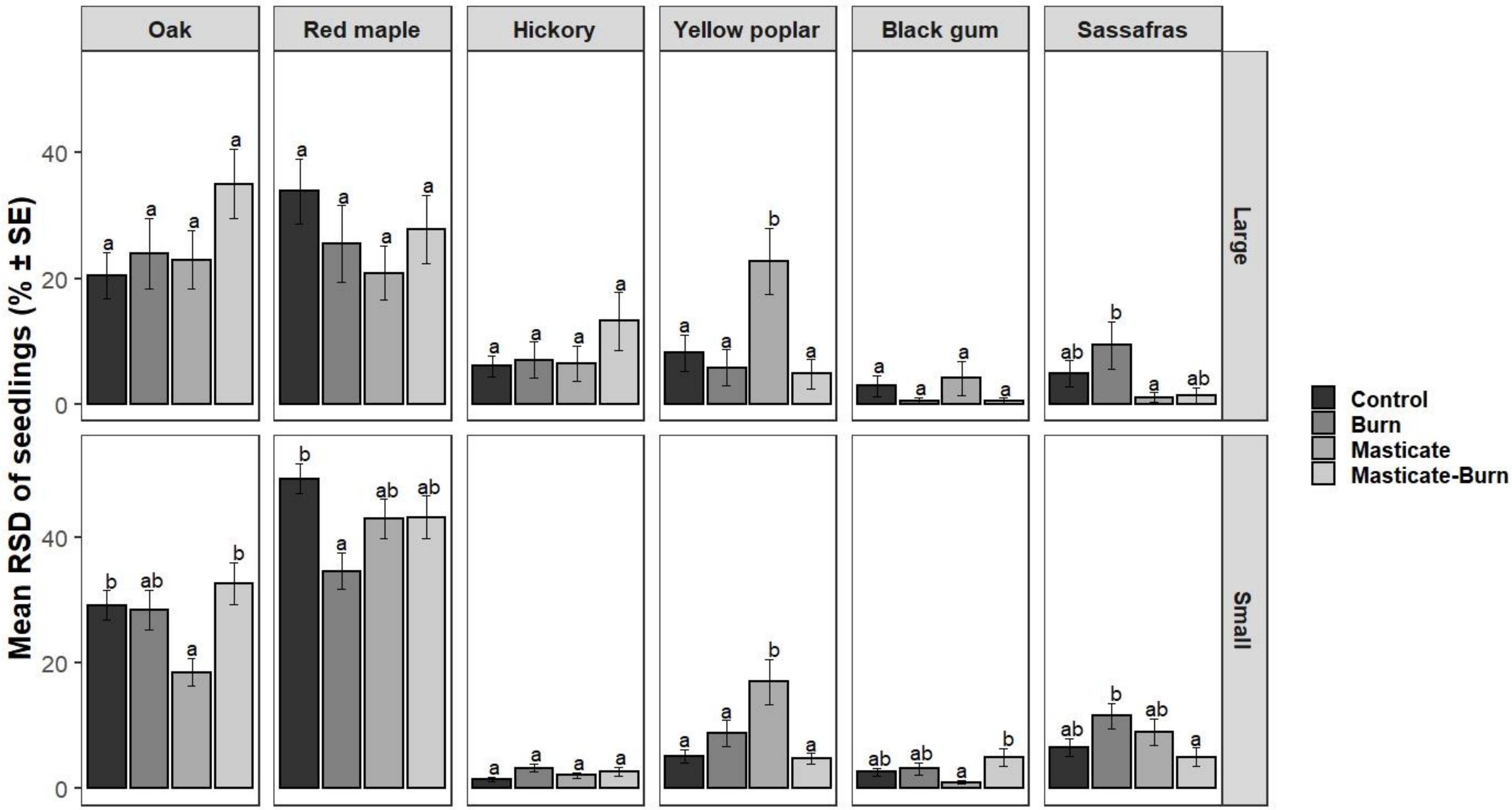


Figure 3.1 Mean relative stem density (RSD) of small seedlings (< 0.6 m in height) and large seedlings (≥ 0.6 m in height) for key species in 2019 among each of the four treatments in the Daniel Boone National Forest. Letters indicate significant differences ($p < 0.05$) among treatments within species and seedling size group

Among species, oaks and red maple had the highest relative stem densities (RSD) for both the large and small seedlings in 2019 (Figure 3.2). Mean red maple RSD of both large and small seedlings was significantly higher than oak species in the control treatment ($p = 0.009$, $p = 0.0001$). However, there were no significant differences between mean RSD of oak and maple for large seedlings in the burn treatment ($p = 0.81$), in the masticate treatment ($p = 0.69$) or in the masticate-burn treatment ($p = 0.26$). Similarly, there were no significant differences between RSD of small oak and red maple seedlings in the burn treatment ($p = 0.35$) or in the masticate-burn treatment ($p = 0.09$). Mean RSD of small red maple seedlings in the masticate treatment remained significantly higher than oak ($p < 0.0001$; Figure 3.2).

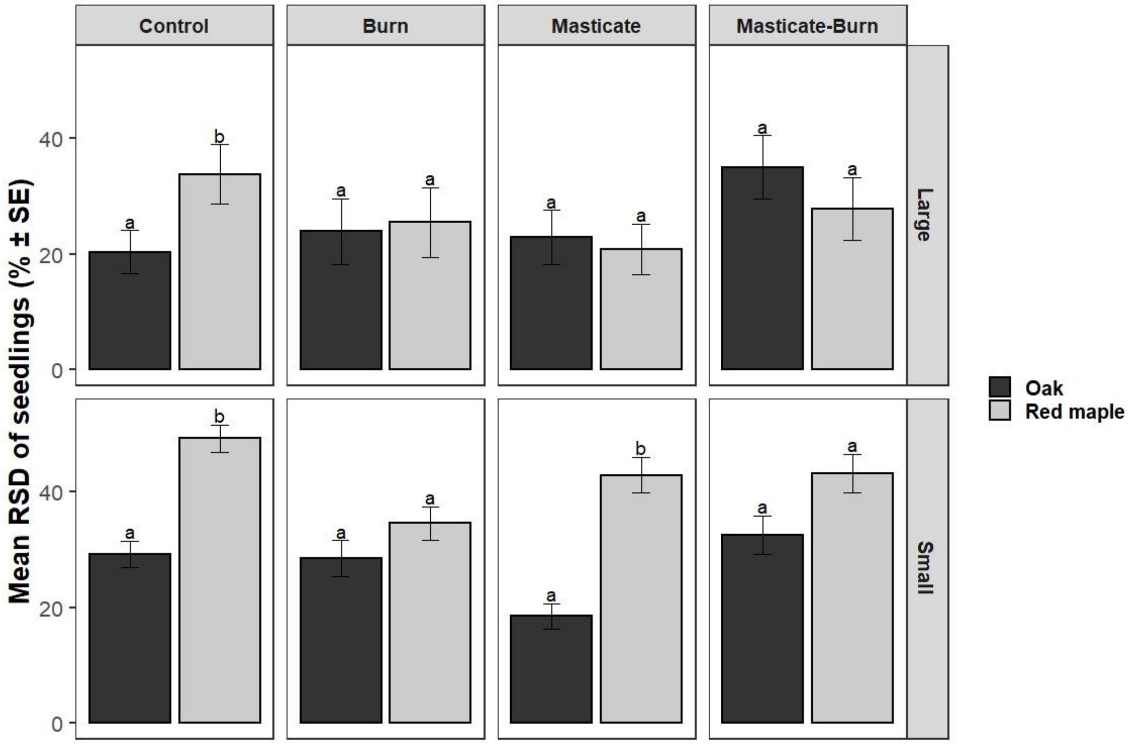


Figure 3.2 Mean relative stem density (RDS) of large seedlings (≥ 0.6 m in height) and small seedlings (< 0.6 m in height) of oak (*Quercus* spp) and red maple (*Acer rubrum*) in 2019 among each of the four treatments in the Daniel Boone National Forest. Letters indicate significant differences ($p < 0.05$) between species within treatment and seedling size group.

Herbaceous cover 2017 – 2019

Results following midstory mastication found that native forb cover increased by 204% ($p < 0.001$), cover of native graminoids increased by 552% ($p < 0.001$), and cover of the invasive Japanese stiltgrass increased by 700% ($p < 0.001$) compared to the control treatment (Black et al., 2019). Cover of native vines was significantly reduced by prescribed fire in both masticated (M2–MB) and non-masticated (C2–B) treatments while cover in non-burned treatments (C1–C, M1–M) remained unchanged (Table 3.5). The only statistically significant increase in Japanese stiltgrass was in the masticate-burn treatment (M2–MB) which experienced a 281.4% increase from 2017 to 2019 (Table 3.5).

Table 3.5 Change in mean percent cover from 2017 to 2019 in the Daniel Boone National Forest. C1–C are control plots that remained unburned, C2–B are the subset of control plots burned in 2018, M1–M are masticated plots unburned, and M2–MB are the subset of plots that were masticated and burned. SE given in parenthesis. Significance determined at α = 0.05

Life form	Contrast	2017	2019	% change	p-value
Native Shrubs	C1 - C	38.4 (10)	29.8 (7.4)	- 22.4	0.94
	C2 - B	39 (8.2)	21.4 (5.2)	- 45.1	0.25
	M1 - M	37.3 (5)	21.3 (4.3)	- 42.9	0.23
	M2 - MB	32 (6.4)	37 (6.7)	+ 15.6	0.97
Native forbs	C1 - C	2.8 (0.9)	1.1 (0.45)	- 60.9	0.71
	C2 - B	7.8 (2.7)	6.2 (2.2)	- 20.8	0.93
	M1 - M	7.4 (1.8)	6.3 (2.2)	- 14.2	0.89
	M2 - MB	8 (2)	5.4 (1.4)	- 32.1	0.93
Native Vines	C1 - C	12.6 (2.5)	8.2 (2.7)	- 35.1	0.21
	C2 - B	19.5 (3.4)	7.4 (1.5)	- 62.2	**0.002**
	M1 - M	6.5 (0.84)	6.1 (1.5)	- 6.75	0.99
	M2 - MB	12.4(2.9)	4.5 (1)	- 63.4	**0.02**
Native Graminoids	C1 - C	2.2 (2.1)	1.1 (0.7)	- 50.2	0.97
	C2 - B	2.3 (0.9)	2.9 (0.8)	+ 20.9	0.97
	M1 - M	8 (2.7)	3.8 (1.2)	- 52.3	0.63
	M2 - MB	11.6 (3.4)	13 (4.1)	+ 12.1	0.99
Invasive Grass	C1 - C	5 (3.2)	2 (1.4)	- 59.4	0.98
	C2 - B	0.25 (0.2)	4.1 (2.7)	+ 1,540	0.95
	M1 - M	9.1 (5.4)	20.7 (7.5)	+ 128.2	0.23
	M2 - MB	6.5 (2.6)	24.6 (6.5)	+ 281.4	**0.02**

Herbaceous cover in 2019

The combination of time since mastication and the application of prescribed fire resulted in several significant differences in herbaceous cover in 2019 among treatments (Figure 3.3). Mean percent cover of native forbs was significantly higher in the burn, masticate and masticate-burn treatments compared to the control ($p = 0.001$, $p = 0.003$, $p = 0.002$, respectively). The native forb layer was dominated by hog peanut (*Amphicarpaea bracteata*), New York fern (*Thelypteris noveboracensis*), cinquefoil (*Potentilla* spp), and naked-flowered tick trefoil (*Desmodium nudiflorum*). (See Appendix A3, Table A3-2 for percent species composition of lifeform groups in each treatment.) Native shrub cover was not significantly reduced by either mastication, prescribed fire, or the combination of these treatments. In all treatments, native shrub cover was primarily comprised of sawtooth blackberry (*Rubus argutus*), lowbush blueberry (*Vaccinium pallidum*) and deerberry (*Vaccinium stamineum*) (Figure 3.3, Appendix Table A-2). Cover of tree seedlings was highest in the mastication treatment (36.1 %) but only significantly higher than in the masticate-burn treatment (16.6%, $p = 0.03$).

Both native graminoids and Japanese stiltgrass increased with increasing disturbance. Mean percent cover of native graminoids was significantly higher in the masticate-burn treatment (13%) compared to the control (1.1%, $p < 0.001$), the burn treatment (2.6%, $p = 0.0006$) and the masticate treatment (3.8%, $p = 0.002$). Cover of native graminoids was made up primarily of broomsedge (*Andropogon virginicus*), sedges (*Carex spp*) and panic grasses (*Dichanthelium* spp.) (Figure 3.3, Appendix A3,

Table A3-2). Cover of Japanese stiltgrass was significantly higher in both the masticate-burn (24.6%) and the masticate (20.7%) treatments compared to the control (2%; $p < 0.001$, $p = 0.002$,) and the burn treatment (4.1%; $p = 0.005$, $p = 0.0001$). Japanese stiltgrass dominated the cover of graminoids, accounting for 64.3% to 89% of total graminoid cover across all treatments. (Appendix A3, Table A3-2).

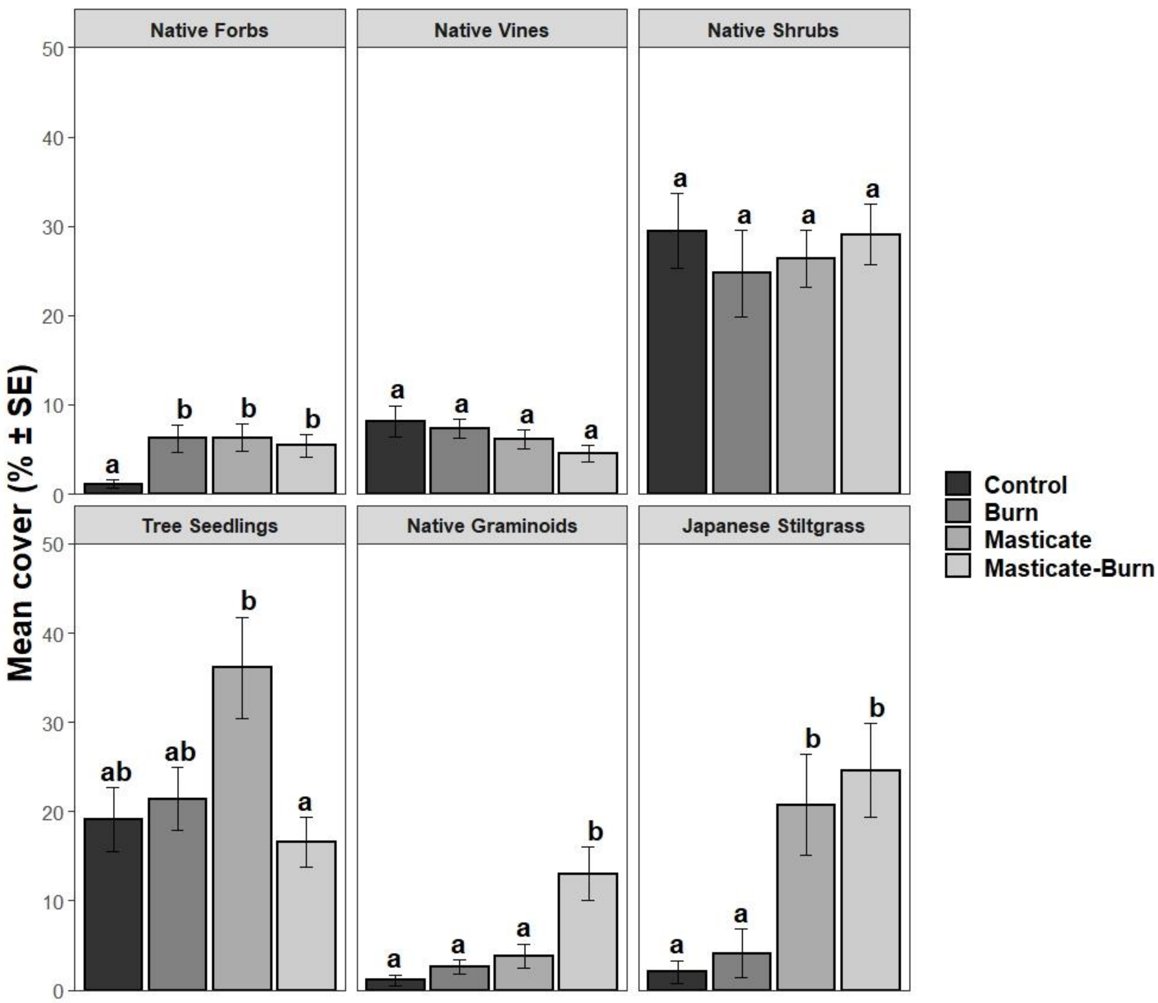


Figure 3.3 Mean percent cover of six life form groups in the herbaceous layer: Native forbs, native vines, native shrubs, tree seedlings, native graminoids, and Japanese stiltgrass in the Daniel Boone National Forest. Letters denote significant difference among treatments for each life form. Significance determined at $\alpha = 0.05$.

Species richness

Mean species richness was significantly higher in all treatments compared to the control (Figure 3.4). In total, 97 species were encountered in the control plots compared to 179 in the burn treatment, 158 in the masticate treatment and 137 in the masticate-burn treatment. (See Appendix A3, Table A3-3 for complete species list for each treatment.) Only two species were found in the control treatment that were not found in the disturbance treatments: American holly (*Ilex opaca*) and Eastern redcedar (*Juniperus virginiana*), while a variety of species were unique to the plots that experienced midstory mastication, prescribed fire, or both.

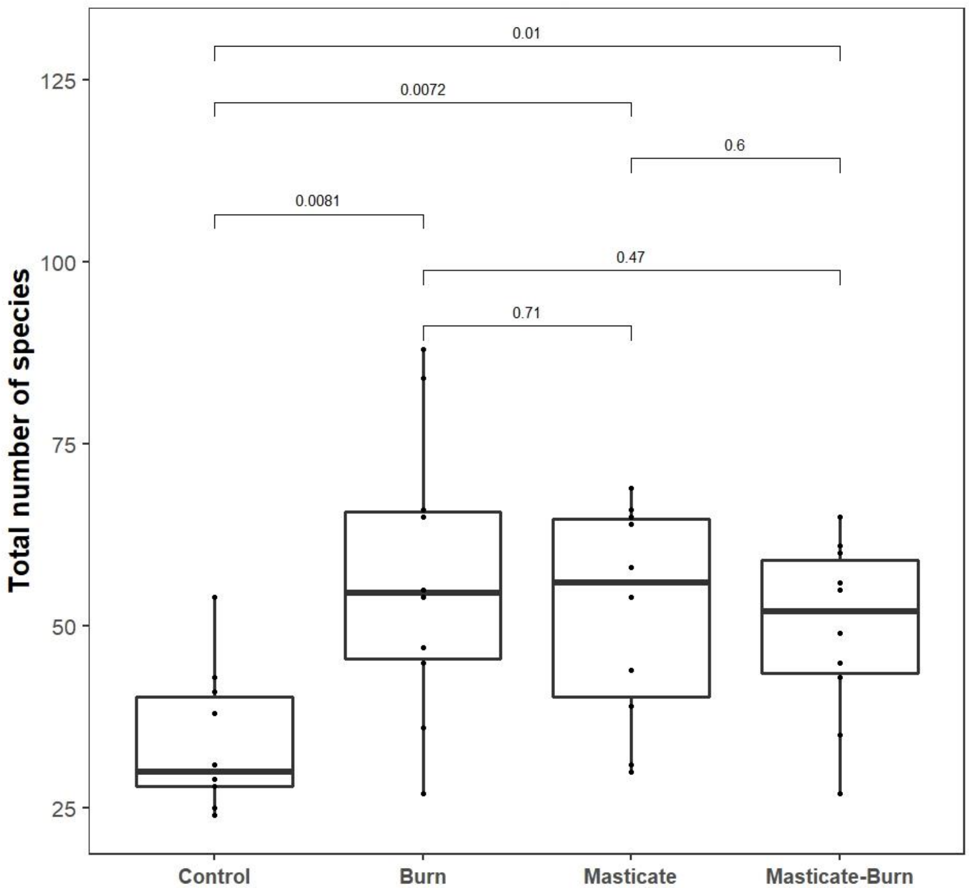


Figure 3.4 Total species richness per plot among treatments in 2019 in the Daniel Boone National Forest. P-values calculated via Kruskal-Wallis test from one-way ANOVA. Significance determined at $\alpha = 0.05$

Coefficient of conservatism

The mean coefficient of conservatism (Mean C) value was higher in the control compared to the masticate-burn treatment (p= 0.01); no other treatments differed (Figure 3.5). Coefficient of conservatism (C) values for individual species ranged from 0 to 8 in the control and masticate-burn treatments, 0 to 10 in the burn treatment, and 0 to 10 in the masticate treatment. (See Appendix A3, Table A3-3 for species list and assigned C values per treatment.)

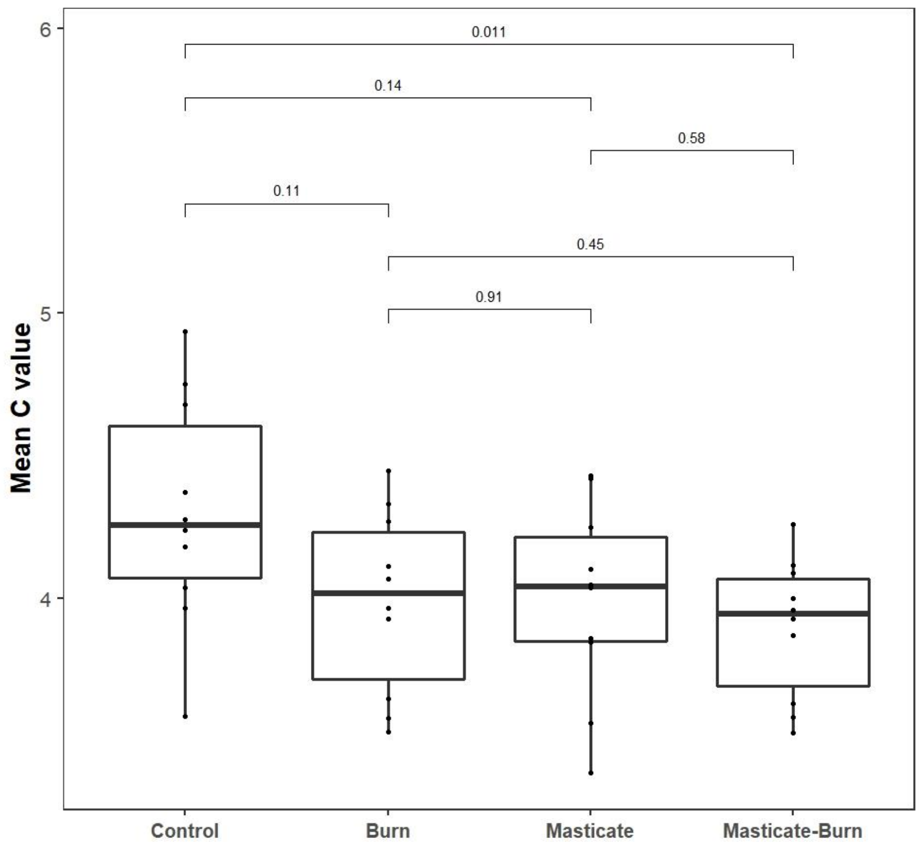


Figure 3.5 Mean coefficient of conservatism (C) among treatments in 2019 in the Daniel Boone National Forest. P-values calculated via Kruskal-Wallis test from one-way ANOVA. Significance determined at α = 0.05.

Floristic quality index

Floristic quality was significantly higher in the burn treatment, with an average FQI value of 29.5, compared to the control (0.009), but not compared to the mastication (p= 0.74) or masticate-burn treatments (p= 0.17). Mean FQI of the masticate and masticate-burn treatments were nearly significantly higher than the control (p = 0.063), indicating a trend of increasing FQI with restoration treatment.

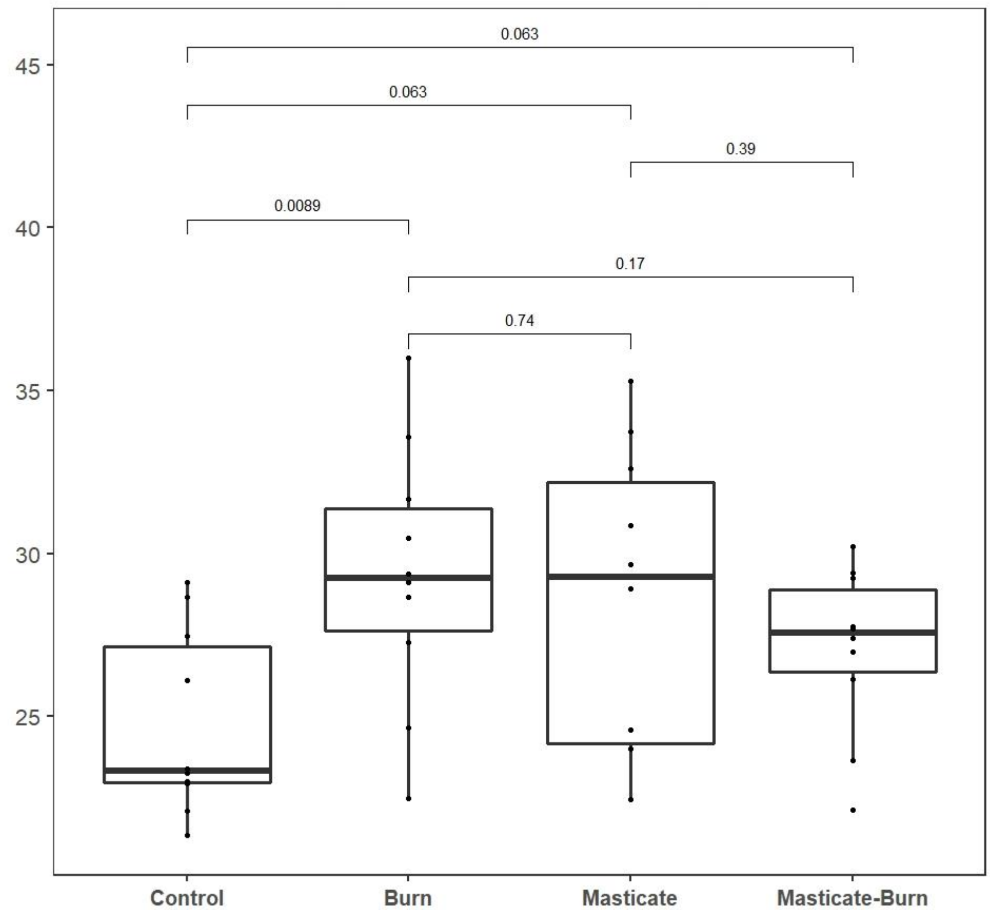


Figure 3.6 Mean floristic quality index (FQI) values among treatments in 2019 in the Daniel Boone National Forest. P-values calculated via Kruskal-Wallis test from one-way ANOVA. Significance determined at $\alpha = 0.05$.

DISCUSSION

Multiple interacting disturbances have played important roles in shaping the structure and composition of forests in the Central Hardwoods Forest Region and are important to consider when managing contemporary forests. Drought, insect outbreaks, wildfire and ice storms are reoccurring disturbances in eastern forests that create a backdrop of varied disturbance and stand structures over which management is applied (Greenberg & Collins, 2016). Ice storms, in particular, are one of the most frequent and destructive disturbance events in temperate regions and can damage extensive areas (Bragg et al., 2003). An ice storm in 2003 impacted several thousand acres in the Cumberland District of the Daniel Boone National Forest, including this study area (USDA Forest Service, 2004). To reduce vulnerability to insect and disease, salvage harvests were implemented and created the conditions which made this oak woodland restoration possible (Black et al., 2019). To this end, management implications for the larger region must consider the complex assortment of natural and anthropogenic disturbance which shaped this study area.

Combination of mechanical removals and prescribed fire

The combination of mechanical removals and prescribed fire has been shown to be an effective restoration tool in eastern oak woodlands. Results from this study area showed that mastication successfully reduced stems per acre and basal area of the midstory, increased light to the understory and promoted growth of native graminoids and forbs characteristic of oak woodland habitat (Black et al., 2019). Results from this study showed that the implementation of a single prescribed fire has begun to shift relative stem densities of tree seedlings to favor oaks. With repeated fire, the status of oak regeneration could be improved further, supported by two studies in Ohio that showed a midstory removal combined with repeated prescribed fire significantly increased oak sapling recruitment over the course of 13 years (Abella et al., 2017; Iverson et al., 2008; Iverson et al., 2017). Reduction in stand density via the combined influences of an ice-storm, salvage harvest and midstory mastication followed by the application of a single prescribed fire increased cover of native graminoids and significantly increased understory species diversity. An oak woodland restoration in Tennessee similarly found that combining mechanical removal and prescribed fire increased herbaceous diversity and cover of native grasses and forbs (Vander Yacht et al. 2017). Likewise, mechanical removals and prescribed fire in Missouri yielded a significant increase in ground cover of woodland indicator species (Kinkead et al., 2013). Despite these encouraging results, persistent dominance of woody vegetation in the understory and the continuing expansion of Japanese stiltgrass present ongoing management concerns.

Tree regeneration

Prolific stump-sprouting and increased cover of woody shrubs at this study site threatened to re-close the canopy two years following mastication (Black et al., 2019). The sprouting response to top-kill is a challenge for restoration management across eastern forests and can lead to an overall increase in stem density in years following disturbance (Blankenship & Arthur, 2006; Arthur et al., 2015). Where targeted herbicide treatments or further mechanical removals are not feasible, prescribed fire has been proposed as a tool for reducing density of these re-sprouts; however, results from single and repeated prescribed fire suggest that sprouting responses are complex and hard to predict. One study in eastern Kentucky found that basal sprouts increased with three subsequent burns, with upwards of 14,275 sprouts per hectare dominated by red maple (Blankenship & Arthur, 2006). In comparison, a different study also conducted along the Cumberland Plateau in eastern Kentucky found that a single fire promoted vigorous sprouting, but that a second fire reduced the density of red maple sprouts (Arthur et al., 1998).

The application of a single prescribed fire in 2018 significantly reduced small seedling stem density in the previously masticated treatment, but not in the burn only treatment. A large proportion of small seedlings in the masticated treatment (M2), prior to burning, were sprouts from top-killed stems (Black et al., 2019), which have been found to be especially susceptible to fire (Arthur et al., 1998). That these sprouts were more readily killed by prescribed burning suggests that combining mechanical removals and subsequent burning for reducing sprout density is an effective strategy.

In 2019, the relative stem density of small seedlings was highest for red maple in all treatments, but only significantly higher than oak in un-burned treatments, suggesting a decrease in relative dominance with the addition of prescribed fire. A study in eastern Kentucky examining seedling mortality following single and repeated fire found that although burning reduced survival of red maple seedlings, it also reduced or left unchanged the survival of oak species (Alexander et al., 2008). Yet other studies have shown that repeated fire can successfully increase oak regeneration within canopy gaps (Hutchinson et al., 2012). The relative stem density of oaks was significantly higher in the masticate-burn treatment compared to the control in 2019. This suggests that the combination of mastication and burning may contribute to the reversal of oak and maple dominance found in the untreated control treatment.

Herbaceous cover

The herbaceous layer in 2019 was dominated by small tree seedlings and woody shrubs across all treatments. Although tree seedlings and native woody shrubs are desirable in oak woodland understories, dense cover can prevent the establishment of native grasses and forbs indicative of open habitats (Elliot et al., 1999; Waldrop et al., 2016; Vander Yacht et al., 2017). In an oak woodland restoration study in Tennessee, woody shrubs and tree seedling cover increased significantly following fire and canopy reduction, and similar to this study, was comprised primarily of sawtooth blackberry. They hypothesized that repeated fire would likely be needed to shift the understory composition to herbaceous species (Vander Yacht et al., 2017). Likewise, certain vine species can also indicate ‘thicketization’, or an increase in stem density (Taft, 2009). Common greenbrier (*Smilax rotundifoliia*) was the dominant species in this group and did exhibit very dense growth form (personal observation). It is promising, then, that cover of native vines decreased significantly in both burn and masticate-burn treatments.

In 2019, mean percent cover of native forbs was significantly higher in all restoration treatments compared to the control. Increased cover of native forbs is a desired management goal for oak woodland restoration and signifies a shift from oak forest to oak woodland habitat (Vander Yacht, 2017). The appearance of several key species within the burned and masticated treatments is also encouraging as they are indicators of fire-maintained open habitat. Downy yellow false foxglove *(Aureolaria* *virginica*), butterfly pea (*Clitoria mariana*), Appalachian flat topped aster (*Doellingeria infirma*), and woodland goldenrod (*Solidago caesia*) in particular are characteristic of open habitats in eastern Kentucky and also have moderately high selectivity in habitat (C values of > 5). Species such as rabbit tobacco (*Pseudognaphthalium obtusifolium*), American burnweed (*Erechtites hieraciifolia*), and woodland sunflower (*Helianthus divaricatus*) similarly appeared in other oak woodland restoration projects and were strongly correlated with fire presence (Vander Yacht et al., 2017; Hutchinson et al., 2005).

The dominance of a legume, hog peanut, in the forb layer of treated sites while absent in the control, is also indicative of restoration of historic woodland herbaceous communities (Nelson, 2002). Legumes have been noted for their importance in herbaceous communities due to their ability to fix nitrogen, promoting invertebrate diversity and providing valuable forage for wildlife (Vander Yacht, 2017). New York fern, the most dominant forb in the burn treatment, is indicative of wet woods and swamps (Jones, 2005), suggesting that several burn plots intersected with areas with reduced depth to water table or where fragipans were present, creating ponded conditions (personal observation). Alternatively, studies have shown an increase in available water content following fire due to the removal of vegetation and the subsequent decrease in evapotranspiration (DeBano et al., 1998).

Cover of native graminoids was only significantly increased in the masticate-burn treatment, suggesting that both mastication and prescribed fire were important in promoting establishment of grass cover. Plots with cover of native grasses exceeding the average were those with canopy cover less than 50%. The plot with the highest percent cover of native grasses (126%) had a canopy cover of 35%, substantially lower than average canopy cover in any treatment, highlighting the importance of light for the restoration of native grass cover. The dominant graminoid species indicate the creation of open habitat and will contribute to the ability of fire to carry in subsequent burns. Cover by native grasses was dominated by broomsedge bluestem (*Andropogon virginica*), a warm-season perennial bunchgrass that grows in a variety of open habitats (Grelen et al., 1986). Although broadly considered a ruderal species (Jones, 2005), this grass species has an extremely high surface area to volume ratio which quickly accumulates flammable fuel that can help carry fire for subsequent burns (Lewis et al., 1976). Twenty-four percent of native grass cover was comprised of three species of panic grasses: Bosc's panicgrass (*Dichanthelium boscii*) is characteristic of semi-open dry oak-hickory woods (Jones, 2005), Cypress panicgrass (*D. dichotomum*), found in both semi-open woods and wet ground, and variable panicgrass (*D. commutatum*), also found in semi-open woodlands (Jones, 2005). Of note, poverty oat grass (*Danthonia spicata*) appeared in both the mastication and burn treatments and is considered an oak woodland indicator species (Taft, 2003). The remaining graminoid species comprised only 9% of the total native graminoid cover (see Appendix A3, Table A3-2).

Invasion of Japanese stiltgrass

The continued expansion of Japanese stiltgrass is a major hurdle for restoration. Numerous studies show a positive relationship between invasive grasses and fire, and Japanese stiltgrass is no exception, having co-evolved with fire in much of its native range (Barden, 1987). Invasive grasses are adept competitors for light, water and nutrients and have complicated oak restoration projects in Japan and North America (Mack, 1981; Tang et al., 1988). Additionally, post-fire conditions can favor the spread of Japanese stiltgrass by reducing the litter layer, reducing competition and increasing available nitrogen (Wagner & Fraterrigo, 2015).

Eradication efforts via hand-pulling were attempted in the study area prior to the implementation of prescribed fire to prevent the spread of Japanese stiltgrass, but proved infeasible across the entire study site. Although the mean percent cover in these treatments ranged between 20 – 25%, there was a great deal of variability in the cover of Japanese stiltgrass; 13 plots had percent coverage exceeding 75% and upwards of 100% (B = 1, M = 5, MB = 7). This near complete cover in a number of plots will have obvious consequences for growth and establishment of other desirable species and demonstrates the severity of the invasion. Japanese stiltgrass grows best in moist soil conditions (Culpepper et al., 2018). The observed wet spots and ponding across both ridges, then, could create refugia for Japanese stiltgrass to grow and spread. A proportion of plots within the burn treatment were located in especially wet sites, as is suggested by the prevalence of New York fern, and careful consideration should be given to site-specific conditions prior to implementing subsequent burns. Forest Inventory and Analysis data show that Japanese stiltgrass is rapidly spreading through diverse forest types in Tennessee, seemingly unbounded in stands with lower than 35 m^2ha^{-1} (152 ft^2ac^{-1}) basal area (Culpepper et al., 2018). This is partially due to physiological plasticity in regard to its shade tolerance. However, it has been demonstrated that Japanese stiltgrass likely needs a disturbance to become established (Barden, 1987).

Invasion of Japanese stiltgrass also has direct implications for oak seedling growth and survival. Japanese stiltgrass was the overwhelming competitor for growth of oak seedlings in a bottomland oak forest in Tennessee, with increases in cover of the invasive grass accounting for 99% of variation in annual growth and overall survival of northern red oak (*Q. rubra*) seedlings (Oswalt et al., 2004). While a benchmark of success in woodland restoration is the return of grass cover, the establishment of Japanese stiltgrass has serious implications for understory diversity and oak regeneration (Brewer et al., 2015).

Floristic quality

Quantifying characteristics of plant communities is critical to evaluate ecological restoration success (Hobbs & Harris, 2001). Numerous metrics exist to describe floral communities, many based on plant abundance (percent cover, density), species richness, evenness of species distribution, or selected indicator species based on expert knowledge or pre-European settlement records (Maginel et al., 2016). The floristic quality index (FQI) is comprised of two such metrics: species richness, and an ecological tolerance value assigned to individual species, the coefficient of conservatism (C value). In general, higher FQI values indicate a plant community with a high diversity of species narrowly adapted to a specific habitat type. C values range from 0 to 10, and while invasive and naturalized species almost exclusively receive a 0 C value assessment, any plant species with a very wide range of ecological tolerance is assigned a low C value. Likewise, species with C values of 10 are often rare species that indicate high quality habitat but can be applied to any species that display high fidelity to narrow habitat requirements (Andreas et al., 2004).

Reintroduction of disturbance resulted in a significant increase in species diversity, most notably in the burn treatment where 179 species were identified representing an 84.5% increase in species richness compared to the control. This increase in species following prescribed fire is supported by other studies in the region which have found that prescribed fire is linked to an increase in species diversity (Maginel et al., 2016; Vander Yacht et al., 2017). Mean C values, however, were highest in the control, although only significantly higher compared to the masticate-burn treatment. Mean C values approximated 4 for all treatments, which indicates plant species with an intermediate range of ecological tolerance that are in stable phase and can persist under disturbance (Andreas et al., 2004).

By looking at the species list within each treatment (Appendix A3, Table A3-3), we can see that the decline in mean C value in the restoration treatments was not driven by the absence of species with high C value. Instead, we find that many of the species that appeared in the burn and masticate treatments had C values of 4 or less. These species included aggressive invasive species such as miniature beefsteak (*Mosla dianthera;* C = 0) and Japanese honeysuckle (*Lonicera japonica;* C = 0), which could potentially pose management issues in the future. As of 2019, frequency and percent cover of these invasive species was far lower than Japanese stiltgrass, but proactive management to prevent the ongoing spread is recommended. Other low C species included common ruderals that have been found to be highly correlated with disturbance such as prescribed fire including American burnweed (C = 2), and rabbit-tobacco (C = 2). The increase in several ruderal species, which are typically the first species to colonize following disturbance, is not necessarily concerning, and in the case that these pioneer species are not aggressive invaders, should be considered a normal early seral stage of a woodland restoration. Additionally, an oak-hickory woodland restoration in north-central Mississippi found that an initial increase in native ruderals diminished with time (Brewer et al., 2015). Importantly, low C value additions also included native species like Canada goldenrod (*Solidago canadensis*; C = 1) and prairie dog-bane (*Apocynum cannabinum*; C = 1) which provide valuable nectar and pollen resources to native pollinators.

Several species with high C values appeared in burned and masticated treatments as well. Broad beech fern (*Phegoptera hexaganoptera*; C = 9), rock muhly (*Muhlenbergia sobolifera*; C = 8), cranefly orchid (*Tipularia discolor*; C = 6), lily-leaved twayblade (*Liparis lillifolia*; C = 5), and small green wood orchid (*Platanthera clabellata*; C = 6) were unique to the restoration treatments. The appearance of several high C species outside of the control is encouraging and suggests that the seedbank of these sites might still have remnants from the pre-fire suppression era.

Although these results suggest that restoration activities increase floristic quality, the floristic quality index has several important limitations. The same FQI value can be calculated with different combinations of species richness and mean C so care should be taken when comparing plant communities based solely on floristic quality index. Additionally, the subjective nature of the C value assignment has garnered some criticism, especially when applied over a large geographic extent (Spyreas, 2019). Lastly, in this formulation for floristic quality, no component of abundance is incorporated. This could result in misleading results if few species comprise a majority of the cover in a habitat. However, in an oak woodland restoration in Missouri, the floristic quality index was found to be effective at quantitatively differentiating burned and unburned habitats, making the index a valuable monitoring tool (Maginel et al., 2016).

Management implications

Results from this oak woodland restoration project suggest several important management implications for the ongoing conservation of oak woodlands in the Central Hardwoods and Southern Appalachian regions. Reduction in canopy cover and prescribed fire were both important to establish woodland indicator species and to increase cover of native grasses. Changes to stand structure were a result of multiple interacting disturbances, including an ice-storm in 2003 and a subsequent salvage harvest (Black et al., 2019), underscoring the importance of historic disturbance regimes in the creation of these more open structured habitats. While the initial salvage harvest and the subsequent midstory mastication did significantly reduce canopy cover (Black et al., 2019), results from this study indicated that further reduction of canopy cover might be needed to facilitate increased cover of native grasses. Results from this study and others in the region show that the combination of mechanical removals and prescribed fire are likely necessary to reach oak woodland restoration goals (Peterson & Reich, 2001; Barefoot et al., 2019; Oakman et al., 2019). Additionally, repeated fire will be necessary to maintain midstory openness, reduce sprout density of hardwoods and woody shrubs, and to continue to encourage cover of fire-adapted forbs and grasses. More specifically, repeated fire with short fire return intervals (~ two years) have been recommended to most effectively combat sprouting while still allowing fine fuel accumulation in the understory (Taft, 1997; Brewer et al., 2015; Oakman et al., 2019). Expanding the burn window to include late growing season burns is also a management option frequently discussed to increase fire intensity and more effectively reduce small stem density (Brewer et al., 2015; Vander Yacht et al., 2017; Oakman et al., 2019).

However, any management effort that is not preceded by Japanese stiltgrass control will only facilitate the continued dominance of the invasive grass. A study comparing non-chemical and chemical control methods for Japanese stiltgrass found that while all treatments reduced cover and seed production compared to an untreated control, the most effective treatments included herbicide application (Ward & Mervosh, 2017). Repeated chemical treatments are required to deplete the seed bank of Japanese stiltgrass, which maintains viability for at least three years in the absence of new introductions from adjacent sites (Judge et al., 2008). Additionally, a significant interaction of soil moisture and fire has been shown to dramatically increase invasibility of Japanese stiltgrass, leading to recommendations to preferentially burn on drier sites where this additive advantage is not seen (Wagner & Fraterrigo, 2015). Where invasive species are present, proactive control efforts are imperative for the successful restoration of the understory layer characteristic of oak woodlands.

APPENDICES

APPENDIX 1: A2 Supplemental information for Chapter 2

Table A2-1 Complete model set for mortality of stems ≥ 10 cm DBH during the fire-free interval (2010-2018) in the Daniel Boone National Forest, KY. All subsets of the global model were ranked using AIC (Aikake's Information Criterion). Variables included in the global model include a composite burn index (CBI), heat load index (HLI), landscape position (LP), and burn treatment (Trt). All models with Likelihood values of > 0.125 or models with delta AIC of < 4 were considered for analysis. Top models are indicated by an * by the model number.

Model	CBI	HLI	LP	Trt	df	ΔAIC	AICWt	Likelihood
10*	0.273			+	5	0	0.272	1
2	0.183				3	1.503	0.129	0.471
12	0.269	0.058		+	6	1.510	0.128	0.469
14	0.264		-0.024	+	6	1.964	0.102	0.374
4	0.184	0.060		+	6	2.962	0.062	0.227
6	0.170		-0.045	+	6	3.233	0.054	0.199
1					2	3.493	0.048	0.174
16	0.262	0.056	-0.017	+	7	3.503	0.047	0.173
9				+	4	4.459	0.029	0.108
5			-0.083		3	4.516	0.029	0.104
8	0.172	0.057	-0.039		5	4.770	0.025	0.092
3		0.059			3	4.997	0.022	0.082
13			-0.076	+	5	5.621	0.016	0.060
11		0.065		+	5	5.806	0.015	0.054
7		0.053	-0.078		4	6.142	0.012	0.046
15		0.059	-0.069	+	6	7.111	0.008	0.029

Table A2-2 Total basal area (m^2 ha^{-1}) for all stems ≥ 2 cm DBH and total stem density (stems ha^{-1}) for overstory (≥ 20 cm DBH), midstory (10-20 cm DBH) and saplings (2-10 cm DBH) in the Daniel Boone National Forest. Density for species dominant species groups reported separately. Values are represented as mean (standard error).The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011

Total Basal Area (m2 ha^{-1})		**2002**	**2010**	**2018**
	Fire Excluded	25.6(2.4)	29.6(1.6)	30.3(1.7)
	Less Frequent	24.8(1.3)	20.1(1.6)	21.8(1.5)
	Frequent	25.4(1.8)	27(1.8)	26.7(2)
Overstory stem density (stems ha^{-1})				
Total	Fire Excluded	205(19.8)	219(13.7)	219(13.8)
	Less Frequent	180(10.1)	170(12.3)	177(11.9)
	Frequent	184(13.7)	197(16.2)	194(18)
Oak	Fire Excluded	123(18)	139(14)	148(15)
	Less Frequent	118(11)	114(11)	110(10)
	Frequent	121(12)	145(16)	136(17)
Maple	Fire Excluded	41(11)	58(12)	58(11)
	Less Frequent	40(6)	40(5)	43(6)
	Frequent	38(8)	42(7)	64(13)
Midstory stem density (stems ha^{-1})				
Total	Fire Excluded	228(15.7)	212(15.9)	191(16.8)
	Less Frequent	239(26.1)	147(18.5)	138(19.6)
	Frequent	224(20.8)	121(16.7)	117(18.6)
Oak	Fire Excluded	80(16)	79(19)	73(19)
	Less Frequent	53(7)	46(10)	42(6)
	Frequent	76(13)	42(7)	50(16)
Maple	Fire Excluded	100(16)	116(17)	104(15)
	Less Frequent	143(13)	86(12)	91(18)
	Frequent	102(22)	98(16)	85(13)
Yellow-Poplar	Fire Excluded	42(17)	25(0)	25(0)
	Less Frequent	25(0)	75(NA)	156(75)
	Frequent	70(39)	25(NA)	25(0)

Appendix Table A2-2 continued				
Sapling stem density (stems ha^{-1})				
Total	Fire Excluded	840(92.9)	673(72.5)	700(71)
	Less Frequent	912(105)	278(72.5)	1617(289)
	Frequent	813(100)	260(79.8)	843(190)
Oak	Fire Excluded	100(0)	0	0
	Less Frequent	183(48)	0	375(166)
	Frequent	133(33)	0	475(131)
Maple	Fire Excluded	533(98)	470(85)	382(62)
	Less Frequent	643(130)	279(91)	480(68)
	Frequent	565(113)	300(131)	286(69)
Yellow-Poplar	Fire Excluded	0	0	0
	Less Frequent	100(NA)	0	1091(259)
	Frequent	100(NA)	0	100(NA)
Sassafras	Fire Excluded	100(NA)	0	600(NA)
	Less Frequent	100(0)	200(NA)	1029(480)
	Frequent	100(0)	0	1900(964)
Sourwood	Fire Excluded	140(24)	100(0)	188(30)
	Less Frequent	225(95)	300(NA)	682(169)
	Frequent	129(18)	100(0)	400(98)

Table A2-3 Importance values of oak (*Quercus*) and maple (*Acer*) species in the midstory (10-20 cm DBH) within each landscape position in the Daniel Boone National Forest, Kentucky. Values represent the sum of the relative frequency, abundance and dominance of the species group and can range from 0 to 300. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011.

Species	Treatment	Landscape Position	2002	2010	2018
Oak	Fire Excluded	Sub-mesic	59.6	60.9	53.2
		Intermediate	71.4	48.4	43.2
		Sub-xeric	81.5	80.4	53.7
	Less Frequent	Sub-mesic	31.6	27	32.7
		Intermediate	66.8	62.9	53.6
		Sub-xeric	70.9	116.1	129.4
	Frequent	Sub-mesic	33.8	26.2	14.1
		Intermediate	66.8	58.8	32.7
		Sub-xeric	80.3	92.3	99.8
Maple	Fire Excluded	Sub-mesic	101.1	111.6	142.2
		Intermediate	101.1	136.2	149.1
		Sub-xeric	116.5	116.8	106.1
	Less Frequent	Sub-mesic	127.8	149.2	116.9
		Intermediate	96.5	105.4	82.2
		Sub-xeric	111.6	82.7	61.9
	Frequent	Sub-mesic	132.1	155.4	168.4
		Intermediate	86.8	94.1	87.2
		Sub-xeric	132.1	57.5	67.3

Table A2-4 Model selection table for oak (*Quercus*) sapling recruitment in the Daniel Boone National Forest, Kentucky. All subsets of the global model were ranked using AICc (Aikake's Information Criterion). Variables included in the top model are basal area (BA), heat load index (HLI), landscape position (LP), composite burn index (CBI), and burn treatment (Trt). All models with Likelihood values of >0.125 or models with delta AICc of <4 were included in the analysis. Top models are indicated by an * by the model number. are basal area (BA), heat load index (HLI), landscape position (LP), composite burn index (CBI), and burn treatment (Trt). All models with Likelihood values of >0.125 or models with delta AICc of <4 were included in the analysis. Top models are indicated by an * by the model number. ***Global Model: glmmTMB(Oak sapling recruitment ~ Trt + BA + HLI + MI + CBI + (1|SiteTrt), family= nbinom1)***

Model	BA	HLI	LP	CBI	Trt	df	ΔAICc	Likelihood	AICcWt
1*	-2.33	0.64				5	0	1	0.19
2*	-2.74		-0.65			5	0.109	0.947	0.18
3	-2.72	0.55	-0.54			6	0.242	0.886	0.168
4	-2.26					4	1.592	0.451	0.086
5	-2.31	0.61			+	6	2.425	0.297	0.056
6	-2.69		-0.63		+	6	2.498	0.287	0.054
7	-2.68		-0.64	0.09		6	2.499	0.287	0.054
8	-2.76	0.57	-0.54	-0.06		7	2.757	0.252	0.048
9	-2.7	0.55	-0.54		+	7	2.783	0.249	0.047
10*	-2.25				+	5	3.048	0.218	0.041
11*	-2.11			0.24		5	3.52	0.172	0.033
12	-2.63		-0.62	0.08	+	7	4.989	0.083	0.016
13	-2.12			0.16	+	6	5.309	0.07	0.013
14	-2.76	0.57	-0.53	-0.07	+	8	5.387	0.068	0.013
15				0.97	+	5	20.273	0	0
16		0.46		0.86	+	6	20.884	0	0
17				0.97		4	21.24	0	0
18			-0.18	0.91	+	6	22.498	0	0
19		0.45	-0.15	0.83	+	7	23.254	0	0
20		0.53	-0.08	0.87		6	23.452	0	0
21		0.62			+	5	24.96	0	0
22						3	25.825	0	0
23		0.59	-0.34			5	25.95	0	0
24		0.55	-0.41		+	6	26.051	0	0
25			-0.47			4	26.332	0	0
26			-0.49		+	5	26.403	0	0

Table A2-5 Model selection table for non-oak sapling recruitment in the Daniel Boone National Forest, KY. All subsets of the global model were ranked using AICc (Aikake's Information Criterion). Variables included in the global model are basal area (BA), composite burn index (CBI), heat load index (HLI), landscape position (LP), and burn treatment (Trt). All models with Likelihood values of >0.125 or models with delta AICc of <4 were included in the analysis. Top models are indicated by an * by the model number. ***Global Model: glmmTMB(Sapling recruitment ~ Trt + BA + HLI + MI + (1|SiteTrt), family= nbinom1)***

Model	BA	CBI	HLI	MI	Trt	df	ΔAICc	AICcWt	Likelihood
1*	-0.32	0.48		-0.48	+	8	0	0.331	1
2		0.61		-0.52	+	7	1.035	0.197	0.596
3	-0.32	0.45	0.11	-0.48	+	9	1.741	0.139	0.419
4	-0.4	0.33		-0.5		6	2.526	0.094	0.283
5*		0.59	0.11	-0.52	+	8	2.729	0.085	0.256
6	-0.42	0.33	0.12	-0.48		7	4.103	0.043	0.129
7	-0.5			-0.55		5	5.057	0.026	0.08
8	-0.47			-0.53	+	7	5.199	0.025	0.074
9		0.44		-0.51		5	5.654	0.02	0.059
10	-0.48		0.16	-0.51	+	8	6.336	0.014	0.042
11	-0.51		0.1	-0.54		6	6.81	0.011	0.033
12		0.44	0.09	-0.5		6	7.468	0.008	0.024
13	-0.35	0.57			+	7	10.293	0.002	0.006
14				-0.61	+	6	10.378	0.002	0.006
15				-0.6		4	11.134	0.001	0.004
16			0.14	-0.6	+	7	11.806	0.001	0.003
17	-0.35	0.53	0.11		+	8	12.021	0.001	0.002
18		0.75			+	6	12.46	0.001	0.002
19			0.08	-0.59		5	13.068	0	0.001
20	-0.36	0.43				5	13.277	0	0.001
21		0.71	0.12		+	7	14.037	0	0.001
22	-0.39	0.42	0.15			6	14.444	0	0.001
23		0.54				4	15.306	0	0
24		0.54	0.11			5	16.89	0	0
25	-0.51				+	6	17.568	0	0
26	-0.48					4	17.909	0	0
27	-0.53		0.16		+	7	18.546	0	0
28	-0.5		0.13			5	19.23	0	0
29						3	23.576	0	0
30					+	5	23.81	0	0
31			0.14		+	6	25.051	0	0
32			0.11			4	25.062	0	0

Table A2-6 Importance values of maple (*Acer*) species, oak (*Quercus*), sourwood (*Oxydendrum arboreum*), sassafras (*Sassafras albidum*), and yellow-poplar (*Liriodendron tulipifera*) saplings (2-10 cm DBH) within each landscape position in the Daniel Boone National Forest, Kentucky. Values represent the sum of the relative frequency, abundance and dominance of the species group and can range from 0 to 300. The Less Frequent treatment was burned in 2003 and 2009. The Frequent treatment was burned in 2003, 2004, 2006, 2008 and 2011.

Species	Treatment	LP	2010	2018
Maple	Fire Excluded	Sub-mesic	87.4	92.2
		Intermediate	146.7	127.4
		Sub-xeric	109.4	108.5
	Less Frequent	Sub-mesic	204	132.6
		Intermediate	176.8	59.6
		Sub-xeric	104.7	80.2
	Frequent	Sub-mesic	245.9	273.2
		Intermediate	74.6	37.6
		Sub-xeric	0	70.5
Oak	Fire Excluded	Sub-mesic	0	0
		Intermediate	0	0
		Sub-xeric	0	0
	Less Frequent	Sub-mesic	0	0
		Intermediate	0	29.4
		Sub-xeric	0	15.8
	Frequent	Sub-mesic	0	0
		Intermediate	0	41.1
		Sub-xeric	0	34.3
Sourwood	Fire Excluded	Sub-mesic	0	0
		Intermediate	17.3	25.8
		Sub-xeric	13.6	41.5
	Less Frequent	Sub-mesic	0	0
		Intermediate	13.9	31.2
		Sub-xeric	0	87
	Frequent	Sub-mesic	0	0
		Intermediate	0	65
		Sub-xeric	0	36

Appendix Table A2-6 continued				
Sassafras	Fire Excluded	Sub-mesic	0	0
		Intermediate	0	11.3
		Sub-xeric	0	0
	Less Frequent	Sub-mesic	0	0
		Intermediate	10.5	40.8
		Sub-xeric	0	7.5
	Frequent	Sub-mesic	0	0
		Intermediate	0	77.4
		Sub-xeric	0	8.2
Yellow-poplar	Fire Excluded	Sub-mesic	0	0
		Intermediate	0	0
		Sub-xeric	0	0
	Less Frequent	Sub-mesic	0	19.1
		Intermediate	0	65
		Sub-xeric	0	36
	Frequent	Sub-mesic	0	0
		Intermediate	0	4.1
		Sub-xeric	0	0

APPENDIX 2: Supplemental information for Chapter 3

Table A3-1 P-values for percent change in relative stem density (stems ha^{-1}) of small seedlings (< 0.6 m, < 2 ft in height) large seedlings (≥ 0.6 m, ≥ 2 ft in height) from 2017 to 2019 in the Daniel Boone National Forest. C1–C are control plots that remained unburned, C2–B are the subset of control plots burned in 2018, M1–M are masticated plots unburned, and M2–MB are the subset of plots that were masticated and burned. SE given in parenthesis.

Species	Seedling Size	Contrast			
		C1 – C	C2 – B	M1 – M	M2 - MB
Oak	Small	0.99	1.00	0.94	0.73
	Large	1.00	0.98	0.74	0.84
Maple	Small	0.46	0.48	0.95	0.92
	Large	0.99	1.00	0.95	0.98
Yellow-poplar	Small	1.00	1.00	0.99	0.94
	Large	0.67	1.00	0.76	1.00
Sassafras	Small	0.96	1.00	1.00	0.98
	Large	0.98	1.00	0.91	0.97
Blackgum	Small	0.98	1.00	0.07	0.75
	Large	0.75	0.80	0.98	0.94
Hickory	Small	1.00	1.00	1.00	1.00
	Large	1.00	1.00	0.28	0.66

Table A3-2 Dominant species (% of total cover within that lifeform and treatment) of forbs, vines, shrubs, and graminoids in each treatment in 2019. All plots located in the Daniel Boone National Forest, KY. Species were listed until > 70% of cover was reached within each treatment for forbs, vines and shrubs. Due to the dominance of Microstegium vimineum in the graminoid cover, species were listed until they fell below 1% of total cover.

Lifeform	Control	Burn	Masticate	Masticate-Burn
Forbs	*Desmodium nudiflorum* 45%	*Thelypteris noveboracensis* 26.3%	*Amphicarpaea bracteata* 21.3%	*Potentilla* spp. 23.9%
	Polygonatum biflorum 13.5%	*Amphicarpaea bracteata* 13.5%	*Eupatorium serotinum* 12.6%	*Eupatorium serotinum* 11.6%
	Toxicodendron radicans 8%	*Desmodium nudiflorum* 10%	*Desmodium nudiflorum* 11.4%	*Lysimachia quadrifolia* 9.5%
	Boehmeria cylindrica 6.7%	*Dioscorea villosa* 6.9%	*Mosla dianthera** 8.3%	*Pseudognapthalium obtusifolium* 7.8%
	-	*Potentilla* spp. 4.8%	*Potentilla* spp. 8.3%	*Mosla dianthera** 7.1%
	-	*Toxicodendron radicans* 4.6%	*Actaea racemosa* 6%	*Verbena urticifolia* 4.3%
	-	*Ageratina altissima* 4%	*Lysimachia quadrifolia* 3.7%	*Erechtites hieraciifolius* 4%
	-	-	-	*Bidens frondosa* 3.2%
Vines	*Smilax rotundifoliia* 87.2%	*Smilax rotundifoliia* 90.4%	*Smilax rotundifoliia* 85%	*Smilax rotundifoliia* 76.6%
	Smilax glauca 11%	*Smilax glauca* 4%	*Parthenocissus quinquefolia* 6.2%	*Smilax glauca* 9.1%
Shrubs	*Vaccinium pallidum* 59%	*Rubus argutus* 47.4%	*Rubus argutus* 41.3%	*Rubus argutus* 58.9%
	Vaccinium stamineum 19.5%	*Vaccinium stamineum* 26.5%	*Rubus allegheniensis* 24.9%	*Rubus allegheniensis* 18.8%
	-	-	*Vaccinium stamineum* 13.7%	-

Appendix Table A3-2 Continued

Lifeform	Control	Burn	Masticate	Masticate-Burn
Graminoids	*Microstegium vimineum** 89%	*Microstegium vimineum** 64.3%	*Microstegium vimineum** 85.5%	*Microstegium vimineum** 65.5%
	Dichanthelium dichotomum 5.5%	*Andropogon virginicus* 9.8%	*Andropogon virginicus* 6.7%	*Andropogon virginicus* 23.3%
	Andropogon virginicus 3.3%	*Carex* spp. 8.2%	*Diarrhena americana* 3.4%	*Carex* spp. 3.5%
	Carex spp. 1%	*Dichanthelium commutatum* 5.7%	*Carex spp. 1.5%*	*Dichanthelium boscii 3%*
	-	*Brachyelytrum erectum* 3.9%	*Dichanthelium commutatum* 1.5%	*Dichanthelium commutatum* 2.5%
	-	*Dichanthelium boscii* 3.1 %	-	*Dichanthelium dichotomum* 1.6%
	-	*Diarrhena americana* 2.7%	-	-
	-	*Leersia virginica* 1.2%	-	-
	-	*Dichanthelium dichotomum* 1%	-	-

Table A3-3 Species list for each treatment in 2019 in the Daniel Boone National Forest. Scientific and common name given for each plant identified to species, and the assigned C value. * indicates invasive species.

Scientific name	Common Name	C value	Control	Burn	Masticate	Masticate-Burn
Acalypha virginica	Virginia mercury	0				X
Acer rubrum	Red maple	2	X	X	X	X
Acer saccharum	Sugar maple	5		X		
Actaea racemosa	Black cohosh	7			X	
Ageratina altissima	White snakeroot	3		X	X	X
Agrostis gigantea	Redtop	0				X
*Ailanthus altissima**	Tree-of-Heaven*	0	X		X	
Ambrosia artemisiifolia	Annual ragweed	0		X		
Amelanchier arborea	Downy serviceberry	5	X	X	X	X
Amphicarpaea bracteata	Hog peanut	4		X	X	X
Andropogon virginicus	Broomsedge bluestem	3	X	X	X	X
Apocynum cannabinum	Prairie dogbane	1			X	X
Aquilegia canadensis	Eastern red columbine	6		X		
Aralia spinosa	Devil's walking stick	5	X	X		
Arisaema triphyllum	Jack in the pulpit	3	X		X	
Aristolochia serpentaria	Virginia snakeroot	7		X	X	X
Asclepias quadrifolia	Fourleaf milkweed	6		X		
Asimina triloba	Pawpaw	6	X	X	X	X
Asplenium platyneuron	Ebony spleenwort	3	X	X		
Aureolaria virginica	Downy yellow false foxglove	8			X	X
Bidens frondosa	Devil's beggartick	2	X	X	X	X
Boehmeria cylindrica	Smallspike false nettle	4	X	X	X	X
Botrychium biternatum	Southern grape fern	4		X		
Botrychium obliquum	Cutleaf grape fern	4		X		
Brachyelytrum erectum	Bearded shorthusk	5		X	X	X
Carex pensylvanica	Pennsylvania sedge	3	X	X	X	X
Carpinus caroliniana	American hornbeam	5			X	
Carya cordiformis	Bitternut hickory	5			X	
Carya glabra	Pignut hickory	5	X	X	X	X
Carya ovalis	Red hickory	5		X		
Carya ovata	Shagbark hickory	6		X		
Carya tomentosa	Mockernut hickory	6	X	X	X	X
Castanea dentata	American chestnut	6	X		X	X
Cercis canadensis	Eastern redbud	3		X	X	
Chamaecrista nictitans	Sensitive partridge pea	4		X	X	X
Circaea lutetiana	Enchanter's nightshade	3		X	X	
Claytonia virginica	Virginia springbeauty	2	X			
Clitoria mariana	Butterfly pea	6		X		
Collinsonia canadensis	Horsebalm	5				X
Conoclinium coelestinum	Mistflower	3		X		X
Convolvulus arvensis	Field bindweed	0				X
Conyza canadensis	Canadian horseweed	0	X	X	X	X

Appendix Table A3-3 Continued						
Scientific name	**Common Name**	**C value**	**Control**	**Burn**	**Masticate**	**Masticate-Burn**
Coreopsis major	Greater tickseed	7	X	X	X	X
Cornus florida	Flowering dogwood	5	X	X	X	X
Cunila origanoides	Common dittany	6	X	X		X
Danthonia spicata	Poverty oatgrass	4		X	X	X
Dentaria diphylla	Two-leaved toothwort	4		X		
Desmodium canescens	Hoary ticktrefoil	4		X		
Desmodium glutinosum	Pointedleaf ticktrefoil	5		X	X	
Desmodium nudiflorum	Naked-Flowered ticktrefoil	5	X	X	X	X
Desmodium paniculatum	Panicledleaf ticktrefoil	3		X	X	X
Desmodium rotundifolium	Prostrate ticktrefoil	6	X	X	X	X
Diarrhena americana	American beakgrain	7		X	X	
Dichanthelium commutatum	Variable panicgrass	5	X	X	X	X
Dichanthelium dichotomum	Cypress panicgrass	4	X	X	X	X
Dichanthelium boscii	Bosc's panicgrass	6	X	X	X	X
*Dioscorea polystachya**	Chinese yam*	0		X	X	X
Dioscorea villosa	Wild yam	4	X	X	X	X
Diphasiastrum digitatum	Southern ground-cedar	1			X	
Doellingeria infirma	Appalachian white-aster	8				X
Elephantopus carolinianus	Carolina elephantsfoot	4		X		
Erechtites hieraciifolius	American burnweed	2		X	X	X
Erigeron philadelphicus	Philadelphia fleabane	2	X	X		X
Eupatorium altissimum	Tall boneset	0			X	
Eupatorium perfoliatum	Common boneset	3		X	X	X
Eupatorium serotinum	Late boneset	2	X	X	X	X
Eupatorium sessilifolium	Upland boneset	4		X		
Euphorbia corollata	Flowering spurge	4		X	X	
Eurybia divaricata	White wood aster	5	X	X		
Eurybia macrophylla	Large-leaved aster	10		X		
Euthamia graminifolia	Flat-top goldentop	2		X	X	X
Fagus grandifolia	American beech	7	X	X	X	
Frangula caroliniana	Carolina buckthorn	4		X	X	
Fraxinus americana	White ash	6	X	X	X	
Fraxinus pennsylvanica	Green ash	3		X	X	
Galium circaezans	Wild Licorice	4	X	X	X	X
Gamochaeta purpurea	Spoonleaf purple everlasting	3		X	X	
Gaultheria procumbens	Eastern teaberry	5	X	X	X	
Gaylussacia baccata	Black huckleberry	6	X	X	X	
Geranium maculatum	Spotted geranium	4		X	X	
Geum canadense	White avens	2		X		
Goodyera pubescens	Downy rattlesnake plantain	6		X	X	X
Hackelia virginiana	Virginia stickseed	2		X		
Hamamelis virginiana	Witch-hazel	5		X		
Helianthus divaricatus	Woodland sunflower	4	X	X	X	X
Helianthus microcephalus	Small woodland sunflower	5	X	X	X	X
Hieracium gronovii	Gronovius' Hawkweed	5		X	X	X

Appendix Table A3-3 Continued

Scientific name	Common Name	C value	Control	Burn	Masticate	Masticate-Burn
Houstonia caerulea	Azure bluet	3		X		
Houstonia purpurea	Venus' pride	5			X	X
Hypericum mutilum	Dwarf St. John's wort	3				X
Hypericum stragulum	St. Andrew's cross	6	X	X	X	X
Ilex opaca	American holly	6	X			
Impatiens capensis	Jewelweed	2	X	X	X	
Iris cristata	Dwarf crested iris	5			X	X
Isotria medeoloides	Small whorled pogonia	7	X			X
Juncus effusus	Common rush	1				X
Juniperus virginiana	Eastern red cedar	3	X			
Krigia virginica	Virginia dwarfdandelion	5		X		X
Lactuca biennis	Tall blue lettuce	1			X	
Lactuca canadensis	Lactuca canadensis	1		X		
Leersia virginica	Cutgrass	4		X	X	X
*Lespedeza cuneata**	Sericea lespedeza*	0		X		X
Lespedeza hirta	Hairy lespedeza	5				X
Lespedeza intermedia	Violet lespedeza	3	X	X	X	X
Lespedeza repens	Creeping lespedeza	6		X		X
Lindera benzoin	Northern spicebush	5	X		X	
Liparis liliifolia	Lily-leaved twayblade	5		X		
Liriodendron tulipifera	Yellow-poplar	6	X	X	X	X
Lobelia inflata	Indian-tobacco	1	X	X	X	X
Lobelia puberula	Downy lobelia	5		X	X	
Lobelia siphilitica	Great blue lobelia	3		X		
Lonicera japonica	Japanese honeysuckle	0		X		
Ludwigia alternifolia	Seedbox	3	X	X	X	X
Lycopus americanus	American bugleweed	3		X		X
Lygodium palmatum	American climbing fern	5				X
Lysimachia quadrifolia	Whorled yellow loosestrife	5	X	X	X	X
Magnolia tripetala	Umbrella magnolia	8	X	X	X	
Maianthemum canadense	Canada mayflower	6	X	X		
Maianthemum racemosum	False solomon's seal	4	X	X	X	X
Medeola virginiana	Indian cucumber	6				X
Menispermum canadense	Common moonseed	5	X	X		X
*Microstegium vimineum**	Japanese stiltgrass*	0	X	X	X	X
Mitchella repens	Partridgeberry	5		X		
Morus rubra	Red mulberry	7			X	
Mosla dianthera	Miniature beefsteak	0		X	X	X
Muhlenbergia sobolifera	Rock muhly	8		X		
Nyssa sylvatica	Blackgum	7	X	X	X	X
Obolaria virginica	Virginia pennywort	7		X		X
Osmorhiza claytonii	Sweet cicely	4		X		
Ostrya virginiana	Hophornbeam	5			X	
Oxydendrum arboreum	Sourwood	7	X	X	X	X
Panicum anceps	Beaked panicgrass	3		X		X

Appendix Table A3-3 Continued						
Scientific name	**Common Name**	**C value**	**Control**	**Burn**	**Masticate**	**Masticate-Burn**
Parthenocissus quinquefolia	Virginia creeper	2	X	X	X	X
Phegopteris hexagonoptera	Broad beechfern	9			X	
Pinus rigida	Pitch pine	7	X	X	X	X
Plantago major	Common plantain	0			X	
Platanthera clavellata	Small green wood orchid	6			X	
Platanus occidentalis	American sycamore	7				X
Poa annua	Annual bluegrass	0		X		
Polygonatum biflorum	Smooth solomon's seal	4	X	X	X	X
Polygonum persicaria	Spotted ladysthumb	0				
Polystichum acrostichoides	Christmas fern	3	X	X	X	
Porteranthus stipulates	American ipecac	6				X
Potentilla spp.	Cinquefoil	3	X	X	X	X
Prenanthes altissima	Tall white lettuce	4	X	X	X	X
Prenanthes serpentaria	Lion's foot	5			X	
Prosartes lanuginose	Yellow fairybells	7			X	
Prunella vulgaris	Common selfheal	0		X		
Prunus serotina	Black cherry	3	X	X	X	X
Pseudognaphalium obtusifolium	Rabbit-tobacco	2		X	X	X
Pycnanthemum incanum	Hoary mountain mint	6		X		
Pycnanthemum tenuifolium	Slender mountain mint	4		X		
Quercus alba	White oak	6	X	X	X	X
Quercus coccinea	Scarlet oak	6	X	X	X	X
Quercus montana	Chestnut oak	7	X	X	X	X
Quercus muhlenbergii	Chinkapin oak	7	X	X	X	X
Quercus rubra	Northern red oak	6	X	X	X	
Quercus velutina	Black oak	7	X	X	X	X
Rhus copallinum	Winged sumac	4		X	X	X
Rhus glabra	Smooth sumac	2		X		X
Robinia pseudoacacia	Black locust	0	X	X	X	X
Rosa Carolina	Carolina rose	4	X	X	X	X
*Rosa multiflora**	Multiflora rose*	0	X	X	X	X
Rubus allegheniensis	Allegheny blackberry	1	X	X	X	X
Rubus argutus	Sawtooth blackberry	1	X	X	X	X
Rubus flagellaris	Northern dewberry	1	X	X	X	X
Rubus occidentalis	Black raspberry	1	X	X	X	X
*Rubus phoenicolasius**	Wine raspberry*	0	X	X	X	X
Sabatia angularis	Common rose pink	4				X
Sambucus nigra	Black elderberry	2			X	
Sanguinaria canadensis	Bloodroot	5		X	X	
Sanicula canadensis	Canadian blacksnakeroot	3	X	X	X	X
Sanicula smallii	Small's blacksnakeroot	10			X	
Sassafras albidum	Sassafras	3	X	X	X	X
Scirpus cyperinus	Woolgrass	1			X	X
Scutellaria elliptica	Hairy skullcap	5			X	
Smilax bona-nox	Saw greenbrier	3	X	X	X	X

Appendix Table A3-3 Continued						
Scientific name	**Common Name**	**C value**	**Control**	**Burn**	**Masticate**	**Masticate-Burn**
Smilax glauca	Cat greenbrier	5	X	X	X	X
Smilax herbacea	Smooth carrionflower	4		X		X
Smilax rotundifolia	Roundleaf greenbrier	4	X	X	X	X
Solanum carolinense	Horse nettle	0		X	X	X
Solidago caesia	Wreath goldenrod	5			X	
Solidago canadensis	Canada goldenrod	1		X	X	X
Solidago flexicaulis	Zigzag goldenrod	5		X	X	
Solidago gigantea	Giant goldenrod	3			X	
Solidago hispida	Hairy goldenrod	4		X		
Solidago nemoralis	Gray goldenrod	2				X
Symphyotrichum vimineus	Small white aster	3		X	X	X
Taraxacum officinale	Common dandelion	0			X	
Thalictrum thalictroides	Rue anenome	4			X	
Thelypteris noveboracensis	New york fern	4	X	X	X	X
Thelypteris palustris	Eastern marsh fern	6			X	X
Tipularia discolor	Crane-fly orchid	6		X		
Toxicodendron radicans	Poison ivy	1	X	X	X	
Ulmus rubra	Slippery elm	3		X	X	
Uvularia perfoliata	Perfoliate bellwort	5	X	X	X	X
Uvularia sessilifolia	Spreading bellwort	5	X	X	X	
Vaccinium corymbosum	Highbush blueberry	6		X		
Vaccinium pallidum	Hillside blueberry	6	X	X	X	X
Vaccinium stamineum	Deerberry	6	X	X	X	X
Verbena gigantea	Tall verbena	2		X		
Verbena urticifolia	White vervain	3	X	X	X	X
Viburnum acerfolium	Maple-leaved viburnum	6	X	X	X	X
Viburnum dentatum	Southern arrowwood	2	X	X	X	X
Viola hirsutula	Southern woodland violet	6		X	X	X
Viola pubescens	Downy yellow violet	4		X	X	
Viola sagittata	Arrowleaf violet	4	X	X	X	X
Viola sororia	Missouri violet	1	X	X	X	X
Vitis aestivalis	Summer grape	4	X	X	X	X
Vitis vulpina	Frost grape	3	X	X	X	X

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