EFFECTS OF FIRE AND THINNING ON KANSAS OAK WOODLANDS

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and hereby certify that, in their opinion, it is worthy of acceptance.

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DEDICATION

I dedicate the following in loving memory to my dad.
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ABSTRACT

Prior to Euro-American settlement, a mosaic of prairie, savanna, woodland, and forest existed within the Forest-Prairie Transition Region of the United States, with anthropogenic fire acting as an important driver in the perpetuation of open-oak communities. As fire suppression became a regular practice throughout the 20th century, these historically open communities became threatened by encroaching fire-sensitive, and often shade-tolerant, species. This study evaluated the effects of prescribed fire and thinning treatments as methods to achieve woodland restoration objectives, which commonly include reducing stand density, reducing mesophytic oak-competitors, increasing canopy openness, increasing herbaceous plant cover, and promoting the regeneration of oak. We investigated the effects of six treatment types on the structure and composition of a Kansas oak woodland. Treatments included: prescribed fire (burn), thin to 60 ft²/acre basal area (T60), thin to 30 ft²/acre basal area (T30), the combination of fire and thin to 60 ft²/acre basal area (BT60), the combination of fire and thin to 30 ft²/acre basal area (BT30), and an untreated control. Additionally, we examined the effect of fire on advance regeneration survival probability for five tree species: chinkapin oak (Quercus muehlenbergii), black oak (Q. velutina), bitternut hickory (Carya cordiformis), sugar maple (Acer saccharum), and eastern redcedar (Juniperus virginiana).

Following a single girdle and herbicide application thinning treatment, we found low mortality in the first year, especially for sugar maple. As a result, the reduction in overstory basal area did not meet our intended targets. A single dormant season prescribed burn was effective at reducing large and small seedling densities of sugar maple and other oak-competitors, and increased forb and legume cover in the understory.
However, the burn only treatment had no effect on overstory stand metrics, including basal area, tree density, percent stocking, and canopy openness. Thinning of the overstory and midstory in combination with prescribed fire resulted in similar effects to seedling densities and ground flora cover as the burn only treatment, but also created reduced tree density in the sapling layer and greater canopy openness. Additionally, the effect of the burn only treatment on advance regeneration revealed that significant relationships exist between pretreatment stem basal diameter and height and the probability of surviving a single fire for some of the species. These initial results are for the first year following treatments and over time we expect vegetation dynamics to continue to respond to treatments.
Chapter I: LITERATURE REVIEW

Oaks in the eastern United States

Oak (*Quercus*) species have dominated the eastern United States for thousands of years (Johnson et al. 2009, Hanberry and Nowacki 2016). Warmer and drier climatic periods since the beginning of the Holocene epoch coincide with oak domination, and by 10,000 years B.P., oak forests occurred throughout most of the eastern US (Abrams 1992). Currently within the eastern US, the distribution of oaks extends from the Humid Tropical Domain in southern Florida to the Humid Temperate Domain, reaching into southern Canada (Bailey 1995). Approximately 40 oak species and varieties occur east of the 100th meridian, a longitudinal marker that separates eastern and western oak species (Little 1979, Johnson et al. 2009). The genus is found across a gradient of topoedaphic conditions including rich, hydric floodplains, productive mesic coves, xeric uplands, and mountain ridgetops (Johnson et al. 2009). Historical tree surveys reveal that prior to Euro-American settlement in the eastern-central US, oaks represented the prototypical foundation genus, comprising 30-80% of presettlement forests (Abrams 2003, Nowacki and Abrams 2008, Hanberry and Nowacki 2016). Within the Forest-Prairie Transition Region, oak savannas and woodlands covered 27-32 million acres immediately prior to 1840 (Nuzzo 1986). Oak abundance historically increased on an east-to-west gradient, with oak composition comprising 66% of the Prairie Division before European settlement (Hanberry and Nowacki 2016).

Currently, oak forest types represent 51% of all forestland in the eastern US, and oak savanna and woodlands have been reduced to 0.02% of their presettlement area (Nuzzo 1986, Smith et al. 2009). Since 1968 in the eastern US, a 7-9% decline in the
proportion (trees ≥12.7 cm dbh) of each northern red oak (*Quercus rubra*), black oak (*Q. velutina*), and white oak (*Q. alba*) has been observed (Hanberry 2013). Forest compositions are shifting toward more mesophytic species across eastern forests and woodlands, with species such as red maple (*Acer rubrum*) and sugar maple (*Acer saccharum*) becoming more abundant (Pallardy et al. 1988, Nowacki and Abrams 2008, Hanberry 2013). In central Missouri, one study found sugar maple sapling density increased even on the most xeric of sites between 1968 and 1982, while oak sapling density decreased on all upland site types (Pallardy et al. 1988). Abrams and Nowacki (2008) have hypothesized that a period of fire suppression in the 20th century has led to the “mesophication” of eastern forests, in which there is a region-wide shift in species composition to shade-tolerant, fire-sensitive species.

**Oak Woodlands**

Oaks in the eastern US exist in a variety of community types, ranging from open savanna to woodland to closed canopy forest. Savannas are defined as plant communities of open-grown trees, widely spaced or in small groves, with an herbaceous, primarily grassy understory (Bray 1955, Nuzzo 1986, Nelson 2010). Savanna canopy cover is generally less than 30 percent (Nelson 2010, Dey et al. 2017, Hanberry et al. 2017). In the central US, they are strongly associated with prairies and primarily occur where upland topography is level or gently rolling (Nelson 2010). Conversely, Nelson (2010) defines a forest as “an area dominated by trees forming a closed canopy and interspersed with multilayered shade-tolerant subcanopy trees, shrubs, vines, ferns, and herbs.” The Dictionary of Forestry gives a similar definition of a forest: “an ecosystem characterized by a more or less dense and extensive tree cover” (Helms 1998).
Woodland communities exist as an intermediary between savanna and forests. Many definitions exist for woodlands and a variety of woodland types have been defined. Nelson (2010) describes eighteen distinct woodland types in Missouri alone, but generally defines them as “highly variable natural communities with a canopy of trees ranging from 30 to 100 percent closure with a sparse understory (or midstory) and a dense ground flora rich in forbs, grasses, and sedges.” In terms of canopy closure, Faber-Langendoen (2001) provides a similar definition, stating that canopy cover in woodlands is not completely closed, occurring between 25-80%. In contrast to forests, woodlands characteristically exhibit small, short-boled trees, relative to their crown depth (Helms 1998).

Within the eastern deciduous forest biome (Braun 2001), including at its western margins, a mosaic of forest, woodland, savanna, and prairie existed (Grimm 1984, Batek et al. 1999, Nowacki and Abrams 2008, Hanberry et al. 2012, Hanberry and Nowacki 2016). The variation in landscape has been attributed to a multitude of factors: density of human settlement, site factors such as soils, hydrology, geology, landform, and climate, as well as periodic disturbances including grazing, browsing, harvests, and fire (Anderson et al. 1999, Anderson et al. 2006, Dey and Kabrick 2015, Hanberry and Nowacki 2016). Historically, stand openness increased moving east-to-west across the central US (Patterson and Backman 1988.). Additionally, it has been shown that the ratio of trees/grass increases as precipitation increases (Curtis 1971, Anderson et al. 1999, Anderson et al. 2006). However, in many areas along the transitional ecotone between eastern deciduous forests and the tallgrass prairie of the Central Great Plains, where savannas and woodlands were once abundant, precipitation and soil moisture are
sufficient to support forests (Anderson et al. 1999). This suggests that a landscape-level factor apart from climate was creating and maintaining open vegetation communities prior to Euro-American settlement: fire (Abrams 1985, Abrams 1992, Anderson et al. 1999, Peterson and Reich 2001).

Increasing evidence based on dendrochronology, palynology, paleocharcoal, and paleoecology strongly suggests the central US was subjected to a spatially and temporally variable fire regime. This resulted in the diverse assemblage of plant community structure and composition that was historically present, and many credit high-frequency fire regimes with creating and maintaining woodland and savanna communities (Abrams 1985, Abrams 1992, Delcourt and Delcourt 1997, Anderson et al. 1999, Guyette et al. 2006, Stambaugh et al. 2006a). Although lighting strikes can be an ignition source for wildfire, most fire occurring in the central US has been linked to anthropogenic sources, influenced by human population density, culture, and land use (Guyette et al. 2006, McEwan et al. 2007, Stambaugh et al. 2011, Brose et al. 2013b, Stambaugh et al. 2014, Dey and Kabrick 2015).

**Oak and Fire Ecology**

The historically dominant presence of oaks in the eastern US is often associated with recurring fire, as partially evidenced by the co-occurrence of oak domination and increased charcoal abundance beginning around 9000 years B.P. (Abrams 1992). It has been hypothesized that oak forests require periodic fire disturbances for their successful regeneration and conservation (Abrams 1992). Several physiological and morphological traits in oaks support Abrams’ (1992) oak-fire hypothesis, such as improved germination with reduced litter layer, early development of taproot, ability of seedlings and stumps to
resprout after being top-killed, thicker bark than competitor species, and the ability to compartmentalize and recover from wounds (Krajicek 1960, Smith and Sutherland 1999, Nowacki and Abrams 2008, Johnson et al. 2009). Additionally, oaks are generally intolerant of shade and historically benefited from the open-canopied conditions that resulted from frequent fire regimes (Johnson et al. 2009).

Oaks produce large seeds (acorns) that provide a large store of carbohydrates to newly germinated seedlings. However, before a seedling can become established, a suitable seedbed for acorn germination is necessary that will allow penetration of the seed’s radicle into the soil (Johnson et al. 2009). Studies have shown that a reduction of leaf litter, such as by burning, can help to facilitate more direct contact between acorn and mineral soil, and thus increase the rate of seedling establishment (Krajicek 1960, Johnson et al. 2009). As seedlings become established, oaks characteristically develop a strong taproot while delaying shoot growth (Johnson et al. 2009). This allocation of carbon results in a large root-to-shoot ratio, allowing sprouts to opportunistically-and more quickly than true seedlings- respond to disturbance such as fire (Bond and Midgley 2001, Johnson et al. 2009).

Many upland oak species are considered vigorous sprouters after shoot die-back: a result of the characteristic large root-to-shoot ratio described above, as well as the presence of below ground buds at the root collar (Johnson et al. 2009). In mixed-oak forests in Ohio, Iverson and Hutchinson (2002) found average fire temperature 10 cm above the ground to be 220°C during a prescribed burn, while temperature sensors buried in the soil only recorded a mean temperature spike of 9.6°C above ambient soil temperatures. Mineral soil is a poor conductor of heat, with only approximately 5% of
heat energy released by a surface fire partitioned to the soil (Raison 1979). For oak reproduction, the location of buds below the ground surface protects them from the thermal damage of a fire, with soil acting as an insulating layer. Individual oaks can resprout from root collar buds one or more times after shoot dieback, and often seedling and stump sprouts are the predominant oak reproduction in a forest understory (Johnson et al. 2009). In the Ozark Highlands, Dey and Hartman (2005) found that, while fire damaged and occasionally killed advance oak reproduction, a high proportion of damaged stems resprouted in the following summer and greater mortality occurred in non-oak competitors. In central Virginia, Brose and Van Lear (1998) found oaks to be more resilient sprouters than red maple following fire, especially as fire intensity increased.

The flammability of oak leaf litter further reveals the ecological role of fire in perpetuating oak-dominated forests and woodlands. Oak litter has fast leaf drying rates and leaves curl when dry, creating a flammable, fine fuel bed that can more easily carry a surface fire (Johnson et al. 2009, Kreye et al. 2013). Unlike foliage from many mesophytic oak associates, oak litter is slow to decompose, due in part to a high lignin content, and remains on the forest floor as a flammable fuel source (Nowacki and Abrams 2008, Arthur et al. 2012). Without fire, oak is often outcompeted by shade-tolerant but fire-sensitive species such as maples, and if allowed to dominate a stand, such species may eventually produce the majority of leaf litter. In a historically oak-dominated community in Kentucky, Alexander and Arthur (2014) found that as maples became more dominant, the rate of forest floor decomposition increased and fuel loads
were reduced, a condition that further promoted the absence of fire and the dominance of maples and other mesophytic species over oak.

Other fire-adaptations of oaks include thick bark and the ability to recover from injury through the process of compartmentalization. In the Central Hardwood Region, species in the white oak group have the thickest bark, followed by red oaks (*Quercus* section *Lobatae*) (Hengst and Dawson 1994). Species such as maples and black cherry have much thinner bark than oaks, exposing these species to a greater risk of fire injury (Smith and Sutherland 2006). Additionally, bark growth in oaks is often faster than in competing species, with oak species generally adding more bark with incremental diameter growth (Hengst and Dawson 1994, Smith and Sutherland 2006). Bark characteristics are important for determining fire tolerance because bark insulates, and thus protects, the vascular cambium; species with thicker bark are generally less prone to cambial injury from fire (Smith and Sutherland 1999). Additionally, white oak species have a superior ability to compartmentalize decayed or injured tissue, improving survivorship after fire (Abrams 2003, Smith and Sutherland 1999). The process of compartmentalization promotes oak survivorship by limiting the spread of injury or infection, allowing an individual to overcome a disturbance such as fire and continue to competitively grow in a community (Sutherland and Smith 1999).

Oak seedlings and saplings are not well adapted to low light conditions (Abrams 1992, Johnson et al. 2009). To regenerate and recruit into the midstory and overstory, most upland oak species require periodic disturbances, such as fire, to create canopy gaps. For oaks, the rates of photosynthesis continue to increase to about one-third full light, an amount of sunlight considered ideal for oak development and recruitment into
larger size classes (Johnson et al. 2009). In the Central Hardwoods Region, one-third full light is achieved at about 60% stand stocking (Blizzard et al. 2013, Dey et al. 2017). The suppression of fire throughout the 20th century has led to increased tree densities, creating more closed-canopied, overstocked stands that block light to the understory and inhibit the growth and development of oak regeneration (Abrams 1992, Nowacki and Abrams 2008, Hanberry et al. 2014a, 2014b, 2014c).

**Fire History**

To understand the role historical fire regimes have played in shaping plant communities, fire scar dating techniques have reconstructed fire history chronologies across oak woodlands and savannas in the Central Hardwood and Forest-Prairie Transition Regions. Notably, humans in the eastern-central US are credited as a main source of wildfire ignition, with human population density and land use affecting fire regimes (Abrams 1992, Guyette et al. 2006). In the Mark Twain National Forest, MO, Cutter and Guyette (1994) found a mean fire interval (MFI) of 2.8 years before settlement, with fire frequency decreasing to an interval of 24 years after 1850. A similar return interval of 4.3 years was found to occur at Caney Mountain Wildlife Refuge, MO, 60 miles south of the Mark Twain NF sites (Guyette and Cutter 1991). A third Missouri study found a presettlement MFI of 3.7 years (Dey et al. 2004). Moving west of Missouri, Abrams (1985) calculated MFIs of 11 to 20 years for the period 1862-1983 in gallery forests in Konza Prairie in northeast Kansas based on fire scars. However, Abrams speculates that actual MFIs for these forests are likely shorter, as not all fires result in identifiable scars. He suggested that actual MFIs were closer to the historical presettlement interval of 2-3 years for the prairies of the Flint Hills (Abrams 1985).
Historically, these regular intervals of fire are credited with perpetuating open woodland communities by reducing woody encroachment, promoting fire-tolerant species/reducing fire-sensitive species, and encouraging herbaceous ground flora.

**Fire effects**

The relationship between oak forests and woodlands and fire has been well-documented, and numerous studies have attempted to quantify the effects of fire on the composition and structure of vegetation communities (Abrams 1992, Arthur et al. 2015, Hutchinson et al. 2012, Alexander et al. 2008, Knapp et al. 2015). Fire, as a natural disturbance, can create canopy openings, but these effects vary based on fire frequency, intensity, and duration. In Kentucky, a single prescribed burn resulted in a 3% decrease in canopy cover, but returned to pre-burn conditions the following year, suggesting only an ephemeral effect (Alexander et al. 2008). However, that same study showed that repeated burning resulted in canopy cover 5% lower than pre-burn cover, and 5 and 7% lower than sites unburned and burned once, respectively (Alexander et al. 2008). Another study in Kentucky found that 25% of overstory trees on sub-xeric and intermediate landscape positions experienced crown dieback after fire, with burn treatments having a higher proportion of trees that declined in crown dieback class over the duration of the study compared to fire-excluded sites (Arthur et al. 2015). An earlier study in Kentucky and Tennessee investigating effects of thinning and burning treatments concluded that canopy cover changes only occurred where thinning treatments were applied in addition to fire, with no decrease in cover with fire only treatment (Franklin et al. 2003). In southern Ohio, closed-canopy conditions essentially persisted with canopy openness averaging <6% for twice burned, four times burned, and unburned treatments (Hutchinson et al. 2012).
After 60 years of prescribed fire in the Missouri Ozarks, researchers found leaf area index to be greater and canopy openness to be less in no burn plots than in annually or periodically burned plots, indicating that while short-term or single burn studies show little to no change in canopy openness, long-term prescribed fire use may have a significant effect on reducing canopy cover (Knapp et al. 2015).

The effects of fire on overstory stand structure are highly variable, governed in part by existing stand composition and structure, fuels, and fire behavior. In a California oak woodland, fire treatments did not alter, and were only able to maintain, stand structural characteristics, including species composition, tree density, basal area, and crown closure (Fry 2008). Similarly, studies within the Central Hardwood Region have also found overstory tree density and basal area unaffected by fire (Blake and Schuette 2000, Franklin et al. 2003, Hutchinson et al. 2005, Kinkead et al. 2013). Conversely, in a Minnesota oak woodland, Peterson and Reich (2001) observed basal area reductions of 4-7% and density reductions of 6-8% per year in stands burned four or more times. In the central Appalachian region of Kentucky, burning reduced overstory (>10 cm DBH) stem density by 30% compared to no significant change in the fire-excluded units. However, changes occurred primarily in stems 10-20cm DBH, with no significant mortality in stems >20cm DBH (Blankenship and Arthur 2006). A long-term study in the Missouri Ozarks revealed that after 60 years of annual or periodic fire, the greatest differences in stand structure were seen in smaller size classes, with fire inhibiting ingrowth (Knapp et al. 2017).

While large changes in overstory structure are generally not produced from prescribed fire, the effects on the midstory can be very significant. In Kentucky,
Blankenship and Arthur (2006) found that after nine years of burning, midstory (2-10 cm DBH) stem density was reduced by 91% and midstory basal area by 86%. In fire-excluded units, midstory density only decreased 24% and basal area 28%, with red maple experiencing greater reductions than oaks. A later study in Kentucky found the greatest reductions in density and basal occurred in the sapling (trees 2-10 cm DBH) and midstory tree strata (Arthur et al. 2015). In the Missouri Ozarks, after 10 years of burning (MFI = 3.6 years) there was a 50% reduction in density of midstory trees (1.5-4.5 inches DBH) (Dey and Fan 2009). Hutchinson et al. (2012) found sapling (3-9.9 cm DBH) density reduce by 76%, and midstory trees (10-25 cm DBH) reduced by 34% after 13 years of periodic burning in Ohio. High frequency burning over 32 years in upland oak savannas and woodlands in Minnesota resulted in virtually no sapling layer, however over that same time a lower frequency treatment (4 burns in 32 years) resulted in very dense sapling thickets, suggesting fire frequency is an important factor controlling density of midstory layer (Peterson and Reich 2001).

As with the midstory tree stratum, the effects of fire on understory are highly species-dependent. A meta-analysis of 32 prescribed fire studies in the eastern U.S. by Brose et al. (2013a) revealed that generally prescribed fire preferentially selected for oak reproduction and against mesophytic hardwood reproduction, and that oak seedlings tended to establish at greater rates in burned areas than in unburned areas. In eastern Kentucky, prescribed burns were able to temporarily reduce red maple survival. Neither single nor repeated burns placed oaks in an improved competitive position after 6 years, as small diameter oaks experienced low post-burn survival (Alexander et al. 2008). Conversely, an earlier study in that same area of Kentucky found higher numbers of oak
regeneration on the burned treatments (burned either two or three times), but red maple stems were higher as well, and red maple saplings far exceeded oak saplings (Blankenship and Arthur 2006). In mature forests of the Ozark Highlands, MO, Dey and Hartman (2005) found that after a single prescribed fire >90% of seedlings and saplings of all species survived, and probability of survival was significantly related to initial stem size. That study showed that oak regeneration, especially post (*Quercus stellata*), black (*Quercus velutina*), and white oaks (*Quercus* section *Quercus*), had higher probability of survival following repeated burning than their mesophytic hardwood competitors (Dey and Hartman 2005).

Woodland communities are partially defined by the rich herbaceous plant communities that grow on the forest floor (Nelson 2010). Increasing tree densities and canopy cover may increase shade-tolerant understory species, while heliophytic species abundances are reduced. The dense shade of a closed woodland reduces the variability of resources, namely light, and can thus reduce heterogeneity in the plant community structure and composition, with ground flora beginning to resemble that of a forest (Dey and Kabrick 2015). Additionally, increased litter accumulation from the canopy can further impede the growth of herbaceous ground flora, where species diversity is often much greater than in the woody midstory and overstory (Taft et al. 1995, Stambaugh et al. 2006b, Peterson and Reich 2001, Peterson and Reich 2008, Dey and Kabrick 2015). The removal of litter by fire can promote woodland herb germination by removing dense layers of litter and organic soil layers, a requirement for many woodland graminoids and forbs (Burton et al. 2011). Following 17 years of annual prescribed fire in an Illinois oak forest, understory herbaceous cover and species richness significantly increased, while
understory woody species decreased (Bowles et al. 2007). In a Minnesota oak woodland, Peterson and Reich (2008) found highest species richness and cover with biennial fires for forbs and with annual fires for grasses, and greatest overall species richness occurred with biennial fires. Fire frequency was found to have a positive linear relationship in species richness and cover of forbs and C3 graminoids in an Oklahoma oak woodland (Burton et al. 2011). Conversely, Hutchinson et al. (2005) did not find significant increases in grasses and forbs five years following fire, with annual and periodic fire resulting in similar vegetation response.

**Thinning effects**

While prescribed fire is a common tool used for various oak woodland restoration objectives (e.g. reduce competitor abundance and increase understory light), fire alone is commonly ineffective at changing overstory structure and canopy openness, especially where overstory tree densities have become relatively high. The effects of low to moderate intensity fire are often seen in the understory and smaller midstory stratum, with no significant effects occurring to larger midstory and overstory trees (Franklin et al. 2003, Abrams 2005, Hutchinson et al. 2005, Kinkead et al. 2013, Arthur et al. 2015). In Missouri, it has been shown that repeated burning can meet restoration objectives for closed woodlands (Knapp et al. 2015), but this is considered a long process (e.g., 30-60 years) and the formation of canopy gaps is crucial for successful oak regeneration (Brose 2014). If more rapid oak recruitment is a management objective, additional silvicultural treatments such as thinning may be necessary to reduce midstory and overstory densities and create greater canopy openness. Additionally, mechanical/chemical thinning treatments, unlike fire, have the advantage of being more precise in removing select
species and individuals, as well as obtaining desired spatial arrangements of trees, stand stocking levels, and diameter distributions (Dey et al. 2017).

Several studies have found varying results on the effects of thinning and/or application of chemical herbicide on stand structure and composition, as well as ground flora response. In a savanna restoration study in Iowa, Brudvig and Asbjornsen (2007) found that thinning treatments were initially successful at reducing encroaching sapling layers, but that vigorous stump sprouting from mesophytic species continued to outcompete oaks in the regeneration layer. In Ohio, Albrecht and McCarthy (2006) found similar results, with thinning promoting early-successional regeneration and accelerating sprouting of oak competitors, thus providing little benefit for oak regeneration.

Promoting the abundance, richness, and cover of herbaceous ground flora is often an objective of thinning/harvesting treatments in a woodland restoration setting. In the Missouri Ozarks, Zenner and others (2006) found harvesting increased the abundance of herbaceous vegetation, with ground flora percent cover and species richness increasing with harvest intensity. Similarly, Kinkead et al. (2013) found the cover of woodland indicator species increased in harvested stands versus control, with greatest increases observed in stands that were burned and harvested.

**Project Overview**

The loss of historically open oak woodland and savanna communities has been well-documented across the central United States. Increasingly, there is an interest in the use of silvicultural practices for restoring these rare community types. However, there is a paucity of research on the effects of restoration treatments, namely prescribed fire and
thinning treatments, on woodland communities at the westernmost margins of the eastern deciduous forests. This project was established at that Marais des Cygnes and La Cygne State Wildlife Areas, KS in an effort to better understand restoration treatment effects in a prairie-forest ecotone, and to provide a comparison to established studies that occur farther east in the Central Hardwood Region.

The goals of this project included determining treatment effects on woodland structure and composition, and fire effects on advance regeneration (i.e., survival and sprouting) in the first growing season following treatments. Treatments included prescribed fire, thin to 60 ft²/acre basal area, thin to 30 ft²/acre basal area, the combination of fire and thin to 60 ft²/acre basal area, the combination of fire and thin to 30 ft³/acre basal area, and an untreated control. In Chapter II, the effects of the six treatments on stand structure and composition in the first growing season following treatments are discussed. The specific objectives for Chapter II were to: 1) determine treatment effects on overstory basal area, density, percent stocking, and diameter distributions, 2) determine treatment effects on sapling, large seedling, and small seedling density by species group, 3) determine treatment effects on canopy openness, and 4) determine treatment effects on ground cover by physiognomic class. In Chapter III, the effects of fire on the advance regeneration of five species are discussed. The specific objectives for Chapter III were to: 1) determine percent mortality and percent shoot dieback with sprouting for five different species 2) analyze relationships between initial stem size (basal diameter and height) and probability of survival, 3) determine the effect of fire temperature on the probability of survival, and 4) compare how mortality, sprouting, and survival probability differ between the five species.
Chapter II: The Effects of Burning and Thinning on Oak Woodland Stand Structure and Composition

Introduction

Oaks (*Quercus*) are a widespread genus in the eastern United States, represented by approximately 40 species east of the 100th meridian (Johnson et al. 2009). In presettlement eastern US forests, oaks were the most abundant genus, ranging from 40 to 70% of the total composition (Abrams 1992, Hanberry and Nowacki 2016). The greatest proportion of oak occurred along the western margins of the eastern deciduous forests within the Forest-Prairie Transition Region (Braun 2001, Bailey 1995, Bailey 1997, Hanberry and Nowacki 2016). In this ecotone between forests and the grasslands of the Great Plains, oak savannas and woodlands covered 27 to 32 million acres prior to 1840 (Nuzzo 1986, Abrams 2003, Nowacki and Abrams 2008, Hanberry and Nowacki 2016).

In the central US, oaks existed in a mosaic of prairie, savanna, woodland, and forest (Nuzzo 1986, Anderson et al. 1999, Nelson 2010). Increasing evidence strongly suggests this region was subjected to a spatially and temporally variable fire regimes, with open woodlands and savannas occurring where fire was frequent (Abrams 1985, Guyette and Cutter 1991, Guyette et al. 2006, Stambaugh et al. 2006, Rooney 2017). It is widely recognized that the dominance of oaks in these vegetation communities is a result of the adaptations of oak to periodic fire regimes (Abrams 1992, Arthur et al. 2012, Brose et al. 2001, Nowacki and Abrams 2008). Furthermore, the regular suppression of fire throughout the 20th century is regarded as a main driving force behind declines in general oak abundance in the eastern US, and specifically to the loss of historically open-canopy
communities (Nuzzo 1986, Abrams 2003, Nowacki and Abrams 2008). Currently, oak savannas and woodlands have been reduced to 0.02% of their presettlement area, and are among the most threatened communities in the US (Nuzzo 1986, Noss 1995, Smith et al. 2009).

Coinciding with the decrease in oak dominance, mesophytic species, such as red maple (*Acer rubrum*) and sugar maple (*Acer saccharum*), are increasing throughout the eastern US (Nowacki and Abrams 2008, Hanberry 2013, Hanberry et al. 2014b). Historically, fire-sensitive, mesophytic species were limited to areas of infrequent fire, such as rock outcrops, mesic coves, north aspects, and riparian areas. As a result, fire-tolerant oaks could thrive in areas of frequent fire without the competition from more fire-sensitive, shade-tolerant species. The encroachment of mesophytic species into historically open oak-dominated woodlands and savannas has resulted in increased overstory tree densities, altered tree composition, creation of a dense mid-story, greater canopy cover, and reductions in herbaceous plant cover and diversity (Abrams 1992, Nowacki and Abrams 2008, Hanberry 2014a, 2014b). Furthermore, the loss of fire as a natural disturbance has led to major declines in oak regeneration, a result of more shaded conditions and the increased competition from faster growing, shade-tolerant species (Abrams 1992, Nowacki and Abrams 2008, Johnson et al. 2009).

There is a growing interest by researchers and land managers in the restoration and maintenance of oak woodlands, with common objectives including decreasing overstory density and canopy cover, reducing the midstory tree stratum, eliminating undesired fire-sensitive species, promoting herbaceous ground flora, and promoting the regeneration of oaks (Dey et al. 2017). Past studies have analyzed the reintroduction of
fire to meet restoration objectives and often find low-intensity prescribed fires ineffective at removing encroaching woody species (Franklin et al. 2003, Abrams 2005, Dey and Hartman 2005, Hutchinson et al. 2005, Kinkead et al. 2013, Arthur et al. 2015). Often in areas where fire has been suppressed since the mid-1900s, fire-sensitive species have attained large enough sizes that allow for a high probability of survival following fire (Dey and Hartman 2005). For this reason, combining prescribed fire with additional silvicultural practices is often necessary to obtain desired woodland structural and compositional objectives.

Numerous studies across the Central Hardwood Region have investigated the use of prescribed fire accompanied by mechanical or chemical removal of large diameter overstory and midstory stems (Brose et al. 1999, Franklin et al. 2003, Albrecht and McCarthy 2006, Kinkead et al. 2013, Iverson et al. 2017). While thinning is an efficient method for reducing stem density and canopy cover, thinning alone may merely act as a release event for faster growing competitors in the understory if oak advance regeneration is not adequate (Brudvig and Asbjornsen 2007, Schweitzer et al. 2015). Prescriptions that include fire and thinning treatments often are most effective at promoting oak regeneration and herbaceous ground flora because they increase light and growing space to understory vegetation, while also reducing competition from encroaching fire-sensitive, shade-tolerant species (Brose et al. 1999, Franklin et al. 2003, Albrecht and McCarthy 2006, Kinkead et al. 2013, Iverson et al. 2017).

Oak woodland restoration treatment effects may differ depending on site factors, pre-treatment structure and composition, land use history, and geographic region. Despite the relative ubiquity of oak woodlands and savannas in the presettlement Forest-Prairie
Transition Region, oak woodland studies have largely been confined to the Central Hardwood Region. Much of the existing research on fire and thinning effects of oak communities examines white oak (*Quercus alba*) communities that are encroached upon by red maple (*Acer rubrum*) (Albrecht and McCarthy 2006, Brudvig and Asbjornsen 2007, Hutchinson et al. 2012, Iverson et al. 2017, Vander Yacht et al. 2017). At our sites in Kansas, chinkapin oak (*Quercus muehlenbergii*) and post oak (*Quercus stellata*) communities are heavily invaded by sugar maple: an analogous, but unique composition that may produce different outcomes to restorative treatments than more eastern studies.

This study examines the effects of experimental restoration treatments in oak woodlands at the Marais des Cygnes and La Cygne State Wildlife Areas, Kansas. The goal of this study was to examine the effects of prescribed fire, two levels of thinning intensity (residual basal area 30 ft²/acre and 60 ft²/acre), and the combination of prescribed fire and two thinning treatments on woodland structure and composition in the first growing season following treatments. The specific objectives included: 1) determine treatment effects on overstory basal area, density, percent stocking, and diameter distributions, 2) determine treatment effects on sapling, large seedling, and small seedling density by species group, 4) determine treatment effects on canopy openess, and 5) determine treatment effects on ground cover by physiognomic class.
Methods

Site Description

This study was conducted at Marais des Cygnes (N38°15'25.7" W94°40'59.9"") and La Cygne State Wildlife Areas (N38° 24' 33.6744" W94° 39' 42.4908") in Linn and Miami Counties of eastern Kansas (Figure 1). These areas are managed by the Kansas Department of Wildlife, Parks, and Tourism (KDWPT), an agency responsible for maintaining habitat for multiple game species. Marais des Cygnes Wildlife Area (MDC), located along the floodplain of the Marais des Cygnes River, is 3,100 hectares of mainly wetlands and bottomland hardwood forest, but also includes a smaller upland forest component where the MDC replication of this study is located. Located 16 kilometers north of MDC, La Cygne Wildlife Area (LCWA) is 810 hectares of mostly wooded uplands adjacent to La Cygne Lake, a 1,050-hectares cooling reservoir for a coal-fired generating plant owned by Kansas City Power and Light. Two replications are located within the upland forest of LCWA.

Located in the Forest-Prairie Transition ecoregion (Bailey 1995, Bailey 1997), the sites occur within the Wooded Osage Plain physiographic region of Kansas (McNab and Avers 1994, Chapman et al. 2001). The climate is temperate, humid, and continental, characterized by large daily and annual variations in temperature. Between 1981 and 2010, the average annual air temperature was 12.9 °C, with an average daily temperature of -1.1 °C in January and 25.7 °C in July (NOAA). During that same time period, average annual precipitation was 1,017 mm, with heaviest precipitation occurring in late spring and early summer.
Figure 1. Map of site locations (purple polygons) at Marais des Cygnes and La Cygne State Wildlife Areas, KS. Inset is the location of wildlife areas (purple star) within the Forest-Prairie Transition Region of the United States (Bailey 1997).
The Wooded Osage Plains ecoregion is a broad transition region shifting from prairie to woodland, characterized by a series of roughly parallel southwest- to northeast-oriented escarpments, separated by gently rolling to level plains (McNab and Avers 1994; Chapman et al. 2001). Historically, tallgrass prairie was the dominant vegetation of the ecoregion (70%), with corridors of oak-hickory forest along drainageways (McNab and Avers 1994). Today, agriculture, particularly corn and soybeans, is the predominant land use in the ecoregion, although pastureland and rangeland comprise a smaller, but sizable component of the area (Penner 1981). Most land is privately owned, however numerous federal- and state-owned areas are found throughout the region. Remnant natural communities cover less than 10% of the total area of the region, are highly fragmented, and mainly occur on the least agriculturally productive sites (Hamilton et al. 2000).

The study areas encompass mainly ridge, shoulder, and backslope topographic positions, with elevations ranging from 260 to 310 m and 0 to 45% slope. Soils are Mollisols, mainly of the Clareson series, with limestone rock outcrops, and are moderately deep and well-drained (Web Soil Survey). Tree species composition commonly included chinkapin oak, post oak, black oak (*Quercus velutina*), bur oak (*Q. macrocarpa*), sugar maple, and hickories (*Carya* spp) in the overstory, with sugar maple, eastern redbud (*Cercis canadensis*), elm (*Ulmus* spp.), ironwood (*Ostrya virginiana*), and roughleaf dogwood (*Cornus drummondii*) in the mid- and understories. Common shrub and vine species include coralberry (*Symphoricarpos orbiculatus*), fragrant sumac (*Rhus aromatica*), blackberry (*Rubus* spp.), Virginia creeper (*Parthenocissus quinquefolia*), and grapevine (*Vitis* spp.). Prior to study initiation, stands averaged 80% stocking (Gingrich 1967), basal areas commonly averaged >20 m² per hectare, and closed-canopy conditions
existed. Many of the oaks at the three replications display tree architectures expressing past open canopy, woodland conditions, namely spreading crowns, primary branches located low on the bole, and shorter stature trees (Nelson 2010).

Fire scarring on tree stems reveal these areas once experienced some interval of fire in the past. On April 7, 2004, KDWPT implemented a prescribed burn at the MDC site. The fire was described as low to moderate severity. A second prescribed burn was conducted the following year on April 14, 2005. The 2005 burn was of a lower intensity due to reduced fuel load following the 2004 burn. Previous to the 2004 and 2005 prescribed burns, there is no history of fire occurring for at least 50 years. No history of recent fire exists for LCWA.

Study Design

The study is a complete randomized block design, with three sites (replications) established on separate areas of MDC and LCWA (Figures 2-4). Each of the three sites are approximately 36 acres in size. This study is designed with six treatments applied across each of the three sites. Each of the six treatment types were randomly assigned to an approximately six-acre treatment unit within each of the three sites. Treatments included prescribed fire (burn), thin to 60 ft²/acre basal area (T60), thin to 30 ft²/acre basal area (T30), the combination of fire and thin to 60 ft²/acre basal area (BT60), the combination of fire and thin to 30 ft²/acre basal area (BT30), and an untreated control. Within each treatment unit, six permanent vegetation plots were randomly located, creating 36 plots per replication and a total of 108 plots. All plots were established at least 18.3m from a treatment unit boundary.
Figure 2. Map of the treatment units and plots at the Marais des Cygnes site (MDC).
Figure 3. Map of the treatment units and plots at the La Cygne Replication 1 site (LC1).
Figure 4. Map of the treatment units and plots at the La Cygne Replication 2 site (LC2).
Vegetation Sampling

The establishment and pre-treatment data collection of 108 permanent, fixed-radius 0.08-hectare vegetation plots occurred in summer 2015 (May-August) (Figure 5). The aspect, percent slope, and hillslope position was recorded for each plot. All trees \( \geq 3.8 \) cm in diameter at breast height (dbh) were inventoried for the entire 0.08-hectare plot. Recorded data included tree species and dbh. Additionally, trees were assigned into four crown classes: dominant, codominant, intermediate, and suppressed. The dominant class is defined by trees with crowns extending above the general level of the main canopy that receive full light from above and partly from the sides, while codominant tree crowns form the general level of the main canopy, receiving full light from above and comparatively little from the sides (Smith 1986). Intermediate trees have crowns that extend into the lower portion of the main canopy and receive little direct light from above and no light from the sides (Smith 1986). Lastly, suppressed trees have crowns entirely below the general level of the canopy and do not receive any direct sunlight (Smith 1986).

Four nested 0.004 hectare subplots were placed 8 m north, east, south, and west of the plot center. All trees \( >1 \) m tall and \( <3.8 \) cm dbh were tallied by species in each of the four subplots. Four 1 m² quadrats were nested within each subplot north, east, south, and west of subplot center; the inner corner of quadrats were 2.2 m from subplot center and outer corners touched the edges of subplot (4 quadrats/subplot \* 4 subplots = 16 total quadrats/plot). Within each quadrat all tree seedlings \( <1 \) m tall were identified to species, foliar cover was estimated to the nearest percent, and each individual was tallied into one of ten height classes: \(<10 \) cm, 10-20 cm, 20-30 cm, 30-40 cm, 40-50 cm, 50-60 cm, 60-
70 cm, 70-80 cm, 80-90 cm, 90-100 cm. Additionally, within each quadrat foliar cover up to 1 m in height was estimated for the following physiognomic (i.e., functional) classes: forbs, grasses, sedges, legumes, trees, shrubs, and woody vines. Cover of stem, wood, rock, bare soil, bryophyte, and litter was also recorded in each quadrat. Post-treatment data were collected in summer of 2016, the first growing season after application of treatments.

Figure 5. Design of the 0.08-hectare vegetation plot. Four 0.004-hectare subplots were placed in each cardinal direction from plot center and sixteen 1 m² quadrats were nested within subplots.
Photography

All 108 plots were photographed before treatments were implemented during the summer of 2015. All photographs were taken with a Canon EOS 5D digital single-lens reflex (DSLR) camera and Canon EF 50 mm lens. Photographs were captured in a raw image format in order to preserve the most information of the captured image, and images were later converted to a JPEG file format. Photographs were taken using a tripod from plot center in four cardinal directions at breast height (1.37 m) to provide a visual reference of plots for long-term comparison. Photographs were taken in a horizontal (landscape) orientation, and were centered on a photo board attached to 6 ft tall pole placed at the edge of the plot in each cardinal direction. All plot photographs were retaken in the summer of 2016 to document the initial changes following treatments.

Additionally, five hemispherical photographs of the canopy were taken at each plot using a Sigma 8mm fisheye lens. In every plot, a canopy photograph was taken at the plot center and the center of each of the four subplots. A tripod was used to take photographs at a height of 1 m above the ground. For each photograph, the camera was oriented north and both the tripod and camera were carefully leveled before capturing each photograph. To avoid lens flare, which includes bright streaks or a washed out image as a result of bright light, canopy photographs were only captured shortly after sunrise and before sunset when the sun was low on the horizon, or on overcast days. Taking the photographs during the lower light conditions described also allowed for a better contrast between the canopy and sky. All hemispherical canopy photographs were retaken post-treatment in summer of 2016.
Hemispherical photographs were analyzed using WinSCANOPY (WinSCANOPY Basic 2016a, Regent Instruments Canada Inc.) imaging software to estimate canopy openness, defined as the proportion of open sky unobstructed by vegetation in a specified region of the real canopy above the lens. No editing or cropping of the original photograph was done, however, as with the plot photographs, canopy photographs were taken in raw image format and converted to JPEG before being analyzed. To determine canopy openness, the WinSCANOPY software classifies each pixel into canopy and sky based on grey levels (thresholds). Pixels brighter than the threshold value are classified as sky (white) and the others as canopy (black), and a ‘pixels classification image’ is generated (Figure 6). Automatic thresholds were created by the software and used for pixel classification, and a subset of photographs were closely reviewed for accuracy of the threshold. A batch analysis was performed and canopy openness determined based on proportion of sky (white) pixels in the image.
Figure 6. (A) Original hemispherical canopy image. (B) Pixel classification image generated by WinSCANOPY software.
Treatments

The thinning treatments were implemented in January 2016 when sap flow was minimal. Prior to the thinning treatment, leave trees were marked with paint and chosen based on desired species and form to achieve appropriate residual basal area in treatment units. The priority species marked as leave trees were oaks, hickories, and walnut; best efforts were made to not leave sugar maple and other mesophytic species. All trees not marked and > 5.1 cm dbh received a single girdle treatment completely around the stem (smaller trees > 5.1 cm were completely felled at the discretion of the contractor).

Herbicide (Triclopyr product) was applied to cut stumps or girdles to prevent re-sprouting of undesirable stems. Herbicide was not applied on the following girdled or felled species to allow sprouting to occur: oak, hickory, walnut, and Kentucky coffee tree (*Gymnocladus dioicus*). Additionally, no herbicide was applied to eastern redcedar (*Juniperus virginiana*), as this species does not sprout following shoot death or injury.

Oaks not marked as leave trees in the 5.1 to 12.7 cm size class were not treated, as this size class contributes less to overall basal area and large advance oak regeneration was lacking at all sites.

Thinning treatments were defined by basal area in English units (e.g., 30 ft²/acre), and the naming of the treatment types reflect this unit of measurement. However, the stand metrics calculated and presented throughout the results are in metric units. Table 1 displays the target residual basal area of each treatment type in both systems of measurement as a reference.

During the 2016 post-treatment sampling of the vegetation plots, every tree measured in the 0.08-hectare vegetation plot (trees ≥ 3.8 cm) was documented as either a
leave tree (presence of paint mark) or not a leave tree. To determine the accuracy of our marking efforts, the basal area of leave trees was calculated for each plot and averaged for each thinning treatment (Table 2).

Prescribed fire was applied at the MDC site on March 5, 2016, and at LC1 and LC2 sites on March 17, 2016. All fires occurred before leaf emergence of woody species. Fire temperature and fire behavior measurements were recorded at the three sites. See Chapter III Methods for details on burn day conditions, fire temperatures, and fire behavior.

Table 1. Thinning treatments with the targeted residual basal area (BA) of prescription in English and metric units.

<table>
<thead>
<tr>
<th>Thinning treatment</th>
<th>English units BA</th>
<th>Metric Units BA</th>
</tr>
</thead>
<tbody>
<tr>
<td>T60</td>
<td>60 ft²/acre</td>
<td>13.8 m²/hectare</td>
</tr>
<tr>
<td>T30</td>
<td>30 ft²/acre</td>
<td>6.9 m²/hectare</td>
</tr>
<tr>
<td>BT60</td>
<td>60 ft²/acre</td>
<td>13.8 m²/hectare</td>
</tr>
<tr>
<td>BT30</td>
<td>30 ft²/acre</td>
<td>6.9 m²/hectare</td>
</tr>
</tbody>
</table>

Table 2. The mean, minimum, and maximum residual basal area of marked ‘leave’ trees.

<table>
<thead>
<tr>
<th>Thinning treatment</th>
<th>Mean BA (ft²/acre)</th>
<th>Minimum BA (ft²/acre)</th>
<th>Maximum BA (ft²/acre)</th>
</tr>
</thead>
<tbody>
<tr>
<td>T60</td>
<td>49.1</td>
<td>28.4</td>
<td>65.8</td>
</tr>
<tr>
<td>T30</td>
<td>33.0</td>
<td>14.1</td>
<td>51.2</td>
</tr>
<tr>
<td>BT60</td>
<td>53.1</td>
<td>34.6</td>
<td>70.7</td>
</tr>
<tr>
<td>BT30</td>
<td>30.2</td>
<td>16.8</td>
<td>41.8</td>
</tr>
</tbody>
</table>

Data Analysis

For each year of data collection (2015 and 2016), we calculated plot-level means for overstory variables (total trees per hectare (TPH), basal area, and percent stocking),
sapling density (TPH), large and small seedling density (TPH), percent canopy openness, and percent groundcover by physiognomic class (forb, graminoid, legume, tree, shrub, and vine). Data on tree densities for the sapling size class were grouped based on species abundance and functional and taxonomic similarity: oak (*Quercus muehlenbergii*, *Q. stellata*, *Q. macrocarpa*, *Q. velutina*, *Q. rubra*), hickory and black walnut (*Carya cordiformis*, *C. glabra*, *C. ovata*, *C. tomentosa*, *Juglans nigra*), sugar maple and “other species” (e.g., *Ulmus americana*, *Ulmus rubra*, *Cercis canadensis*, *Fraxinus pennsylvanica*, *Fraxinus americana*, *Cornus drummondii*, *Ostrya virginiana*, *Morus rubra*). For seedling size classes, additional species (e.g., *Cercis canadensis*) were analyzed due to the higher densities that existed in smaller classes.

The general linear mixed models procedure (Proc GLIMMIX, SAS version 9.4, SAS Institute, Cary, NC) was used to examine the effects of treatment on the 2016 plot-level variables described above (burn, T60, T30, BT60, BT30, and control). We developed separate ANCOVA models for each dependent variable. The fixed-effects included treatment type (control, burn, T60, T30, BT60, BT30) and the 2015 pretreatment plot means as a covariate. Random-effects included site (MDC, LC1, LC2) as the random block variable and the site*treatment interaction. Prior to modeling, each dependent variable was tested for normality and transformed using a square root, cube root, or natural logarithm function when necessary, and equality of variance was graphically observed. Normality was determined using Wilk’s test (W>0.90). For marginal models that did not achieve normality even through transformations, a robust variance estimator was used (empirical statement in Proc GLIMMIX). In one case, an ANOVA model, rather than ANCOVA, was developed to examine treatment effect on
canopy openness, with the post- to pre-treatment difference as the dependent variable. Significant differences (α=0.05) among treatment types were determined using Tukey's Honestly Significant Difference tests.

Results

Thinning Treatment Mortality

A total of 2,042 individual trees were treated with either a single girdle or a single girdle and herbicide application within the plots of the T60, T30, BT60, and BT30 treatment units (Table 3). In the first growing season following the thinning treatments, 38.0% of all treated trees were dead. Mortality of the treated trees varied by species. Osage orange (Maclura pomifera), red mulberry (Morus rubra), elm (Ulmus spp.), and eastern redbud (Cercis canadensis) all experienced at least 90% mortality following the thinning treatment. Oak species experienced between 40 and 45% mortality. Sugar maple experienced the lowest mortality, with only 12.8% of the 853 trees that were treated experiencing mortality.
Table 3. Total number of trees treated by a single girdle (oak, hickory black walnut) or a single girdle and herbicide application (all other species) in the T30, T60, BT30, and BT60 treatments, and the percent dead in the first growing season following treatment.

<table>
<thead>
<tr>
<th>Species</th>
<th>Number Treated</th>
<th>Number Dead</th>
<th>Percent Dead</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total</td>
<td>2042</td>
<td>775</td>
<td>38.0</td>
</tr>
<tr>
<td>Chinkapin Oak (<em>Quercus muehlenbergii</em>)</td>
<td>236</td>
<td>106</td>
<td>44.9</td>
</tr>
<tr>
<td>Other oak species (<em>Quercus spp.</em>)</td>
<td>258</td>
<td>106</td>
<td>41.1</td>
</tr>
<tr>
<td>Sugar Maple (<em>Acer saccharum</em>)</td>
<td>853</td>
<td>109</td>
<td>12.8</td>
</tr>
<tr>
<td>Hickory (<em>Carya spp.</em>)</td>
<td>134</td>
<td>27</td>
<td>20.2</td>
</tr>
<tr>
<td>Black walnut (<em>Juglans nigra</em>)</td>
<td>44</td>
<td>23</td>
<td>52.3</td>
</tr>
<tr>
<td>Osage orange (<em>Maclura pomifera</em>)</td>
<td>136</td>
<td>131</td>
<td>96.3</td>
</tr>
<tr>
<td>Red mulberry (<em>Morus rubra</em>)</td>
<td>65</td>
<td>63</td>
<td>96.9</td>
</tr>
<tr>
<td>Elm (<em>Ulmus spp.</em>)</td>
<td>81</td>
<td>75</td>
<td>92.6</td>
</tr>
<tr>
<td>Ash (<em>Fraxinus spp.</em>)</td>
<td>108</td>
<td>20</td>
<td>18.5</td>
</tr>
<tr>
<td>Hackberry (<em>Celtis occidentalis</em>)</td>
<td>51</td>
<td>45</td>
<td>88.2</td>
</tr>
<tr>
<td>Eastern redbud (<em>Cercis canadensis</em>)</td>
<td>47</td>
<td>46</td>
<td>97.9</td>
</tr>
<tr>
<td>Other species</td>
<td>29</td>
<td>24</td>
<td>82.8</td>
</tr>
</tbody>
</table>

2016 Diameter Distributions

In 2015, all pretreatment diameter distributions conformed to either a reverse-J shape (negative exponential) or rotated sigmoid curve (Inset of Figures 7-12). This is indicative of an uneven-aged forest (Janowiak et al. 2008), with greater density of smaller trees than larger trees. The diameter distribution of oaks in both 2015 and 2016 follow a unimodal distribution, indicative of an even-aged cohort of oaks within the stands.

In 2016, the control treatment diameter distribution maintained the reverse-J pattern found in 2015 (Figure 7), with high tree densities in diameter classes under 10 cm. The most abundant diameter class was 4-6 cm, which averaged 170 trees/hectare: nearly twice as many as any other diameter class. In the 4-6 cm diameter class, sugar maple and eastern redbud had the greatest relative densities, comprising 29.3% and 27.3%, respectively. Sugar maple and eastern redbud also had highest densities in the
smallest diameter class (2-4 cm), together accounting for 71.2% of stems. Conversely, oaks accounted for only 1.5 and 1.4% of the 2-4 cm and 4-6 cm diameter classes, respectively, while diameter classes less than 10 cm were <15% in relative density. As diameter increased above 10 cm, the relative amount of oaks increased as well, with oaks accounting for at least half of all stems in nearly all diameter classes.

Similar to the control, the burn treatment exhibited a reverse-J shape (Figure 8). Generally, the total trees/hectare decreased with increasing tree diameter. Like the control, trees 4-10 cm had the greatest densities for the burn only treatment, with each diameter class averaging at least 80 trees/hectare. Also, like the control, the 4-6 cm diameter class had the greatest average density, with 167 trees/hectare. In the 4-6 cm diameter class, sugar maple and eastern redbud accounted for 29.5% and 20.1% of all stems, respectively. Other species, particularly American hop hornbeam (*Ostrya virginiana*) and hackberry (*Celtis occidentalis*), accounted for 29.4%. There were no oaks in the 2-4 cm diameter class, and oaks were less than 20% in all diameter classes up to a diameter of 12 cm. The relative density of oak increased with diameter class, with oaks over 14 cm representing at least 50% of stems in all diameter classes. Sugar maple, on the other hand, decreased in relative density when diameter increased above 24 cm, with sugar maple representing less than 20% relative density in most diameter classes. Hickories and black walnut had a relative density between 10-20% for most classes, except below a diameter of 20 cm, where frequencies below 10% were common. At diameters greater than 36 cm, each diameter class always had less than 10 trees/hectare, and only oaks reached diameters greater than 54 cm.
The T60 treatment in 2016 displayed a unimodal diameter distribution (Janowiak et al. 2008), with tree densities of 40 trees/hectare or less across all diameter classes (Figure 9). Differing from the densities present in the control and burn treatments, sugar maple and eastern redbud in the T60 treatment had relatively low densities in diameter classes less than 6 cm; instead, red oak, black walnut, and hickory species had the highest relative densities under 6 cm diameter, accounting for 68.4% of all stems. Sugar maple had the greatest densities in diameter classes from 6-18 cm, ranging from 42.3-52.7% relative density in each class. Oak species, including mainly white oak species, accounted for at least 54.3% of stems in each diameter class greater than 24 cm, with the greatest oak density (24.7 trees/hectare) and relative density (72.3%) occurring in the 26-28 cm diameter class. Generally as diameter increased, the relative density of oaks increased, while sugar maple decreased.

In 2016, the T30 treatment conformed to a unimodal diameter distribution, with the majority of classes having less than 20 trees/hectare (Figure 10). The relative density of eastern redbud for the 2-4 cm and 4-6 cm diameter classes was 71.4% and 21.4% percent, respectively. Conversely, sugar maple accounted for 0-10.7% in diameter classes less than 8 cm. Sugar maple had a relative density greater than 43.3% for diameters 8 to 18 cm, however, the relative density decreased above 18 cm. The relative density of oak species generally increased with increasing diameter, as oaks accounted for at least 60% of trees/hectare in nearly each diameter class greater than 20 cm.

The 2016 diameter distribution of the BT60 treatment was unimodal. The greatest density of any diameter class (12-14 cm) was 35 trees/hectare, and nearly all diameter classes had less than 30 trees/hectare (Figure 11). In the smallest diameter class, 2-4 cm,
oaks and sugar maple had the same relative density of 33.3%. However, for the diameter classes between 4-8 cm, oaks had a higher relative density (48%) than sugar maple (11%). Relative density of maple was at least 35% for all diameter classes between 8-22 cm, but decreased with increasing diameter above 22 cm. Conversely, the relative density of oaks generally increased with increasing diameter above 22 cm.

In 2016, the BT30 treatment conformed to a unimodal diameter distribution. The maximum trees/hectare was 36.4, with most diameter classes having less than 30 trees/hectare (Figure 12). In the smallest diameter classes (<8 cm) sugar maple occurred at a relative density of 50-55%, while oaks were 0-10%. The relative density of sugar maple was greatest for that species between diameters of 8-16 cm, and generally the relative density of sugar maple decreased as diameters increased above 16 cm. Conversely, the relative density of oaks increased with increasing diameter.
Figure 7. Post-treatment (2016) diameter distribution by species group for the control (C) treatment. Inset: Pretreatment (2015) diameter distribution for all species.
Figure 8. Post-treatment (2016) diameter distribution by species group for the burn (B) treatment. Inset: Pretreatment (2015) diameter distribution for all species.
Figure 9. Post-treatment (2016) diameter distribution by species group for the thin to 60 ft² acre⁻¹ (T60) basal area treatment. Inset: Pretreatment (2015) diameter distribution for all species.
Figure 10. Post-treatment (2016) diameter distribution by species group for the thin to 30 ft² (T30) acre⁻¹ basal area treatment. Inset: Pretreatment (2015) diameter distribution for all species.
Figure 11. Post-treatment (2016) diameter distribution by species group for the burn and thin to 60 ft$^2$ acre$^{-1}$ (BT60) basal area treatment. Inset: Pretreatment (2015) diameter distribution for all species.
Figure 12. Post-treatment (2016) diameter distribution by species group for the burn and thin to 30 ft$^2$ acre$^{-1}$ basal area treatment. Inset: Pretreatment (2015) diameter distribution for all species.
Overstory Tree Density, Basal Area, and Percent Stocking

Prior to treatment, overstory tree densities (dbh $\geq$11.4 cm) ranged from 383.7 to 443.4 trees/hectare, with an average of 413.7 (Table 4). Following treatments, the greatest tree density occurred in the burn treatment, which had 411.2 trees/hectare. With 406.4 trees/hectare, the control treatment was comparable to the burn treatment, and was the only treatment where density increased from 2015 to 2016. The T30 treatment resulted in the lowest density in 2016 with 293.1 trees/hectare. The BT60 had the greatest change in density, decreasing by 132.4 trees/hectare following treatment. Accounting for 2015 pretreatment tree densities, 2016 tree density was only significantly different than the control for the BT60 treatment. All other treatments did not have significantly different 2016 tree densities, even though their densities decreased.

In 2015, the basal area of overstory trees (dbh $\geq$11.4 cm) ranged from 10.8 to 32.9 m$^2$/hectare, with an average of 21.6 m$^2$/hectare. In 2016, basal area in the burn and control treatments increased slightly, while all other treatments decreased (Table 4). The T60 treatment had the lowest basal area in 2016, 17.7 m$^2$/hectare, and the BT60 treatment had the greatest change in basal area, with a decrease of 3.8 m$^2$/hectare. Although all thinned treatments showed a decrease in basal area, there were no significant differences in 2016 basal areas between treatment types.

Prior to treatments, the percent stocking ranged from 75.1% to 87.9%, with an average of 80.2% (Table 4, Figures 13-15). Following treatments, the burn and control treatments showed a slight increase in percent stocking; all other treatments decreased in percent stocking. In 2016, the T60 and BT30 treatments had the lowest percent stocking, both 66.2%, and the T30 treatment had the greatest change in percent stocking,
decreasing by 16.1%. Percent stocking of the control in 2016 was only significantly different than the BT60 treatment. The 2016 percent stocking for the burn, T60, T30, BT60, and BT30 treatments were not significantly different from each other.
Table 4. Mean (and one standard error) tree density, basal area, and percent stocking (trees dbh ≥ 11.4 cm) for each of six treatment types in 2015 and 2016. For 2016 values, the same superscript letter indicates no significant difference between treatments.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Year</th>
<th>Treatment</th>
<th>Control</th>
<th>Burn</th>
<th>T60</th>
<th>T30</th>
<th>BT60</th>
<th>BT30</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tree Density</td>
<td>2015</td>
<td>399.5 (32.4)</td>
<td>443.4 (41.9)</td>
<td>417.9 (22.4)</td>
<td>405.7 (26.6)</td>
<td>431.7 (25.0)</td>
<td>383.7 (18.9)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>406.4 (33.4)(^a)</td>
<td>411.2 (33.4)(^ab)</td>
<td>343.0 (19.7)(^ab)</td>
<td>293.1 (25.2)(^ab)</td>
<td>299.3 (26.8)(^b)</td>
<td>314.4 (18.2)(^ab)</td>
<td></td>
</tr>
<tr>
<td>Tree Basal Area</td>
<td>2015</td>
<td>21.3 (0.8)</td>
<td>20.9 (1.0)</td>
<td>19.8 (0.7)</td>
<td>24.0 (0.9)</td>
<td>22.6 (1.3)</td>
<td>21.0 (1.1)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>21.4 (0.8)(^a)</td>
<td>21.1 (0.9)(^a)</td>
<td>17.7 (0.9)(^a)</td>
<td>20.0 (1.1)(^a)</td>
<td>18.8 (1.3)(^a)</td>
<td>18.1 (0.9)(^a)</td>
<td></td>
</tr>
<tr>
<td>Percent Stocking</td>
<td>2015</td>
<td>79.1 (2.9)</td>
<td>78.5 (3.3)</td>
<td>75.1 (2.3)</td>
<td>87.9 (3.4)</td>
<td>83.3 (4.2)</td>
<td>77.5 (3.7)</td>
<td></td>
</tr>
<tr>
<td>(Gingrich 1967)</td>
<td>2016</td>
<td>79.7 (3.0)(^a)</td>
<td>78.8 (2.6)(^ab)</td>
<td>66.2 (3.0)(^ab)</td>
<td>71.8 (3.9)(^ab)</td>
<td>67.9 (4.4)(^b)</td>
<td>66.2 (3.1)(^ab)</td>
<td></td>
</tr>
</tbody>
</table>
Figure 13. Gingrich (1967) diagram displaying mean percent stocking in 2015 and 2016 for the control and burn treatments.
Figure 14. Gingrich (1967) diagram displaying mean percent stocking in 2015 and 2016 for the thin to 60ft$^2$ acre$^{-1}$ basal area (T60) and the thin to 30ft$^2$ acre$^{-1}$ basal area (T30) treatments.
Figure 15. Gingrich (1967) diagram displaying mean percent stocking in 2015 and 2016 for the burn and thin to 60ft$^2$ acre$^{-1}$ basal area (BT60) and the burn and thin to 30ft$^2$ acre$^{-1}$ basal area (BT30) treatments.
Sapling Density

Prior to treatments, total sapling (3.8 \( \leq \) dbh < 11.4 cm) density ranged from 382.3 to 538.8 trees/hectare, with an average density of 444.6 trees/hectare in 2015. Pretreatment oak densities ranged from 8.2 to 55.6 trees/hectare (Figure 16a.). Following treatments, the density of oak saplings in 2016 was significantly less than control in the T30 and BT30 treatments, however, the two treatments were not different from each other or other treatments types. The greatest density of oaks both pre- and post-treatment occurred in the control treatment, which in 2016 had an average density of 59 trees/hectare. Oak density sapling decreased by 8.2 trees/hectare or less in all treatment types except control, where density increased by 3.4 trees/hectare.

Pretreatment sugar maple sapling density was greater than oak density, with an average of 163.5 trees/hectare and range of 86.5 to 223.1 (Figure 16b). All thinned treatments (T30, T60, BT30, and BT60) had significantly different densities than control in 2016. The burn treatment was not significantly different than any treatment type; however, greater decreases of sapling density were observed in T60, T30, BT60, and BT30 than the burn treatment.

In 2015, hickory and black walnut sapling density was similar to oak density, ranging from 19.9 to 70.7 trees/hectare, with an average of 43. Like oak, hickory and walnut sapling density was significantly different than control for only the T30 and BT30 treatments (Figure 16b). The burn treatment and all thinned treatments were not significantly different than each other.
Prior to treatment, sapling density of other species – including eastern redbud 
(*Cercis canadensis*), slippery elm (*Ulmus rubra*), American elm (*Ulmus americana*), ash 
species (*Fraxinus* spp.), common hackberry, (*Celtis occidentalis*), red mulberry (*Morus 
rubra*), American hop hornbeam (*Ostrya virginiana*), and others – ranged from 167.5 to 
235.5 trees/hectare, with an average of 196.9 (Figure 16d.). Sapling density of other 
species was not significantly different between the control and burn treatment. The 
sapling densities for the T30, T60, BT60, and BT30 treatments were significantly 
different than the control and the burn treatments. The decreases in density of the T60, 
T30, BT60, and BT30 treatments ranged from 163.4 to 188.1 trees/hectare.

The post-treatment changes among the species groups reveal that sugar maple and 
other species saw greater magnitudes of change than oak, hickory, and walnut. The large 
decreases for sugar maple and other species occurred primarily in the T60, T30, BT60, 
and BT30 treatments. Despite the larger changes for sugar maple and other species, their 
2016 densities were similar to 2016 densities of oak, hickory, and walnut.
Figure 16. Saplings (3.8 ≤ dbh < 11.4 cm) per hectare by species or species group in 2015 and 2016 for each of six treatment types (mean and one standard error). For 2016, the same superscript letter indicates no significant difference between treatment types.
Large Seedling Density

In 2015, total large seedling (dbh <3.8 cm and height >1 m) density ranged from 1333.6 to 2793.7 trees/hectare, with an average of 1558.8. Pretreatment oak large seedling density ranged from 3.4 to 127.0 trees/hectare, with an average of 30.5 trees/hectare (Figure 17a). In 2016, the control was not significantly different than any treatment types. The only significant difference was between the burn and BT30 treatments. Between 2015 and 2016, oak density only increased or decreased by about 10 trees/hectare or less across all treatment types.

The density of sugar maple large seedlings in 2015 ranged from 37.8 to 892.3 trees/hectare, with an average of 269.9 (Figure 17b). Following treatments, sugar maple density decreased for all treatment types, with the largest changes occurring in the BT60 and BT30 treatments. There were no significant differences between treatment types.

Prior to treatments, the density of hickory and black walnut large seedlings ranged from 6.9 to 85.8 trees/hectare, with an average of 47.6 (Figure 17c). No 2016 densities were significantly different than the control; however, the burn treatment was significantly different than T60 and T30 treatments, and T60 and BT30 treatments were significantly different from each other. Density decreased in the burn, BT60, and BT30 treatments, while increasing in the other three treatments.

In 2015, large seedling density of eastern redbud ranged from 305.5 to 1197.8 trees/hectare, with an average of 775.4; the greatest pretreatment density of any species in this size class (Figure 17d). After treatments, only densities in the burn and BT60
treatment were significantly different than the control. Except the T30 treatment, all treatments showed a decrease in density, with the largest change in the BT30 treatment.

For large seedlings of other species, pretreatment densities ranged from 159.9 to 895.8 trees/hectare, with an average of 433.3 (Figure 17e). Following treatments, no significant differences existed between treatment types for large seedlings of other species. Densities increased for control, T60, and T30, while densities decreased for burn, BT60, and BT30. The largest change occurred in the burn treatment, which decreased by 573.1 trees/hectare.

In the large seedling class, pretreatment tree densities of sugar maple, eastern redbud, and other species were commonly 10 times greater than the densities of oak, hickory, and black walnut. Generally, the burn, BT60, and BT30 treatments resulted in decreased density of large seedlings. Response to the T60 and T30 treatments varied more by species group, but often resulted in increased densities or decreases that were small in magnitude.
Figure 17. Large seedlings (dbh < 3.8 cm and height > 1 m) per hectare by species or species group in 2015 and 2016 for each of six treatment types (mean and one standard error). For 2016, the same superscript letter indicates no significant difference between treatment types.
Small Seedling Density

In 2015, total small seedling (height < 1 m) density ranged from 41,618 to 67,431 trees/hectare, with an average of 49,859. Before treatments, oak small seedling density ranged from 2,757 to 7,604 trees/hectare (Figure 18a). After treatments, only the density in the BT30 treatment was significantly different than the control. The largest decreases in the density of oak occurred in burn, BT60, and BT30 treatments. Oak density only increased in the control treatment.

The pretreatment density of sugar maple small seedlings ranged from 8,750 to 29,306 trees/hectare, with an average of 15,538 (Figure 18b). No 2016 densities were significantly different than control, however, the burn and T60 treatments were significantly different from each other. As with oak density, the largest differences between 2015 and 2016 occurred in the burn, BT60, and BT30 treatments, where sugar maples decreased following treatments. Sugar maple densities also decreased in T60 and T30 treatments, but by a smaller magnitude than the other treatments.

Prior to treatments, the density of small hickory and black walnut seedlings ranged from 1,632 to 3,576 trees/hectare, with an average of 2,538 (Figure 11c). Following treatments, only the burn treatment density was significantly different than the control. Generally, the magnitude of change for all treatments was not large for hickory and walnut.

The pretreatment density of small eastern redbud seedlings ranged from 5,035 to 7,743 trees/hectare, with an average of 6,046 (Figure 18d). The post-treatment density of
eastern redbud increased in all treatments except the BT60 treatment in 2016. Only the T60 treatment had a significantly different 2016 density than the control.

The pretreatment density of small ash seedlings ranged from 1,066 to 5,278 trees/hectare (Figure 18e). Following treatments, the density of ash in burn, BT60, and BT30 decreased, however, no treatments were significantly different than control. Prior to treatment, density of small elm seedling ranged from 5,590 to 11,493 trees/hectare. After treatments, only the BT60 and BT30 treatments had ash densities significantly different than the control.
Figure 18. Small seedlings (height < 1 m) per hectare by species or species group in 2015 and 2016 for each of six treatment types (mean and one standard error). For 2016, the same superscript letter indicates no significant difference between treatment types.
Canopy Openness

Prior to treatments, the canopy openness ranged from 5.1 to 7.4% with an average of 6.4% across all sites and treatment units (Table 5). Following treatments, percent canopy openness increased in all treatment units except for control, which showed a small decrease of 0.1% (Figure 19). The burn treatment had the smallest increase in canopy openness, with a 1.0% increase. The T30 treatment increased 10.0%, the greatest change of any of the treatments. The change in canopy openness was significantly different than control for all treatments except for the burn treatment. The change in percent canopy openness was not significantly different for all thinned treatments (T60, T30, BT60, and BT30), and all except the T60 treatment were significantly different than the burn treatment. The absolute difference in canopy openness ranked in accordance with treatment severity, that is T30 and BT30 had greater decreases than T60 and BT60, which in turn had greater decreases than burn and control treatments.
Table 5. Percent canopy openness (mean and one standard error (S.E.)) in each of six treatment types in 2015, 2016, and the difference between the two years (2016 – 2015). The same letters for 2016 or difference between 2016 and 2015 indicate no significant difference among pair-wise comparisons for each respective group (P < 0.05).

<table>
<thead>
<tr>
<th>Treatment</th>
<th>2015 Mean (S.E.)</th>
<th>2016 Mean (S.E.)</th>
<th>Difference (2016-2015) Mean (S.E.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control (C)</td>
<td>6.6 (0.7)</td>
<td>6.5 (0.5)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>-0.1 (0.5)&lt;sup&gt;a&lt;/sup&gt;</td>
</tr>
<tr>
<td>Burn (B)</td>
<td>6.5 (0.7)</td>
<td>7.4 (0.5)&lt;sup&gt;ab&lt;/sup&gt;</td>
<td>0.9 (0.5)&lt;sup&gt;ab&lt;/sup&gt;</td>
</tr>
<tr>
<td>Thin to 60 ft&lt;sup&gt;2&lt;/sup&gt; acre&lt;sup&gt;-1&lt;/sup&gt; basal area (T60)</td>
<td>7.4 (0.4)</td>
<td>12.6 (1.0)&lt;sup&gt;bc&lt;/sup&gt;</td>
<td>5.1 (1.0)&lt;sup&gt;bc&lt;/sup&gt;</td>
</tr>
<tr>
<td>Thin to 30 ft&lt;sup&gt;2&lt;/sup&gt; acre&lt;sup&gt;-1&lt;/sup&gt; basal area (T30)</td>
<td>6.4 (0.5)</td>
<td>16.4 (0.8)&lt;sup&gt;c&lt;/sup&gt;</td>
<td>10 (0.7)&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
<tr>
<td>Burn and Thin to 60 ft&lt;sup&gt;2&lt;/sup&gt; acre&lt;sup&gt;-1&lt;/sup&gt; basal area (BT60)</td>
<td>5.1 (0.8)</td>
<td>12.5 (1.1)&lt;sup&gt;c&lt;/sup&gt;</td>
<td>7.4 (0.7)&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
<tr>
<td>Burn and Thin to 30 ft&lt;sup&gt;2&lt;/sup&gt; acre&lt;sup&gt;-1&lt;/sup&gt; basal area (BT30)</td>
<td>6.2 (0.8)</td>
<td>15.0 (1.0)&lt;sup&gt;c&lt;/sup&gt;</td>
<td>8.8 (1.0)&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
</tbody>
</table>
Figure 19. Changes in percent canopy openness (mean and one standard error) by treatment type. The p-value is the level of significance from the ANOVA test for treatment effect, and the same letters on bars indicate no significant difference among pair-wise comparisons for each respective group (P < 0.05).
Ground Flora Treatment Effects

Prior to treatment, average percent cover of all herbaceous ground flora (<1 m tall) per plot ranged from 2.4 to 65.7%, and average percent cover of woody species ranged from 1.1 to 36.4%. Forbs were the most abundant physiognomic group prior to treatment, averaging 8.9% cover, followed by legumes (7.5%), vines (5.1%), trees (4.7%), graminoids (2.6%), and shrubs (1.7%).

Following treatments in 2016, mean percent cover of forbs increased in all treatment types (Figure 20a). The BT30 and BT60 treatments had the largest changes in percent cover with 11.5% and 9.9% increases, respectively. Mean percent cover of forbs in 2016 was not significantly different than the control for all treatment types except BT30, the most aggressive of the six treatments, which was significantly greater than the control (Table 6). The mean change in graminoid percent cover, including grasses and sedges, was less than 2% (increase or decrease) for all treatment types (Figure 20b). Graminoid cover decreased in burn and control, and increased in all thinned treatments (T60, T30, BT60 and BT30). The post-treatment percent cover of graminoids in T60, T30, and BT30 treatments was significantly different than control; burn and BT60 treatments were not significantly different than control or other treatments (Table 6). Mean legume cover increased in all treatments (Figure 20c). The greatest increase in legume cover occurred in the BT60 treatment (12.6%), which was also the greatest nominal change of any physiognomic group and treatment combination. The change in legume cover increased by at least 7% for all thinned treatments (BT60, T30, BT30, and T60), while the control and burn treatments only increased by about 3% or less. No
significant differences in percent legume cover existed between treatment types in 2016 (Table 6).

Mean tree cover increased in control, T60, T30, and BT30, while a decrease occurred in burn and BT60 (Figure 20d). The largest nominal change was a 4.2% increase in the T60 treatment; conversely, the smallest change occurred in the BT60 treatment, which decreased by only 0.2%. The post-treatment percent tree cover was not significantly different than control in all treatments; the only significant difference between any treatment types occurred between burn and T30 treatments (Table 6). As indicated above, shrub cover had the lowest mean percent cover of any physiognomic group prior to treatments. The changes to shrub cover were also small in magnitude, with all treatments resulting in less than 1% change (Figure 20e). The three treatments including fire (burn, BT60, and BT30) decreased in percent shrub cover, while control, T60, and T30 increased. The post-treatment percent shrub cover was not significantly different than control for any of the treatments (Table 6). However, percent shrub cover for the T30 treatment was significantly greater than in burn, BT60, and BT30 treatments. Mean percent cover of vines increased in control, T60, and T30 (Figure 20f). Vine cover in the T30 treatment had the greatest nominal change of any woody physiognomic group, with a 4.4% change in cover. The BT60 treatment also increased in vine cover, but only by 0.1%, while burn and BT30 treatments decreased slightly. Post-treatment vine cover was not significantly different between any of the treatment types (Table 6).
Figure 20. Changes in percent cover (mean and one standard error) for each of six treatment types by physiognomic life form of ground flora.
Table 6. Percent cover (mean and one standard error (S.E.)) of physiognomic groups for each of six treatment types in 2015 and 2016. For 2016, the same superscript letter indicates no significant difference between treatment types within a physiognomic group.

<table>
<thead>
<tr>
<th>Physiognomic Group</th>
<th>Year</th>
<th>Treatment</th>
<th>Control</th>
<th>Burn</th>
<th>T60</th>
<th>T30</th>
<th>BT60</th>
<th>BT30</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forb</td>
<td>2015</td>
<td>8.4 (1.7)</td>
<td>7.0 (1.2)</td>
<td>8.3 (1.3)</td>
<td>12.3 (1.6)</td>
<td>9.9 (2.0)</td>
<td>7.4 (1.0)</td>
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<tr>
<td></td>
<td>2016</td>
<td>12.1 (2.3)</td>
<td>13.7 (2.1)</td>
<td>12.9 (1.4)</td>
<td>21.0 (2.0)</td>
<td>19.7 (3.0)</td>
<td>18.9 (2.3)</td>
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<td></td>
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<td>a</td>
<td>a</td>
<td>a</td>
<td>a</td>
<td>b</td>
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<tr>
<td>Graminoid</td>
<td>2015</td>
<td>5.5 (2.7)</td>
<td>2.7 (1.6)</td>
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<tr>
<td></td>
<td>2016</td>
<td>4.4 (2.1)</td>
<td>2.0 (0.8)</td>
<td>3.1 (0.6)</td>
<td>2.9 (0.5)</td>
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<td>b</td>
<td>ab</td>
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<tr>
<td>Legume</td>
<td>2015</td>
<td>7.6 (1.3)</td>
<td>6.9 (1.0)</td>
<td>5.9 (1.0)</td>
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<td>7.0 (1.1)</td>
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<td>0.6 (0.2)</td>
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<td>6.3 (0.9)</td>
<td>10.5 (1.5)</td>
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Discussion

In this study, we analyzed treatment effects on stand structure and composition in Kansas oak woodlands. Despite the location of our study on the western boundaries of the eastern deciduous forest of the US, the initial stand conditions were similar to woodlands across the east that are threatened by woody encroachment (Nowacki and Abrams 2008, Hanberry et al. 2014a, 2014b). Once a common plant community in the central US, open woodlands are vanishing as a result of wide-spread fire suppression. At these sites in eastern Kansas, the lack of fire through the 20th century resulted in stands where both oak species and sugar maple dominated the nearly fully-stocked overstory, and oaks were far outnumbered by mesophytic competitors in the regeneration layer. The diameter distribution for the control reveals that while an even-aged cohort of larger oaks exists, smaller diameter classes have low relative oak density, especially the white oak group. It is likely that without management action, the stands at these sites were eventually destined to succeed to a community dominated by non-oak species.

The prescriptions for the thinned treatments aimed to reduce basal area to either 60 ft²/acre (13.8 m²/hectare) or 30 ft²/acre (6.9 m²/hectare). On average, the T30 and BT30 treatment units were marked close to intended residual basal area (Table 2). However, efforts to mark the T60 and BT60 treatment units resulted in average basal area below the desired prescription (e.g., the mean basal area of leave trees in the T60 treatment was 49.1 ft²/acre). As the full effect of the T60 and BT60 thinning treatments develop over time, we expect to possibly see these treatment units at a lower residual basal area than intended.
In the first growing season following the thinning treatments, mortality of treated trees was less than expected. Of the more than 2,000 trees that were girdled or girdled with application of herbicide, only 38% were dead in the first year. The effect of the treatments on mortality varied widely by species, with sugar maple having the greatest resistance to treatment. Sugar maple had the most individuals treated, 853, of which only 12.8% experienced mortality. One possible explanation for such a high proportion of sugar maple surviving is the timing of the treatments in relation to sap flow. Although best efforts were made to implement the treatments before heavy sap flow in sugar maple (first 3 weeks of January 2016), minimal sap flow may have occurred during treatments. No data collection occurred in the second growing season after treatments were implemented, but observations made in the summer of 2017 indicate that increased sugar maple mortality is occurring. In Indiana, Rathfon and Saunders (2013) observed 100% morality in sugar maple that received a single girdle and herbicide application after 3 years. We anticipate that with time, continued mortality will occur on treated trees.

As a result of low mortality in trees that were girdled, the post-treatment residual basal area targets were not met for the T60, T30, BT60, BT30 treatments. In fact, for the T30 and BT30 treatments, basal area in the first growing season after treatments was more than double the target basal area. In the Missouri Ozarks, Hanberry et al. (2014c) reconstructed historical stand metrics (1815-1850), including basal area, percent stocking, canopy cover, and tree density, and determined mean basal area in oak savannas and open woodlands was historically 8 m²/hectare and 14 m²/hectare, respectively. In this study, post-treatment basal areas for each of the 6 treatments were between 17.7 and 21.4
m²/hectare, indicating that open woodland conditions were not approximated for any of the treatment types in the first year.

We found post-treatment percent stocking ranged between 66.2 and 79.7% across all treatment types. Generally, stocking in woodlands is maintained between 30 and 70%, with open woodlands between 30% and the B-line (i.e., approximately 60% stocking) (Gingrich 1967, Dey et al. 2017, Hanberry et al. 2017). None of the treatments were effective at reducing percent stocking below B-level stocking (Gingrich 1967) (Figures 13-15). However, stocking in the T60, T30, BT60, and BT30 treatments were sufficiently reduced to be classified as a closed woodland structure.

Percent stocking and basal area in the burn only treatment resulted in almost no change from pretreatment levels. Other studies have also found that a single prescribed burn has little effect on overstory stocking and basal area (Arthur et al. 1998, Hutchinson et al. 2005, Albrecht and McCarthy 2006, Blankenship and Arthur 2006, Kinkead et al. 2013). A single, low intensity fire is often ineffective at inducing mortality even for fire-sensitive species that reach a threshold size (Dey and Hartman 2005), and reductions of about 5% in stand basal area have commonly been reported (Dey and Fan 2009, Hutchinson et al. 2005).

Post-treatment differences in the density of the sapling size class varied by species and treatments. The effects of the thinning treatment were especially evident for this size class. Sugar maple showed large decreases in the T60, T30, BT60, and BT30 treatments. It is worth noting that although overall mortality of girdled sugar maples was low, many stems in this size class were completely felled (versus girdled) at the discretion of the contractor. For sugar maple, no significant differences existed between thin only and burn
and thin treatments, and the density in the burn treatment remained nearly static from pre- to post-treatment. Therefore, we can conclude the thinning treatment was the more effective tool in the reduction of the sapling size class. This result was not unexpected, as our Chapter III results found sugar maple survival probability following a single dormant burn reached 99% at basal diameters of only 3 cm, despite this species being classified as pyrophobic (fire-sensitive) (Nowacki and Abrams 2014). The lower declines for oaks, hickory, and black walnut in thinned treatments were a result of the treatment prescription, as these species were preferentially selected as leave trees. Furthermore, the thinning prescription explicitly did not treat oaks that were less than 12.7 cm.

Reductions to overstory and midstory basal area, stocking, and density in stands invaded by shade-tolerant, mesophytic species are necessary to move toward woodland structure and composition (Nelson 2010, Hanberry et al. 2014). The lack of significant changes to the overstory and midstory strata in the burn only treatments of our study highlight the importance of complementary fire and thinning treatments. Nowacki and Abrams (2008) conceptualize alternative stable states of fire-adapted mesic and xeric upland communities, and conclude that removal of the perturbation of fire shifts communities toward a mesophytic hardwood-dominated state. Although xeric upland communities are more resilient to shifts toward mesophication (Nowacki and Abrams 2008), the lack of fire disturbance at our sites has resulted in the encroachment of mesophytic woody species that have surpassed size thresholds that are vulnerable to mortality from fire.

Consistent with the stand structural metrics already discussed, the post-treatment canopy openness changes ranked in accordance with treatment severity, with the T30 and
BT30 resulting in the most open canopy. Much of the reduction in canopy openness in the thinned treatments can be attributed to the reduced density of sugar maple and other mesophytic species in the sapling layer. The burn only treatment did not change canopy openness. Alexander et al. (2008) and Hutchinson et al. (2012) found only modest reductions in the canopy openness even after three prescribed fires. Canopy openness of 35 and 20% is typical of open and closed woodlands, respectively (Hanberry et al. 2014c). In 2016, canopy openness across all treatments was between 6.5 and 16.4%, indicating that post-treatment canopy openness still reflects closed canopy forest conditions.

The rates of photosynthesis in oak regeneration increase with increasing light intensity up to approximately one-third full sunlight, after which further light intensity increases do not provide additional benefits (Johnson et al. 2009). Blizard et al. (2013) developed relationships between percent stocking, canopy closure, and photosynthetically active radiation (PAR) in oak-hickory stands in the Missouri Ozarks, to determine stand structural attributes that are adequate for the development of oak regeneration. Based on the canopy openness and percent stocking in this study, the amount of above-canopy PAR reaching the understory was between 25 and 30%, with the lower stocking in our thinned treatments being closer to 30% PAR (Blizzard et al. 2013, Dey and Kabrick 2015, Dey et al. 2017). Therefore, the requirements of one-third full sunlight for successful oak regeneration are likely being met in the stands where the T60, T30, BT60, and BT30 treatments were implemented.

While light conditions are becoming more favorable for oak regeneration, pretreatment oak seedling densities were far lower than mesophytic species, such as
eastern redbud and sugar maple, representing a large competitive disadvantage for oak at these sites before treatments were implemented. Seedling response was highly variable between species and treatment types. The changes in oak seedling densities were generally small in magnitude, although small seedlings showed a decline in burn only and burn and thin treatments. Decreases in the densities of sugar maple, the most abundant species in the understory, were much greater than decreases for oak, and resulted in sugar maple small seedlings occurring at lower post-treatment densities than oak in the burn only and thin and burn treatments. Declines in maple seedling densities following a single prescribed fire were also documented in Ohio (Albrecht and McCarthy 2006) and Kentucky (Alexander et al. 2008), however, in both of those studies maple densities eventually returned to pretreatment levels, and oak was ultimately not at a competitive advantage. In Missouri, Knapp et al. (2015) found that annual prescribed burning for 60 years nearly eliminated all seedlings and saplings. Repeated, periodic burning is likely necessary to exhaust the sprouting capability of oak-competitors such as maple. For the burn, BT60, and BT30 treatments, reductions in the density of large eastern redbud seedlings coincided with increases in small seedling densities. This is indicative of large seedlings experiencing shoot dieback and sprouting, as redbud is a vigorous sprouter.

We found significant ground flora response despite thinning treatments not reducing basal area to intended targets. The change in forb and legume cover increased with the severity of the treatments, suggesting these two groups are responding positively to increases in light as well as the effects of the prescribed burn. The limiting factor for graminoid cover appears to be light availability as the cover of this group increased in thinned treatments and decreased in the burn only treatment. In the Missouri Ozarks,
Zenner et al. (2006) found forb and graminoid cover was directly proportional to increasing harvest intensities; however, legume cover in that study decreased due to competition with increasing forb cover. Other studies have also found increases in herbaceous plant cover and species diversity corresponding with decreases in canopy cover, especially for heliophytic woodland indicator species (Peterson et al. 2007, Peterson and Reich 2008 Kinkead et al. 2013). The positive response of herbaceous cover to prescribed fire has been linked to the reductions of competing woody species and to increased germination following reductions to the litter layer (Hutchinson et al. 2005, Stambaugh et al. 2006, Bowles et al. 2007, Peterson et al. 2007).

In this study, the general pattern observed in the woody functional groups was an increase in cover in thin only treatments and a decrease in all treatments involving prescribed fire. Cover of trees deviated from this pattern and this physiognomic group may not be a good indicator of treatment type in the first growing season after treatments. Initial decreases in the cover of trees, vines, and shrubs was expected, as fire often induces topkill of stems in the understory. Albrecht and McCarthy (2006) found similar results, with shrub and vine cover decreasing in the first growing season following burn only and thin and burn treatments; however, they also found that shrubs recovered to pretreatment cover within 4 years in thinned treatments. Alternatively, following two prescribed burns in Missouri, Kinkead et al. (2013) found shrub and vine cover increased significantly in the harvested and burn and harvested stands, while decreasing in the burn only. Following only one or two prescribed fires, it is possible that increased availability of light from thinning (or harvesting) may be a more important factor in the cover of
shrubs and vines, however, repeated burning would likely continue to reduce shrub and vine cover (Bowles et al. 2007, Peterson et al. 2007, Burton et al. 2010).

Taken together, the results of this study reveal the benefits and the necessity of burning and thinning treatments for restoring characteristic woodland structure and composition. A single dormant season prescribed burn was effective at reducing large and small seedling densities of sugar maple and other oak-competitors and increased forb and legume cover in the understory. However, the burn only treatment had no effect on overstory stand metrics, including basal area, tree density, percent stocking, and canopy openness. Thinning of the overstory and midstory in combination with prescribed fire resulted in similar effects to seedling densities and ground flora cover as the burn only treatment, but also created a reduced sapling layer and greater canopy openness. We found low mortality following a single girdle and herbicide application after one year, especially for sugar maple. As a result, the reduction in overstory basal area did not meet our intended targets. Observations in the second growing season (2017) strongly indicate that the effectiveness of thinning treatments on the mortality of treated trees will continue to increase. These initial results are for the first year following treatments and over time we expect vegetation dynamics to continually evolve in response to treatments.
Chapter III: Fire Effects on Oak Woodland Advance Regeneration

Introduction

Prior to Euro-American settlement, fire was an important driver in shaping plant communities in the central United States (Abrams 1992, Anderson et al. 1999, Nelson 2010). In the transitional region between the US eastern deciduous forests and the Great Plains, diverse assemblages of forest, woodland, prairie, and savanna historically existed (Braun 2001, Nuzzo 1986, Anderson et al. 1999, Nelson 2010). While topoedaphic, fuel, and climate factors play an important role, the influence of anthropogenic fire has been credited as an essential element in the development and perpetuation of open plant communities such as savannas and woodlands. (Abrams 1992, Stambaugh et al. 2006a, Guyette et al. 2006, McEwan et al. 2007, Stambaugh et al. 2011, Brose et al. 2013a, Stambaugh et al. 2014). The variation of fire frequency and severity has been shown to be heavily influenced by human population density, culture, and land use; furthermore, the development and succession of plant communities reflect the temporal and spatial shifts of these variables (Abrams 1992, Delcourt and Delcourt 1997, Anderson et al. 1999, Guyette et al. 2006, Stambaugh et al. 2014).

Across the central US, oaks (Quercus spp.) were historically a principal component of savannas and woodlands (Nuzzo 1986, Anderson et al. 1999, Nelson 2010, Hanberry and Nowacki 2016). The link between landscape-level fire and historical oak dominance is a widely discussed hypothesis with a growing body of support (Abrams 1992, Arthur et al. 2012, Brose et al. 2001, Nowacki and Abrams 2008). Oaks have adapted to be fire tolerant through physiological adaptations including thick bark, large
shoot-to-root ratios, ability to sprout after shoot dieback, and wound compartmentalization (Smith and Sutherland 1999, Johnson et al. 2009). Fire as a natural disturbance has perpetuated oak communities by providing necessary canopy openings, as low tolerance to shade is a common trait of oaks (Abrams 1992, Johnson et al. 2009). Frequent fire regimes also promoted oak domination by excluding oak-competitors that were fire sensitive.

As fire suppression became a regular practice through the 20th century, major shifts in plant community structure and composition resulted, with many historically open communities transitioning into closed canopy forests (Nowacki and Abrams 2008, Hanberry et al. 2014a, Hanberry et al. 2014b). This densification of woodlands and forests is often the result of fire-sensitive, shade tolerant, and mesophytic species encroaching into areas where fire previously constrained their existence. Throughout the eastern deciduous forests, species such as red maple (Acer rubrum) and sugar maple (Acer saccharum) have increased in abundance, causing major structural, compositional, and functional changes to these communities (Nowacki and Abrams 2008). This process of “mesophication” creates a positive feedback loop that further inhibits the natural disturbance of fire by promoting cool, damp, and shaded microenvironmental conditions, as well as decreasing the flammability of fuel beds (Nowacki and Abrams 2008).

In many areas of the eastern deciduous forests, a severe lack of oak regeneration has been reported (Abrams 2003, Moser et al. 2006, Hanberry 2013). For managers seeking to maintain or restore oak-dominated communities, prescribed fire has become a common tool to combat encroachment of pyrophobic (fire-sensitive) species and to promote regeneration of pyrophilic (fire-tolerant) oaks. Understanding the effects of fire
on advance regeneration of oak species and their competitors is an important first step in determining the role of prescribed fire in regenerating and restoring upland oak ecosystems. Our study aims to understand how a single dormant season prescribed fire effects advance regeneration of two oaks species, chinkapin oak (*Quercus muehlenbergii*) and black oak (*Quercus velutina*), as well as bitternut hickory (*Carya cordiformis*), sugar maple (*Acer saccharum*), and eastern redcedar (*Juniperus virginiana*). Specifically the objectives of this study are to: 1) determine percent mortality and percent shoot dieback with sprouting for 5 different species 2) analyze relationships between initial stem size (basal diameter and height) and probability of survival, 3) determine the effect of fire temperature on the probability of survival, and 4) compare how mortality, sprouting, and survival probability differ between the 5 species.
Methods

Study Sites and Design

This study was conducted at Marais des Cygnes (MDC) and La Cygne State Wildlife Areas (LCWA) in Linn and Miami Counties of eastern Kansas. See Chapter II for a complete description of the study sites.

Data Collection

Within the established burn and control treatment units described in Chapter II, individuals of 5 tree species were permanently tagged in the summer of 2015, before the prescribed fire treatment. Species were selected based on a range of recognized fire tolerance designations (Nowacki and Abrams 2014), with chinkapin oak, black oak, and bitternut hickory defined as fire-tolerant, and sugar maple and eastern redcedar recognized as fire-sensitive. Additionally, these species were selected based on their occurrence and abundance at all three sites (MDC, LC1, LC2). Tagging occurred on seedling and sapling sized trees (<11.4 dbh), and best efforts were made to tag individuals evenly across this range of sizes. A total of 725 individuals were tagged (approximately equal proportions for the five species) across the burn and control treatment units.

At each individual a wire pigtail stake was placed in the ground 5 to 10 cm to the north of the stem, and a stainless steel tag with a unique number was attached. Most tagged trees were within the boundaries of the 0.08-hectare vegetation plots described in Chapter II, but some were located up to 50 m outside of plots to obtain the desired number of individuals per species. The plot and location within/near each plot were
recorded so that individuals could be relocated. We selected only seedlings and saplings that existed in a continuous fuel bed, such as in herbaceous or tree leaf litter, to ensure tagged stems were not in locations that could not carry a surface fire (e.g., on rocky outcrops). The initial height, stem basal diameter 2.5 cm above the ground, and dbh (if present) were recorded for all tagged individuals. Height was measured to nearest centimeter for individuals up to 2 m tall, but estimated to nearest meter for individuals >2 m tall.

In the summer of 2016, tagged individuals were relocated to determine survival following prescribed fire. An individual was dead if the shoot was killed and no sprouting was evident, indicating complete death of all portions of the tree. The percent mortality for each species was calculated by dividing the number of dead individual by the total number of tagged individual for each respective species. Similarly, a percent shoot dieback was calculated for each species. Shoot dieback is defined as the death of only the aerial portion of the tree, with the rootstock still alive. This was determined by the presence of sprouting after the aerial portion of an individual was killed. If an individual resprouted following shoot dieback, the height of the tallest sprout was measured. Total damage is the sum of mortality and shoot dieback (Dey and Hartman 2005).

Prescribed fire treatment

A dormant season prescribed fire was applied at the MDC site on March 5, 2016, and at the LC1 and LC2 sites on March 17, 2016. All fires occurred before leaf emergence of woody species. The daytime air temperature on burn days ranged from 11 to 18°C, relative humidity ranged from 25 to 45%, and wind speed ranged from 4 to 16 km/hour. No fuel moisture measurements were taken, but fuels were observed to be drier
at the MDC site than LC1 or LC2. Flame lengths varied, but observed values averaged 0.3 to 0.9 m at all three sites. Rates of spread were observed between 39 and 138 cm/min.

At the MDC site the prescribed fire totaled approximately 107 hectares. The MDC site is located along a ridge and managers at the wildlife area included the entire hill as the burn unit. The burns at the LC1 and LC2 sites were approximately 15 hectares each. Fuels at all burns were mainly leaf litter, although cool and warm season grasses also occurred in areas adjacent to study sites.

Fire temperature was measured using Tempilaq brand temperature-sensitive paints applied to aluminum tags. The methods for paint tag use as temperature recorders were adapted from Iverson et al. 2004. Ten different paints were applied to each tag, and the temperatures at which they liquefied were rated as: 79°C, 121°C, 149°C, 177°C, 204°C, 232°C, 260°C, 288°C, 343°C, and 427°C. The paint tags were covered with a blank tag to avoid char and soot accumulation on paint surface. The paint tags were suspended 15 cm above the litter layer using a wire pin flag, and were deployed in the field just prior to prescribed fire (day of burn or day before). Tags were deployed at the center of each 0.08-hectare plot and center of the four 0.004-hectare subplots in the burn, BT60, and BT30 treatment units described in Chapter II (five tags/plot in 54 plots = 270 tags deployed). After each fire, paint tags were recollected and each tag was evaluated by two people independently (a third observer evaluated tags when a discrepancy occurred between original two evaluators). The highest temperature paint that melted was recorded as the maximum fire temperature for that location and used in the analyses that follow.
Data Analysis

For each of the five species listed above, logistic regression was used to model the probability of survival of advance regeneration in the burn treatment units from 2015 (pre-burn) to 2016 (post-burn) (Proc Logistic, SAS version 9.4, SAS Institute, Cary, NC). Independent variables were fire temperature, initial height, and initial basal diameter. Fire temperature was calculated as the mean temperature of the five temperature paint tags for each plot. The binary dependent variable in this analysis was the fate of each individual tagged seedling or sapling (i.e., alive or dead). A priori models for each species included: a null (intercept-only), a single independent variable model (basal diameter, height, and fire temperature), and all possible models with multiple independent variables, including two-way interactions (Table 11). Logistic regression models were also developed for the tagged advance regeneration in the control treatment based on initial basal diameter, height, and their interaction as a baseline to compare the probability of survival in the burn treatment results.

An information-theoretic approach was used to model the survival probability for advance regeneration (Burnham and Anderson 2002). The models were ranked using Akaike's Information Criterion (AIC); ΔAIC and Akaike weight (w_i) were calculated from AIC values and used to determine the best models. Models containing lower ΔAIC, greater w_i, and simpler models with ΔAIC < 2 had the most support for being the best model. All models were evaluated using the Hosmer and Lemeshow goodness-of-fit tests; all models presented were insignificant (p >0.05), indicating goodness-of-fit. Model significance (α=0.05) was determined for the test of the global null hypothesis that all β_i
are equal to zero using the likelihood ratio (-2 log $L$) statistic (Allison 1999, Dey and Hartman 2005).

**Results**

*Mortality and Sprouting of Advance Regeneration*

Mean fire temperatures ranged from 70.5 to 141.2°C across the three study sites (Table 7). The minimum temperature measured was 0°C in all treatment types at all sites except the BT30 treatment, which had a minimum temperature of 79°C. Maximum temperatures recorded at a site ranged from 204°C to 427°C. Only three paint tags reached a temperature of 343°C or greater, and all occurred in the BT60 treatment at LC2.

Table 7. Mean (and one standard error), minimum, and maximum fire temperatures by site and treatment type.

<table>
<thead>
<tr>
<th>Site</th>
<th>Treatment</th>
<th>Mean Fire Temperature (°C)</th>
<th>Standard Error</th>
<th>Minimum Fire Temperature (°C)</th>
<th>Maximum Fire Temperature (°C)</th>
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</thead>
<tbody>
<tr>
<td>LC1</td>
<td>B</td>
<td>114.4</td>
<td>13.9</td>
<td>0</td>
<td>260</td>
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<td></td>
<td>BT30</td>
<td>95.2</td>
<td>12.3</td>
<td>0</td>
<td>204</td>
</tr>
<tr>
<td></td>
<td>BT60</td>
<td>90.7</td>
<td>13.6</td>
<td>0</td>
<td>288</td>
</tr>
<tr>
<td>LC2</td>
<td>B</td>
<td>75.3</td>
<td>13.5</td>
<td>0</td>
<td>232</td>
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<tr>
<td></td>
<td>BT30</td>
<td>70.5</td>
<td>11.3</td>
<td>0</td>
<td>204</td>
</tr>
<tr>
<td></td>
<td>BT60</td>
<td>97.6</td>
<td>20.8</td>
<td>0</td>
<td>427</td>
</tr>
<tr>
<td>MDC</td>
<td>B</td>
<td>89.2</td>
<td>11.4</td>
<td>0</td>
<td>204</td>
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<tr>
<td></td>
<td>BT30</td>
<td>141.2</td>
<td>6.5</td>
<td>79</td>
<td>232</td>
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<td></td>
<td>BT60</td>
<td>81.5</td>
<td>10.3</td>
<td>0</td>
<td>204</td>
</tr>
</tbody>
</table>

In the first growing season following the 2016 dormant season prescribed burns, the total damage to advance regeneration ranged from 38.7 to 69.4% for all five species (Table 8). Chinkapin oak had the greatest total damage, with 72% of tagged stems
damaged. Conversely, sugar maple had the lowest total damage (38.7%). Most of the
damage to black oak, chinkapin oak, and bitternut hickory was attributed to shoot
dieback, where individuals resprouted following the injury. Sugar maple experienced
nearly equal proportions of mortality and shoot dieback, while eastern redcedar
experienced no shoot dieback and 44% mortality. Mortality was less than 4% for oaks
and bitternut hickory.

Table 8. 2015 pre-burn mean basal diameter (BD) and height (HT) for all tagged
advance regeneration in the burn treatment units, and percent damage following 2016
prescribed burn. Height and basal diameter ranges are presented in parentheses.

<table>
<thead>
<tr>
<th>Species</th>
<th>N</th>
<th>BD (cm)</th>
<th>HT (m)</th>
<th>Mortality (%)</th>
<th>Shoot dieback with sprouts (%)</th>
<th>Total Damage (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Black Oak</td>
<td>70</td>
<td>6.0 (0.2-23.0)</td>
<td>3.5 (0.1-10)</td>
<td>2.9</td>
<td>42.9</td>
<td>45.7</td>
</tr>
<tr>
<td>Bitternut Hickory</td>
<td>62</td>
<td>4.2 (0.2-23.0)</td>
<td>2.7 (0.1-11)</td>
<td>3.2</td>
<td>66.1</td>
<td>69.4</td>
</tr>
<tr>
<td>Chinkapin Oak</td>
<td>74</td>
<td>4.0 (0.1-34.0)</td>
<td>1.5 (0.07-10)</td>
<td>2.7</td>
<td>68</td>
<td>72</td>
</tr>
<tr>
<td>Sugar Maple</td>
<td>75</td>
<td>7.1 (0.1-26.8)</td>
<td>3.8 (0.07-11)</td>
<td>18.7</td>
<td>20</td>
<td>38.7</td>
</tr>
<tr>
<td>Eastern Redcedar</td>
<td>81</td>
<td>4.7 (0.1-21.6)</td>
<td>2.1 (0.04-8)</td>
<td>44.4</td>
<td>0</td>
<td>44.4</td>
</tr>
</tbody>
</table>

Mortality is defined as the complete death of the individual and shoot dieback as the death of only
the aerial portion of the tree. Total damage is the sum of mortality and shoot dieback (Dey and
Hartman 2005).

The size of advance regeneration that experienced mortality between 2015 and
2016 varied by species, with mean pre-burn basal diameter ranging from 0.4 to 1.2 cm
and mean height ranging from 14 to 66 cm (Table 9). The maximum basal diameter of
individuals that were dead following the prescribed burn was least in oaks and hickory
(0.6 and 0.7 cm, respectively). The maximum basal diameter of sugar maple and eastern
redcedar was 2.3 cm and 3.9 cm, respectively. Similarly, maximum heights were lower
for oaks and hickory (16 and 56 cm, respectively) than for sugar maple and eastern
redcedar (172 and 200 cm, respectively).
Individuals that experienced shoot dieback with sprouts following prescribed burn ranged from 0.7 to 2.3 cm in mean basal diameter and 45.9 to 191.2 cm in mean height (Table 9). No eastern redcedar individuals experienced shoot dieback with sprouts. The minimum basal diameter of oaks and hickory ranged from 0.1 and 0.3 cm, respectively. The minimum basal diameter for sugar maple was 0.9 cm. The minimum heights of oaks and hickory that sprouted after shoot dieback were between 7 and 16 cm, respectively, while the minimum height of sugar maple was 41 cm.

Similar low mortality rates were found in the control treatment for chinkapin oak, black oak, and sugar maple, which ranged from 1.6 to 3.5% (Table 10). Conversely, sugar maple and eastern redcedar had 0% mortality from 2015 to 2016 in the control treatment, markedly different than the mortality for those species in the burn treatments.
Table 9. The mean, minimum, and maximum initial basal diameter (BD) and height (HT) of advance regeneration that experienced mortality or shoot dieback with sprouts following the 2016 prescribed burn.

<table>
<thead>
<tr>
<th>Species</th>
<th>N</th>
<th>BD mean (cm)</th>
<th>BD minimum (cm)</th>
<th>BD maximum (cm)</th>
<th>HT mean (cm)</th>
<th>HT minimum (cm)</th>
<th>HT maximum (cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dead</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black Oak</td>
<td>2</td>
<td>0.4</td>
<td>0.2</td>
<td>0.6</td>
<td>14</td>
<td>12</td>
<td>16</td>
</tr>
<tr>
<td>Bitternut Hickory</td>
<td>2</td>
<td>0.5</td>
<td>0.2</td>
<td>0.7</td>
<td>25.5</td>
<td>11</td>
<td>40</td>
</tr>
<tr>
<td>Chinkapin Oak</td>
<td>2</td>
<td>0.5</td>
<td>0.4</td>
<td>0.6</td>
<td>38.5</td>
<td>21</td>
<td>56</td>
</tr>
<tr>
<td>Sugar Maple</td>
<td>14</td>
<td>0.6</td>
<td>0.1</td>
<td>2.3</td>
<td>33.7</td>
<td>7</td>
<td>172</td>
</tr>
<tr>
<td>Eastern Redcedar</td>
<td>36</td>
<td>1.2</td>
<td>0.1</td>
<td>3.9</td>
<td>66</td>
<td>4</td>
<td>200</td>
</tr>
</tbody>
</table>

Shoot dieback with Sprouts

<table>
<thead>
<tr>
<th>Species</th>
<th>N</th>
<th>BD mean (cm)</th>
<th>BD minimum (cm)</th>
<th>BD maximum (cm)</th>
<th>HT mean (cm)</th>
<th>HT minimum (cm)</th>
<th>HT maximum (cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Black Oak</td>
<td>32</td>
<td>1</td>
<td>0.3</td>
<td>3.3</td>
<td>90.5</td>
<td>11</td>
<td>300</td>
</tr>
<tr>
<td>Bitternut Hickory</td>
<td>41</td>
<td>2.3</td>
<td>0.3</td>
<td>22.2</td>
<td>158.6</td>
<td>16</td>
<td>900</td>
</tr>
<tr>
<td>Chinkapin Oak</td>
<td>51</td>
<td>0.7</td>
<td>0.1</td>
<td>3</td>
<td>45.9</td>
<td>7</td>
<td>200</td>
</tr>
<tr>
<td>Sugar Maple</td>
<td>15</td>
<td>1.9</td>
<td>0.9</td>
<td>4.1</td>
<td>191.2</td>
<td>41</td>
<td>500</td>
</tr>
<tr>
<td>Eastern Redcedar</td>
<td>0</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

Table 10. 2015 mean basal diameter (BD) and height (HT) for all tagged advance regeneration in the control treatment units and percent mortality in 2016.

<table>
<thead>
<tr>
<th>Species</th>
<th>N</th>
<th>BD (cm)</th>
<th>HT (m)</th>
<th>Mortality (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Black Oak</td>
<td>61</td>
<td>7.2</td>
<td>4.2</td>
<td>1.6</td>
</tr>
<tr>
<td>Bitternut Hickory</td>
<td>86</td>
<td>5.4</td>
<td>3.4</td>
<td>3.5</td>
</tr>
<tr>
<td>Chinkapin Oak</td>
<td>86</td>
<td>6.3</td>
<td>3.4</td>
<td>3.5</td>
</tr>
<tr>
<td>Sugar Maple</td>
<td>66</td>
<td>6.9</td>
<td>3.6</td>
<td>0</td>
</tr>
<tr>
<td>Eastern Red Cedar</td>
<td>38</td>
<td>2.5</td>
<td>1.2</td>
<td>0</td>
</tr>
</tbody>
</table>

Survival Probability

In this study, initial basal diameter was significantly (α=0.05) related to the probability of survival after one fire for all tagged species except chinkapin oak (Table 5). Additionally, no models were significant for predicting survival probability of chinkapin oak (Table 11 and 12). For black oak, the only single predictor model with a
valid model fit included basal diameter; all other models did not have a maximum
likelihood estimate and the validity of the models was questionable. This questionable
validity of some black oak model subsets resulted from a quasi-complete separation of
data points, which occurs when an independent variable is a perfect predictor of the
dependent variable. Height was a significant predictor of probability of survival for
bitternut hickory, sugar maple, and eastern redcedar. The single independent variable
model with height as predictor had the most support as the best overall model for
bitternut hickory (\(w_i=0.24\)) and sugar maple (\(w_i=0.42\)). As the single predictor for
survival probability, fire temperature was not significant for any of the species except
eastern redcedar, however, this model had very low support for being a best model for
that species (\(w_i<0.00001\)). Eastern redcedar was the only species that had all models
significant for the test of the global null hypothesis that all \(\beta_i\) are equal to zero using the
likelihood ratio, while sugar maple was significant for all models except the model with
fire temperature as the sole independent variable.

While single variable models were the best for predicting probability of survival
for bitternut hickory and sugar maple, the multiple logistic regression model including
height and fire temperature had the most support for black oak (\(w_i=0.23\)) and eastern
redcedar (\(w_i=0.17\)) (Table 11). That same subset model was also among the best for sugar
maple (\(w_i=0.16\)) and bitternut hickory (\(w_i=0.14\)).

The ranking of models containing at least three independent variables for
predicting probability of survival varied by species and model subsets (Table 11). The
model with fire temperature, basal diameter, and height had moderate support for black
oak (\(w_i=0.51\)) and eastern red cedar (\(w_i=0.23\), but low support for bitternut hickory
(wi=0.06) and sugar maple (wi=0.06). The model containing fire, basal diameter, and their interaction also had moderate support for being a best for predicting survival probability for eastern redcedar (wi=0.21).

For black oak, bitternut hickory, sugar maple, and eastern redcedar, the probability of survival increased with increasing diameter (Figure 21). Probability of survival for black oak increased from 50.6% to 99.8% as basal diameter increased from 0.2 to 1.0 cm. Probability of survival was 73.0% and 99.8% for bitternut hickory stems with basal diameters of 0.22 and 1.5 cm, respectively. Survival probability of sugar maple was 8.5% and 99.1% for basal diameters of 0.1 and 3.0 cm, respectively. Probability of survival for eastern redcedar increased from 15.8 to 99.2% as basal diameter increased from 0.1 to 9.0 cm. At the smallest basal diameters, black oak was most likely to survive after fire, followed in order by bitternut hickory, sugar maple, and eastern redcedar.

The probability of survival increased with increasing height for bitternut hickory, sugar maple, and eastern redcedar (Figure 22). At a height of 11 cm the survival probability of hickory was 70.2%, increasing to a probability of 99.1% at 70 cm tall. Probability of survival for sugar maple increased from 8.5% to 99.1% as height increased from 7 cm to 2.4 m. Survival probability of eastern redcedar was 6.9% and 99.1% for heights of 4 cm and 3.5 m, respectively. At the shortest heights, bitternut hickory was most likely to survive after, followed by sugar maple and eastern redcedar: the same ranking as for basal diameter, but excluding black oak, which did not have a valid model for this predictor variable.
In the control treatment, many of the survival probability models were not significant. Models with single predictor variables (either basal diameter or height) for black oak were not significant and the maximum likelihood estimate may not exist for the full model. Neither basal diameter nor height were significant in models for chinkapin, however, the full model was significant. All models were significant in predicting survival probability of bitternut hickory. Sugar maple and eastern redcedar models could not be developed for advance regeneration in the control treatment because all observations had the same response (i.e., all individuals were alive in 2016).
Table 11. Ranking of survival probability models for advance regeneration following prescribed burn treatment.

<table>
<thead>
<tr>
<th>Model</th>
<th>Black Oak</th>
<th>Chinkapin Oak</th>
<th>Sugar maple</th>
<th>Bitternut hickory</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>log L</td>
<td>p</td>
<td>K</td>
<td>AIC</td>
</tr>
<tr>
<td>Fire BD HT BDHT</td>
<td>17.0</td>
<td>0.70</td>
<td>4</td>
<td>25.0</td>
</tr>
<tr>
<td>BD HT BDHT</td>
<td>17.1</td>
<td>0.74</td>
<td>4</td>
<td>25.1</td>
</tr>
<tr>
<td>Fire BD BD*Fire</td>
<td>17.5</td>
<td>0.83</td>
<td>4</td>
<td>25.5</td>
</tr>
<tr>
<td>Fire HT HT*Fire</td>
<td>16.9</td>
<td>0.68</td>
<td>4</td>
<td>24.9</td>
</tr>
<tr>
<td>Fire BD</td>
<td>17.1</td>
<td>0.54</td>
<td>3</td>
<td>23.1</td>
</tr>
<tr>
<td>Fire HT</td>
<td>17.5</td>
<td>0.65</td>
<td>3</td>
<td>23.5</td>
</tr>
<tr>
<td>BD HT</td>
<td>16.9</td>
<td>0.48</td>
<td>3</td>
<td>22.9</td>
</tr>
<tr>
<td>Fire</td>
<td>18.4</td>
<td>0.91</td>
<td>2</td>
<td>22.4</td>
</tr>
<tr>
<td>BD</td>
<td>17.2</td>
<td>0.27</td>
<td>2</td>
<td>21.2</td>
</tr>
<tr>
<td>HT</td>
<td>17.6</td>
<td>0.37</td>
<td>2</td>
<td>21.6</td>
</tr>
<tr>
<td>NULL</td>
<td>18.4</td>
<td>1</td>
<td>1</td>
<td>20.4</td>
</tr>
</tbody>
</table>

Log $L$ is the likelihood ratio statistic used to test the global null hypothesis that all $\beta_i$ are equal to zero; the p-value shown corresponds to the likelihood ratio statistic.

$K$ is the number of parameters in each model and includes the intercept and each independent variable.

* Maximum likelihood estimate may not exist for this model. Validity of model fit is questionable. These models were not considered in the comparison and ranking of models.
Table 12. Significant survival probability models for advance regeneration following prescribed fire treatment. Models are listed in their ranked order and parameter estimates are presented with standard error in parentheses.

<table>
<thead>
<tr>
<th>Species models</th>
<th>Intercept (BD)</th>
<th>Basal Diameter (BD)</th>
<th>Height (HT)</th>
<th>Fire Temperature (Fire)</th>
<th>BD*HT interaction (BDHT)</th>
<th>Fire*BD interaction</th>
<th>Fire*HT interaction</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chinkapin oak</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>No significant models</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black oak</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT</td>
<td>2.2412 (4.882)</td>
<td></td>
<td>0.2435 (0.195)</td>
<td>-0.0268 (0.028)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD + HT</td>
<td>8.9240 (6.119)</td>
<td>-20.7395 (13.956)</td>
<td>0.6000 (0.395)</td>
<td>-0.0473 (0.032)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD</td>
<td>1.8698 (3.388)</td>
<td>6.0756 (4.902)</td>
<td></td>
<td>-0.0119 (0.011)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT + Fire*HT</td>
<td>7.3787 (14.303)</td>
<td>-0.0133 (0.480)</td>
<td>-0.0477 (0.080)</td>
<td></td>
<td></td>
<td></td>
<td>0.0009 (0.002)</td>
</tr>
<tr>
<td>BD</td>
<td>-1.2471 (2.265)</td>
<td>7.9501 (5.440)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD + Fire*BD</td>
<td>6.8309 (3.840)</td>
<td>-4.3332 (5.493)</td>
<td>-0.0357 (0.026)</td>
<td></td>
<td></td>
<td></td>
<td>0.0512 (0.048)</td>
</tr>
<tr>
<td>Bitternut hickory</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>HT</td>
<td>0.1416 (1.694)</td>
<td></td>
<td>0.0652 (0.056)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BD</td>
<td>0.0874 (1.790)</td>
<td>4.1279 (3.410)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT</td>
<td>2.2064 (2.894)</td>
<td></td>
<td>0.0700 (0.064)</td>
<td>-0.0118 (0.013)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD</td>
<td>2.1637 (3.002)</td>
<td>4.1501 (3.657)</td>
<td></td>
<td>-0.0112 (0.012)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BD + HT</td>
<td>0.2445 (1.844)</td>
<td>-0.9948 (6.311)</td>
<td>0.0787 (0.106)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT + Fire*HT</td>
<td>-0.6302 (4.757)</td>
<td>0.1741 (0.177)</td>
<td>0.0010 (0.018)</td>
<td></td>
<td></td>
<td></td>
<td>0.0004 (0.001)</td>
</tr>
<tr>
<td>Fire + BD + HT</td>
<td>2.6528 (3.083)</td>
<td>-2.8418 (6.900)</td>
<td>0.1122 (0.135)</td>
<td>-0.0130 (0.013)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire +BD + Fire*BD</td>
<td>-0.6517 (4.680)</td>
<td>10.1328 (9.530)</td>
<td>0.001 (0.018)</td>
<td></td>
<td></td>
<td></td>
<td>-0.0251 (0.030)</td>
</tr>
<tr>
<td>Fire + BD + HT</td>
<td>2.6528 (3.083)</td>
<td>-2.8418 (6.900)</td>
<td>0.1122 (0.135)</td>
<td>-0.0130 (0.013)</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 12 continued.

<table>
<thead>
<tr>
<th>Species models</th>
<th>Intercept</th>
<th>Basal Diameter (BD)</th>
<th>Height (HT)</th>
<th>Fire Temperature (Fire)</th>
<th>BD*HT interaction (BDHT)</th>
<th>BD*Fire interaction</th>
<th>HT*Fire interaction</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Sugar maple</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>HT</td>
<td>-2.5911 (0.917)</td>
<td>0.0302 (0.009)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT</td>
<td>-2.0560 (2.118)</td>
<td>0.0306 (0.010)</td>
<td>-0.0026 (0.009)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BD + HT</td>
<td>-2.6446 (0.973)</td>
<td>0.2164 (1.042)</td>
<td>0.0278 (0.014)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BD + HT + BDHT</td>
<td>-2.8286 (1.057)</td>
<td>0.4834 (1.167)</td>
<td>0.0285 (0.014)</td>
<td>-0.0012 (0.001)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD + HT</td>
<td>-2.0422 (2.129)</td>
<td>0.2497 (1.026)</td>
<td>0.0278 (0.014)</td>
<td>-0.0029 (0.009)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT + Fire*HT</td>
<td>-1.9347 (3.061)</td>
<td>0.0290 (0.031)</td>
<td>-0.0031 (0.014)</td>
<td></td>
<td>0.00001 (0.0001)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>BD</td>
<td>-2.7283 (1.069)</td>
<td>2.4864 (0.897)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD</td>
<td>-2.3319 (1.891)</td>
<td>2.4994 (0.904)</td>
<td>-0.0019 (0.008)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD + Fire*BD</td>
<td>-2.3744 (3.260)</td>
<td>2.5431 (2.870)</td>
<td>-0.0017 (0.014)</td>
<td>-0.0002 (0.012)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD + HT + BDHT</td>
<td>-2.0968 (3.379)</td>
<td>-0.7243 (5.563)</td>
<td>0.0414 (0.059)</td>
<td>-0.0030 (0.014)</td>
<td>-0.0009 (0.002)</td>
<td>0.0045 (0.020)</td>
<td>-0.00005 (0.0002)</td>
</tr>
<tr>
<td>+ BD<em>Fire + HT</em>Fire</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Eastern redcedar</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT</td>
<td>-0.6735 (1.208)</td>
<td>0.0191 (0.005)</td>
<td>-0.0126 (0.007)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD + HT</td>
<td>-0.9429 (1.244)</td>
<td>-0.5998 (0.360)</td>
<td>0.0333 (0.011)</td>
<td>-0.0134 (0.008)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BD + HT + BD*HT</td>
<td>-1.5553 (0.960)</td>
<td>-3.0710 (1.640)</td>
<td>0.0365 (0.013)</td>
<td>0.0106 (0.007)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + HT + Fire*HT</td>
<td>1.6365 (2.457)</td>
<td>-0.0033 (0.020)</td>
<td>-0.0319 (0.020)</td>
<td></td>
<td>0.0002 (0.0002)</td>
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<td></td>
</tr>
<tr>
<td>Fire + BD + HT + BDHT</td>
<td>1.8154 (2.511)</td>
<td>-4.3236 (3.258)</td>
<td>0.0420 (0.051)</td>
<td>-0.0249 (0.020)</td>
<td>0.0098 (0.008)</td>
<td>0.0133 (0.021)</td>
<td>-0.0001 (0.0004)</td>
</tr>
<tr>
<td>+ BD<em>Fire + HT</em>Fire</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>BD + HT</td>
<td>-3.0125 (0.691)</td>
<td>-0.5815 (0.370)</td>
<td>0.0346 (0.011)</td>
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<tr>
<td>HT</td>
<td>-2.6908 (0.639)</td>
<td>0.0212 (0.005)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD + Fire*BD</td>
<td>3.2686 (2.176)</td>
<td>-0.9811 (0.996)</td>
<td>-0.0390 (0.019)</td>
<td></td>
<td>0.0145 (0.009)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire + BD</td>
<td>0.4377 (1.045)</td>
<td>0.6396 (0.205)</td>
<td>-0.0141 (0.007)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>BD</td>
<td>-1.7193 (0.473)</td>
<td>0.7308 (0.212)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire</td>
<td>2.3539 (0.707)</td>
<td>-0.0148 (0.005)</td>
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</table>

Models are in the form of $P = [1 + \exp\{-[\beta_0 + \beta_1 X_1 + \beta_2 X_2 + \ldots + \beta_n X_n]\}]$. $P$ is probability an individual is alive following fire, $\beta_i$ are the parameter estimates, and $X_i$ are the independent variables fire temperature (Fire), basal diameter (BD), and height (HT).
Figure 21. The probability that advance regeneration will be alive in the first growing season following a dormant season prescribed fire based on the species and initial basal diameter. Significance of the logistic models was determined from the test of the global null hypothesis that all $\beta$ are equal to zero using the likelihood ratio statistic. Survival probability plots were derived using equations in Table 12. Open circles represent the basal diameter and fate of tagged individuals.
Figure 22. The probability that advance regeneration will be alive in the first growing season following a dormant season prescribed fire based on the species and initial height. Significance of the logistic models was determined from the test of the global null hypothesis that all $\beta$ are equal to zero using the likelihood ratio statistic. Survival probability plots were derived using equations in Table 12. Open circles represent the height and fate of tagged individuals.
Discussion

In this study, we found the effect of a single dormant season fire on advance regeneration mortality and resprouting varied by species. As we expected, mortality was low for chinkapin oak, black oak, and bitternut hickory in the first growing season after a dormant season burn. In fact, almost identical numbers of individuals for these three species did not survive in the control treatment between 2015 and 2016. While mortality was low, these three species also showed high rates of sprouting after shoot dieback. This finding was expected, as the ability of these species to sprout following shoot dieback has been well-documented, especially for oaks (Dey and Hartman 2005, Blankenship and Arthur 2006, Johnson et al. 2009). Mineral soil is a poor conductor of heat and acts as an insulating layer from the heat of a fire (Iverson and Hutchinson 2002). The location of oak and hickory root collar dormant buds beneath the surface of the soil, a result of hypogeal germination, allows sprouting even when the entire aerial portion of an individual is killed by fire (Smith 1990, Johnson et al. 2009). Bitternut hickory is the most prolific of sprouters of all northern hickories with sprouts arising from stumps, root collar, and roots (Smith 1990). Additionally, oak species preferentially allocate carbohydrates into their root systems, a reserve that supports the growth of sprouts more readily than their other hardwood competitors (Huddle and Pallardy 1998, Brose and Van Lear 2004, Johnson et al. 2009). Dey and Hartman (2005) found similar sprouting rates after one dormant season fire for chinkapin oak and black oak seedling and sapling advance regeneration, with 76% and 70% experiencing shoot dieback with sprouting, respectively.
While sugar maple also responded to fire by resprouting, it did so at a much lower rate than oaks and hickory (20% versus 43-68%). Sugar maple, like most hardwood species that occur in the central US, has the ability to sprout after shoot injury (Dey and Hartman 2005, Blankenship and Arthur 2006). However, sugar maple has epigeal germination, resulting in the root collar buds occurring at or near the surface of the soil, increasing its vulnerability to fire damage and reducing the likelihood of sprouting (Godman et al. 1990, Brose and Van Lear 2004). Furthermore, we found that the proportion of sugar maple experiencing mortality was approximately equal to the proportion that sprouted following shoot dieback. In the control treatment, no sugar maple and eastern redcedar individuals experienced mortality, suggesting the mortality rate in the burn treatment was directly related to fire. As expected, eastern redcedar showed no sprouting following the prescribed burn, as this species does not reproduce naturally by sprouting or suckering (Lawson 1990).

We found that size (basal diameter and height) was an important predictor of survival probability for most species. Other studies have found that as stem size increases, the resistance to mortality from fire also increases (Guyette and Stambaugh 2004, Dey and Hartman 2005, Ward 2015). Larger stems have thicker bark, creating more thermal insulation for the cambium (Harmon 1984). Even small increases in the bark thickness can increase survival likelihood, as the time needed to kill the cambium of a tree is directly proportional with the square of bark thickness (Dickinson and Johnson 2001). Compared to sugar maple and eastern redcedar, oaks and hickories have thicker bark that better protects against cambial injury, especially as size increases (Hengst and Dawson 1994, Smith and Sutherland 1999). The maximum basal diameters and heights of
individuals that experienced mortality following fire were far greater for sugar maple and eastern redcedar than the other species. This suggests that, while oaks and hickories are susceptible to fire-induced mortality at the smallest sizes of advance regeneration, sugar maple and eastern redcedar must reach larger sizes to avoid being killed by fire. Additionally, the minimum size of individuals that experienced shoot dieback with sprouts was at least three times higher for sugar maple than for oaks and hickory, further suggesting the vulnerability of sugar maple to fire-induced mortality. Previous studies have shown that oaks resprouted when relatively small stems were topkilled, while competing species such as maple were completely killed by fire at smaller sizes (Brose and Van Lear 1998, Barnes and Van Lear 1998).

The logistic regression analysis for some of the species resulted in survival probability models that were either not significant or resulted in no maximum likelihood estimate. Several of the models for black oak had no maximum likelihood estimate, a result of quasi-complete separation of the data. For example, height as the independent variable for black oak was a perfect predictor of the survival probability because the only 2 individuals that suffered mortality were the smallest in the dataset. Chinkapin oak was the only species that had no significant models to predict probability of survival. Dey and Hartman (2005) found both height and basal diameter to be significantly related to survival probability for this species. The inability of data to fit a given model form does not belie the underlying ecological phenomenon. The high proportion of insignificant models for oaks in this study is a result of low mortality at any size class, which is indicative of the fire tolerance of oaks. Larger sample sizes and longer observational
studies may better elucidate the relationship between oak size and fire survival probability.

The fire temperatures we measured were similar to other studies in eastern hardwood stands, where maximum temperatures during low intensity surface fires commonly range between 75 and 300°C (Iverson et al. 2004, Blankenship and Arthur 2006, Ward 2015). We found no significant relationship between fire temperature and survival probability for all species except for eastern redcedar. Although significant, fire temperature as a predictor of survival probability for eastern redcedar had very low support amongst all other model subsets. The lack of a fire temperature effect on survival probability for most species was unexpected, as this relationship has been documented in other studies. Brose and Van Lear (1998) found fire intensity was related to red maple mortality, and Ward (2015) found the percent topkill increased with increasing thermocouple temperature for oak, maple, and birch (Betula spp.). Additionally, Dickinson and Johnson (2004) described an exponential dependence between mortality rates and temperature, but also found 60 °C to be the threshold for cambial necrosis within stems exposed to surface fires. Flame residence time, which reflects rate of spread and flame width, may also be an important factor in stem injury and mortality (Dickinson and Johnson 2001). Because our rate of spread and fire temperature measurements were collected at the burn unit-level and plot-level, respectively, the fire conditions that existed at individual stems could not be determined. Paint temperature tags are inexpensive and easy to deploy, making them well-suited for fire ecology research. Future survival studies could obtain a more local fire temperature at each seedling or sapling to better explain the relationships between survival probability and fire temperature for a variety of species.
We anticipate that annual mortality rates and survival probability will change over time, particularly if sites are repeatedly exposed to prescribed fire. Some research suggests different individual- and stand-level responses to prescribed fire depending on fire frequency and season of burn (Waldrop and Lloyd 1991, Arthur et al. 1998, Peterson and Reich 2001, Fan et al. 2012, Arthur et al. 2015, Vander Yacht 2017). For example, Dey and Hartman (2005) found that survival probability after one fire was high for all species examined in the Missouri Ozarks, but decreased for all species after three or four burns. This decrease was most prominent in mesophytic species, such as flowering dogwood (*Cornus florida*) and blackgum (*Nyssa sylvatica*), and least prominent in the white oak group and hickories. We expect that repeated fires will continue to topkill advance regeneration, and that sprouting and survival probability will continue to decrease with each burn as rootstocks become depleted and sprouts take time to recover to pre-burn sizes.

Prescribed fire is a commonly recommended tool for restoring invaded oak woodlands. Our results show that even fire-sensitive species can resist injury and death from fire, especially in the case of a single dormant season burn. All species analyzed in this study had at least a 99% probability of survival at basal diameters above 9 cm. The implication of these data reveal that heavily invaded, fire-suppressed woodlands may not experience major structural and compositional shifts following a single fire. Nowacki and Abrams (2008) conceptualized several alternative stable states of oak communities depending on the fire regime and site type. Beyond a theoretical threshold change in community state, restoring a community to its original state requires considerably greater energy input than that required for its maintenance. Our short-term results support these
theories, and we recommend the use of other treatments in combination with prescribed fire, such as targeted removal of large stems, to accelerate restoration of invaded oak woodlands.
Chapter IV: Management Implications and Conclusions

Spatially and temporally variable pre-settlement fire regimes resulted in a diverse assemblage of natural oak communities in the central United States. With the regular suppression of fire in the 20th century, many open oak communities no longer resemble historical conditions, as increased tree density and invasion of fire-sensitive, mesophytic species have become pervasive issues. The encroachment of sugar maple and other mesophytic species at the Marais des Cygnes and La Cygne State Wildlife Areas, KS, reveal that, even at the westernmost margins of the eastern forests, the removal of periodic fire has created a changed landscape. The results presented in this study are relevant to those interested in restoration of oak woodlands.

Our findings suggest that prescribed fire alone is an effective tool for reducing small advance regeneration of sugar maple. While reductions were also observed in the oak and hickory seedling classes in the burn only treatment, the magnitude of change for those species was less than for sugar maple, as oak and hickory were capable of more vigorous resprouting after topkill. Eastern redbud also responded to the prescribed fire with high rates of sprouting, and the increase of small seedlings was a result of larger seedlings being topkilled and resprouting after fire. These results suggest that even a single prescribed fire may facilitate the short-term competitive position of oak advance regeneration in the understory.

In contrast, one prescribed fire is an inadequate tool for removing large advance regeneration or overstory trees of even fire-sensitive species. We found that sugar maple stems had a 99% probability of survival at only 3 cm basal diameter, and that a burn only
treatment had no effect on overstory stand structure. Repeated burning will be necessary to increase mortality and reduce sprouting potential of large seedling and sapling size classes. Ideally, prescribed fire should be used in conjunction with targeted removal of large stems in order to accelerate the restoration process.

Restoring open woodland structure and composition from closed canopy conditions requires a greater input of resources than the regular maintenance of an existing woodland. The use of mechanical or chemical thinning can quickly and precisely reduce overstory and midstory densities, creating more light availability to understory vegetation. In this study, the use of a single chainsaw girdle and application of herbicide proved especially effective at reducing sapling density in the first growing season after treatment. In contrast, the thinning treatments were not successful at meeting the target residual overstory basal areas and associated woodland structures in 2016, with sugar maple being especially resistant to our single girdle and herbicide treatment. Although no plot data were collected in 2017, by the second growing season we observed considerably higher thinning-induced mortality, as well as greater canopy openness and large increases in herbaceous cover. We are optimistic that with time, this treatment will prove to be an effective method at reducing overstory basal area.

In the absence of prescribed fire, the aggressive thinning treatments prescribed in this study may release undesirable mesophytic species in the understory. Our initial results already reveal an increase in the cover of woody functional groups in the thin only treatment units, likely as a result of greater canopy openness and increased light availability. We recommend following thinning treatments with prescribed fire application to offset the response of smaller size classes to increased light availability.
Moving forward, prescribed burning treatments will need to be applied at short-intervals in order to deplete the rootstocks and reduce sprouting vigor of mesophytic oak competitors. Due to the short timeframe of data collection, the initial ineffectiveness of our overstory thinning, and the observed structural changes between 2016 and 2017, long-term monitoring of these plots will be important to inform oak woodland restoration practices.
Literature Cited


