THINNING AND BURNING IN OAK WOODLANDS

A Thesis presented to the Faculty of the Graduate School
University of Missouri - Columbia

In Partial Fulfillment
Of the Requirements for the Degree

Master of Science

By

CARTER KINKEAD

Dr. John M. Kabrick, Thesis Supervisor

July 2013

The undersigned, appointed by the Dean of the Graduate School, have examined the thesis entitled

THINNING AND BURNING IN OAK WOODLANDS

Presented by Carter Kinkead

a candidate for the degree of Master of Science and hereby certify that in their opinion is worthy of acceptance.

Dr. John M. Kabrick

Dr. Michael C. Stambaugh

Dr. David R. Larsen

Dr. Joshua J. Millspaugh

The following is dedicated to the members of my family, for their unwavering love and support. To Grandma Veta and Paw-Paw Carter, for showing me that adaptive management is applicable to more than just the wild meadow. To Grandma Linda, for her consistent support and positive influence. To my parents, for inspiring me to meet life's challenges and reminding me the true value of things. To my brother, and all friends alike, for teaching me loyalty and perseverance. Lastly, to Victoria-Lynn, for giving meaning to it all.

ACKNOWLEDGMENTS

First and foremost, I would like to thank my advisor Dr. John Kabrick of the USDA Forest Service Northern Research Station, for guidance extending well beyond the office. His patience, humble approach to teaching, and reliability were instrumental to my progress. In addition, I give thanks to the other members of my committee, Dr. Michael Stambaugh, Dr. David Larsen, and Dr. Joshua Millspaugh, who were especially dependable and encouraging. Special thanks to Keith Grabner, from the U.S. Geological Survey, for volunteering many hours of conceptual discussion, revision, and field time. I would also like to thank the field technicians including Joseph DeRuiter, Matthew Bourscheidt, Cory Lindeman, Eric Aiken, Trevor Swearingen, David Buttig, and many other crew members who helped collect impressive amounts of data over the years, especially Texas Nall of the U.S Forest Service. Mike Morris, Randy Jensen, Steve Burm, the Ellington Work Team, and other employees of the Missouri Department of Conservation also deserve much gratitude for their support. Ed Lowenstien and George Hartman were instrumental in study establishment and design. Previous work by Jeremy Kolaks and Erin McMurry was also incredibly helpful and appreciated. Finally, I would like to acknowledge my graduate student peers for the creating a positive environment and sense of community throughout our department.

This project was supported by the Joint Fire Science Program, the National Fire Plan, the USDA Forest Service Northern Research Station, the Missouri Department of Conservation, and the University of Missouri.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

USING PRESCRIBED FIRE AND MECHANICAL THINNING IN OAK WOODLANDS

Carter Kinkead

Dr. John M. Kabrick, Thesis Advisor

ABSTRACT

The legacy of fire suppression in savanna, woodland, and forest ecosystems since the early 20th century has allowed heavy encroachment of fire-intolerant species throughout open-oak communities of Midwestern North America. Shifts in resource gradients and the development of novel environmental-species relationships in many of today's second-growth stands has challenged conventional management techniques for objectives such as regenerating oak and pine, reducing fuel loads, and controlling invasive species. This study evaluates controlled burning and mechanical thinning treatments as methods for achieving these objectives while reversing the departure from natural or historic conditions in fire-suppressed woodland communities. We investigated each of these silvicultural practices, including the combination, for over a decade to address changes in stand structure, species composition, tree growth, and bole damage. We found that overstory stocking was significantly reduced after two prescribed burns and lowest in harvest-burn plots, which also contained the greatest percent coverage of forbs, graminoids, shrubs, and vines. Woodland indicator species responded almost equally to harvest-only and burn-only treatments, while the combination led to significantly greater coverage's than in control. Species diversity, richness, and evenness were also greatest on sites where two prescribed fires were conducted following harvests, although there was no significant difference between other treatments. Fire-induced mortality was significant after two prescribed burns, especially where harvested, but had minimal impact on trees >25 cm. Basal re-sprout in harvested stands was marginally reduced by two prescribed fires, but the density of small

diameter stems (<12 cm) was still greater than pre-treatment levels ten years after treatments were initiated. Results suggest that severe canopy disturbances such as mechanical thinning do improve structure and composition that is characteristic of open oak-pine woodlands; however, repeated prescribed fires are essential for sustaining open conditions. Overall, treatments show great potential for restoring stands from fully-stocked conditions and maintaining resemblance of historic woodland communities.

I. INTRODUCTION & LITERATURE REVIEW

Problem Identification

Accounts of the central and eastern U.S. prior to Euro-American settlement depict woodland communities as having much different structure and composition than is present on most of the landscape today (Houck 1908, Sauer 1920, Schoolcraft 1821, Beilmann and Brenner 1951, Cunningham 2006, Howell and Kucera 1956, Schroeder 1981, Nuzzo 1986, Ladd 1991, Jenkins et al. 1997, Batek et al. 1999, Nigh 1999). Survey records from the General Land Office (1815-1850) commonly describe stands of widely spaced, large diameter oak and pine, with sparse woody mid- and understory vegetation, and a rich herbaceous layer of forbs and grasses (Nuzzo 1986, Ladd 1991, Nelson 1997, Batek et al. 1999, Hanberry et al. 2012). Woodland communities were particularly abundant within the transitional ecotone that divides tall grass prairies of the Central Great Plains and the eastern deciduous forest biome, but also occurred in a variety of environments where topo-edaphic, climatic, and anthropogenic controls resulted in the formation of open hardwood stands (Schroeder 1981, Nuzzo 1986, Nigh 1992, Nigh and Schroeder 2002). Continual emergence of spatial and temporal fire history information through dendrochronology, palynology, archeology, and paleoecology research has linked recurrent fires to a continuum of structural and compositional gradients across the Midwestern prairie-forest border (Delcourt et al. 1986, Guyette and Cutter 1991, Abrams 1992, Westin 1992, Cutter and Guyette 1994, Guyette et al. 2002, Stambaugh and Guyette 2006).

However, large expanses of open oak woodland communities have been structurally and compositionally transformed due to the suppression of fire across a landscape fragmented by urban development and intensive agricultural land-use, including logging (Nigh 1999, Nuzzo 1986, Shifley et al. 2006). Since most temperate woodlands generally receive adequate

precipitation for tree growth (NOAA), the exclusion of fire has facilitated wide-spread "thicketization" or densification of forests and woodlands (Nigh 2004, Stambaugh and Guyette 2006, Nowacki and Abrams 2008, Dey and Fan 2009, Taft 2009, Engber et al. 2011). Fire-sensitive and shade-tolerant species have increased distributions and contributed to the well-known lack of oak regeneration throughout the central hardwoods region (Pallardy et al. 1988, Pallardy et al. 1991, Larsen et al. 1997, Barnes and Van Lear 1998, Hutchinson et al. 2005, Dey and Fan 2009, Johnson et al. 2009). Today, the open conditions of pre-settlement woodlands continue to wane, especially where there lacks a disturbance regime capable of reversing canopy closure and increased dominance of shade-tolerant vegetation.

The consequences of losing woodland habitat are difficult to synthesize, however, several studies have shown that species in both the plant and animal kingdoms utilize the inherent qualities of open oak ecosystems (Atwood and Steyermark 1937, Callahan 1996, Evans and Kirkman 1981, Thompson et al. 1996, Gaskins 2005, Mottl et al. 2006, Grundel and Pavlovic 2007, Mann and Forbes 2007, Masters et al. 2007, Templeton et al. 2011). Likewise, successional alterations in the absence of fire have been correlated with declining biodiversity in numerous regions (Ericksen 1997, Boyles and Aubrey 2006, Yates and Muzika 2006). Wildlife species including several bird [pine warbler (*Dendroica pinus*), Bewick's wren (*Thryomanes bewickii)*, red-headed woodpecker (*Melanerpes erythrocephalus*), Bachman's sparrow (*Aimophila aestivalis*), brown-headed nuthatch (*Sitta pussilla)*], mammal [plains spotted skunk (*Spilogale putorius interrupta*), elk (*Cervis Canadensis*), red bat (*Lasiurus borealis*), Indiana bat (*Myotis sodalis*)], and herpetile [ringed salamander (*Ambystoma annulatum*), northern fence lizard (*Sceloporus undulates*), wood frog (*Rana sylvatica*), western glass lizard (*Ophisaurus attenuatus*), eastern tiger salamander (*Ambystoma tigrinum*), western pygmy rattlesnake

(*Sistrurus miliarius streckeri*)] species inhabit woodland communities (Evans and Kirkman 1981, Callahan 1996, Thompson et al. 1996, Churchwell and Mierzwa 2004, Missouri's Comprehensive Wildlife Conservation Strategy 2005, Nelson 2005, Yates and Muzika 2006, Mann and Forbes 2007, Masters 2007, Mark Twain National Forest 2011).

Vegetative homogenization also currently threatens several Midwestern plant communities, especially where species such as bush honeysuckle (*Lonicera mackii*), wintercreeper (*Euonymus fortunei*), and sericea lespedeza (*Lespedeza cuneata*) have invaded imperiled woodlands (NRCS 2008, Nelson 2011). Current restoration of oak-pine woodlands in Missouri target the recovery of some 350 herbaceous plants, primarily conservative forbs and deep rooted warm-season grasses (Mark Twain National Forest 2011). This is particularly important as nearly 75% of Missouri's flora can be found in the Ozarks region where oak-pine forests and woodlands once covered roughly 2.5 million hectares (Steyermark 1959, Cunningham 2006, Mark Twain National Forest 2011). The Ozark Highlands Region reportedly contains 159 endemic plant and animal species, 77 modal species, 81 globally rare species, and 32 species in decline (Ozarks Ecoregional Assessment Team 2003 *in* Nelson 2011).

Since there are obvious linkages between fire suppression and shifts in woodland communities, it is generally assumed that the application of prescribed fire will do much to reverse structural and compositional trends (Boerner et al. 1988, Ko and Reich 1993, Pyne et al. 1996, Hartman and Heumann 2003, Albrecht and McCarthy 2006, Knapp et al. 2007, Vose 2000, Arthur et al. 1998, Engber et al. 2011, Kitzberger 2012). Yet some studies suggest that burning alone will not suffice in re-opening the canopy of closed woodlands, therefore failing to allow enough light to heliophytic understory vegetation (Dey and Hartman 2005, Hutchinson et al. 2005, Mottl et al. 2005, McMurry et al. 2007, Taft 2009). Fire can also cause undesired effects

on the woody component of open oak stands (Burns 1955, Paulsell 1957, Loomis 1973, Loomis 1974, Brose and Van lear 1999b, Smith and Sutherland 2001, Marschall 2012), and change the successive dynamics of communities with severe departure from historical or reference conditions. In this case, additional management such as thinning or the application of chemical herbicide may be needed to encourage removal of select shrub and tree species, and eliminate vegetative strata which intercept light from the understory.

Purpose

Although there is increasing interest in restoring open woodlands throughout the Midwest (Packard 1993, McCarty 1998, Bowles et al. 2007, Sparks et al. 1988, Sparks et al. 2012), there is not currently an adequate suite of literature to support the silvicultural techniques used in modern restoration efforts. In the Ozark Highlands, the limited ability to predict outcomes of large prescribed fires (200-2800 ha) requires managers to use adaptive management approaches when planning future burns (Dey and Hartman 2005). While some studies have addressed the effects of prescribed fire and commercial harvests, they often focus on objectives such as regeneration or fuel reduction (Kolaks et al. 2004, Fule´ et al. 2005, Engber et al. 2011). Therefore, it is necessary to refine our understanding of prescribed fire and commercial harvests, especially as it pertains to managing for woodland structure and composition. This study attempts to quantify the effects of such treatments and relate our findings to management objectives common to many state, federal, and private organizations.

Although quantitative restoration goals for ground flora compositions are not well established for many community types, an increasing number of studies involving thinning and burning are currently incorporating management objectives that extend beyond the regeneration

of tree species (Kruger and Reich 1997, Arthur et al. 1998, Sparks et al. 1998, Hartman and Heumann 2003, Hutchinson et al. 2005, Zenner et al. 2006, Bowles et al. 2007, McMurry et al. 2007, Phillips 2007, Mark Twain National Forest 2011). Specifically, this research identifies changes in the ground flora and overstory vegetation before and after prescribed fire, commercial thinning, and the combination, while accounting for interactions between aspect and other site variables. Measurable characteristics such as species abundance and diversity are recorded in the understory vegetation, and analyzed according to lifeform (e.g. forb, graminoid, etc.). In addition, other functional groups such as woodland indicator species and legumes are assessed based on their response to each treatment. Overstory data pertaining to stocking, mortality, damage, and growth are also evaluated with the intention of quantifying the temporal dynamics subsequent to these silvicultural practices.

Objectives and Hypotheses

1. Examine changes in stand structure, including canopy cover, as a function of fire, thinning, and the combination by aspect.

   Hypotheses:

   a. Tree density and percent stocking will decrease in all treatments except control.
   b. Small diameter stems are more susceptible to thinning and burning.
   c. Canopy openness and available light will increase where thinned and burned.
   d. South-facing aspects (exposed slopes) will have fewer trees and greater canopy openness than north-facing aspects (protected slopes).

2. Evaluate changes in composition of woody species as a function of fire, thinning, and the

combination by aspect.

Hypotheses:

a. Oaks and pines will exhibit greater tolerance (less mortality and scar frequency) to fire disturbance than other species present.
b. North-facing aspects will have more non-oak, mesic tree species than ridge and south-facing aspects.

3. Quantify and model stem damage, tree mortality, and radial growth of different size classes and species groups following the application of common silvicultural techniques in oak dominated ecosystems.

Hypotheses:

a. Fire scarring will be larger and more abundant where fire-intensity is expected to be greatest (steep, dry, high-density stands).
b. Fire scar frequency and type is positively related to tree mortality and growth.
c. Cumulative mortality is a function of tree size and species adaptation.
d. Growth is a function of diameter, canopy openness, competing vegetation, and treatment severity.

4. Record and discuss species composition and cover of woodland ground flora using physiognomic classifications (forbs, graminioids, vines, shrubs, etc.) and other functional groups (woodland indicators) as a function of treatment and aspect.

Hypotheses:

a. Forbs, graminoids, and legumes will increase in percent cover subsequent to

prescribed fire and commercial thinning.

b. Percent cover of shrubs, vines, and woody species will decrease after each controlled burn, but increase where harvests occur.

c. Understory diversity, richness, and evenness will improve in all treatments compared to control.

d. Species classified as 'woodland indicators' will respond positively to thinning and burning.

e. Woodland indicator species will be more abundant on ridge and south-facing aspects than on north-facing slope aspects.

Characterization Character and Classification

There are several relevant definitions characterizing woodland communities in the central and eastern regions of the United States. The Dictionary of Forestry (SAF 2008) defines a woodland as:

> "*A plant community in which, in contrast to a typical forest, the trees are often small, characteristically short-boled relative to their crown depth, and forming only an open canopy with the intervening area being occupied by lower vegetation, commonly grass*."

Other definitions include:

> "*…open stands of trees usually over 5 m tall with crowns not usually touching, generally forming 25-60% canopy cover and understories dominated by forbs, graminoids, and/or shrubs."* (The Nature Conservancy's Science Department *in* Leach and Ross (1995)

"*Ecologically, true open woodlands are a type of wooded community characterized as having a canopy cover of 30 to 80%, a poorly developed understory, and a diverse herbaceous layer of forbs, grasses, and sedges with 50-100% ground cover*." (NRCS 2008).

"*…highly variable natural communities with a canopy of trees ranging from 30 to 100% closure with a sparse understory (or midstory) and a dense ground flora rich in forbs, grasses, and sedges*." (Nelson 2005, *see also* Taft 1997 and Faber-Langednoen 1999)

Use of term 'woodland' has not been widespread until the recent past (Leach and Ross 1995), in part, because their still remains modern day questions and discrepancies about the classification and functionality of these ecosystems. Conceptually, the structural and compositional gradients on which we place defining thresholds (i.e. canopy openness, forb coverage) vary ecologically in space and time, and at relatively local scales. This has often caused the term 'woodland' to be used interchangeably when describing forest and savanna communities. Since woodland condition generally represent a mid-seral phase of succession, attributes may not be structurally or compositionally sustained for long periods on productive sites, and may co-exist only on the margin of a larger contiguous ecosystem. In contrast, woodland structure and composition may represent climax conditions such as in cases where fluvial dynamics maintain mesic bottomland woodlands or claypan subsoil horizons create 'flatwood' woodlands (Nelson 2004).

Improved Ecological Classification Systems (ECS) are continually stratifying woodland communities within this spatio-temporal continuum. As mapping and defining ecological landscapes based on specific plant associations and other geophysical properties (i.e. landform, soil, geology) becomes more spatially explicit and universal, advanced frameworks may be used

to identify natural communities (Meinert et al. 1997, Grabner et al. 1999, Nigh and Schroeder 2002). For instance, this study takes place in the Current River Hills subsection of Missouri Ozark Highlands Ecoregion, where 4 of 9 Landtype Associations (LTA's) include distinct woodland communities (Schroeder 1981, Grabner et al. 1999, Nigh 1999, Nigh and Schroeder 2002). Here, long-term erosion of the ancient, uplifted Ozark Plateau has carved steep river-breaks that dissect this plains landscape (Nelson 1997, Meinert et al. 1997, Guyette and Kabrick 2002, Nigh and Schroeder 2002). While many of the adjacent valley slopes promote growth of mesic forests, greater fire frequencies on higher, less dissected uplands support the development of open oak-pine woodlands (Howell and Kucera 1956, Jenkins 1997, Batek et al. 1999).

This local contrast in community semblance illustrates the lack of importance in resolving broader definitions for most practical management applications. For the purpose of this study, context of the term 'woodland' will reflect communities with a partially open canopy structure that ensures sufficient light is available to support a dense ground layer of forbs, shrubs, and graminoids (Packard 1993, Taft 1997, Faber- Langednoen 1999). This flora may include species such as wild geranium (*Geranium maculatum*), purpletop (*Tridens flavus*), Virginia wildrye (*Elymus virginicus*), poverty grass (*Danthonia spicata*), sand sedge (*Carex muehlenbergii*), New Jersey tea (*Ceanothus americanus*), daisy fleabane (*Erigeron strigosus*), woodland brome (*Bromus pubescens*), ironweed (*Vernonia baldwinii*), and little bluestem (*Schizachyrium scoparium*). Legumes and composite species are also abundant among the summer and fall blooming woodland herbs, including lespedeza (*Lespedeza procumbens*), stiff aster (*Symphyiotrichum linariifolius*), purple aster (*Symphyiotrichum patens*), and many-rayed aster (*Symphyiotrichum anomalum*). Tree species of mixed age (2-4 cohorts, dominants 50-400

years old) and diameter are present in low densities, and heterogeneously arranged within one or more vertical strata (Nelson 2004, Nigh 2004).

Management Considerations

The importance of sunlight for sustaining oaks, pines, and woodland understory species typically necessitates some form of anthropogenic disturbance in unmanaged woodlands (McCarty 1998, McMurry et al. 2007, Dey and Fan 2009, Taft 2009). Despite the lack of a comprehensive silvicultural system designed for tending and regenerating woodland communities, several traditional forest management techniques provide canopy removals which promote woodland habitat and function. For instance, mechanical thinning and prescribed fire have potential to selectively kill woody species that impede photosynthetically active radiation (PAR) and compete with woodland species for soil water, nutrients, and growing space (Parker and Muller 1982, Boerner et al. 1988, Ko and Reich 1993, Peterson and Reich 2001, Boerner and Brinkman 2003, Franklin et al. 2003, Scharenbroch et al. 2012). Although thinning and burning have been used individually to address multiple land management objectives, few studies have reported the combination of these treatments (McCarty 1998, Brose and Van Lear 1999b, Franklin et al. 2003, Albrecht and McCarthy 2006), especially since conventional stewardship discourages burning in stands with crop trees and/or mechanical thinning in “natural” communities. However, each of these treatments serves different ecological functions meaningful to woodland communities.

Effect of Fire in Woodlands

Mean fire return intervals (MFI) in the Current River Hills watershed were between 6-12 years prior to 1820 (Batek et al. 1999), and became more frequent (≈ 3.7 MFI) throughout the Ozarks until the end of the Euro-American settlement period (circa 1940, Guyette and Larsen 2000) when the cessation of periodic fires took effect nationwide (Cutter and Guyette1994, Guyette and Dey 1997, Guyette et al. 2002). These fires played a number of ecological roles that perpetuated woodland communities. One example of woodland perpetuation is illustrated by the flammability of oak leaves and fine continuous fuels (i.e. forbs and grasses), which ignite and combust more completely than moist litter with low-lignin content, as is commonly found under shade tolerant species such as red maple (*Acer rubrum*), tulip poplar (*Liriodendron tulipifera*), and eastern red-cedar (*Juniperus virginia*) (Byram 1959, Guyette 1999, Stephens and Finney 2002, Boerner and Brinkman 2003, Stambaugh et al. 2006, Nowacki and Abrams 2008, Engber et al. 2011). Most often, thick-barked oaks and pines are not killed by fires conducted in flashy fuel types (oak litter, graminoids) (Hare 1965, Pyne et al. 1986, Hengst and Dawson 1993, Brose and Van Lear 1999a), whereas thin-barked species that compete with oak in the regeneration strata are fairly intolerant to any fire (Nuzzo 1986, Harmon 1994, Pallardy et al. 1988). The basis of this study assumes that the reinstatement of a fire regime capable of continually top-killing small diameter trees is essential to sustain cover of herbaceous lifeforms and ensure that future overstory composition is dominated by fire-adapted oaks and pines.

Several life history attributes of oak and pine contribute to their ability to support and occur in open woodland conditions. Rot-resistance and compartmentalization are among the advantages that endorse oak-pine dominance following cambial injury (Shigo 1984, Smith and Sutherland 2006), which is shown to be lethal at internal temperatures of 60° C (60 second

exposure) in white oaks (Hengst and Dawson 1993 *in* Cutter and Guyette 1994) and air temperatures of 532° C (15-20 minute exposure) in shortleaf pines (Lowery 1968). Prescribed burns in the Midwestern USA have reported temperatures between 83-370° C (Dey and Hartman 2005), therefore it is common for trees with significant cambial mortality to result in overall tree death (Byram 1958, Hare 1965). In fact, a tree may live for several years after a fire-induced injury is inflicted, as pathogens (i.e. disease, fungus, insects) cause varying degrees of fatality and infect wound sites at different rates (Shigo 1984, NWCG 2001, Smith and Sutherland 2006). Yet, most trees compartmentalize and/or "close" scar faces to survive cambial injury. Scrowcroft (1966) found that on one block of a long-term, replicated study in Missouri, 65.8 % of trees survived annual prescribed burns from 1949-1962, and 53% had fire scars. Periodic (every fifth year) fires on the same block were more intense due to increased accumulation of fuels between burns, resulting in 53.3% survival, and scarring 69.1 % of the surviving trees. Scrowcroft (1966) also shows that average scarring percentages were 49% greater in the red oak group (*Quercus* section *Lobatae*) than the white oak group (*Quercus* section *Quercus*) in annual burn plots, and 5 percent greater in periodic burn plots. Similar results have been reported by Burns (1955) and Paulsell (1957) who rate species in descending order of susceptibility as: scarlet oak> white oak> black oak> hickory> post oak. Stevenson (2004) showed comparable results based on scar frequency and size: red oak> black oak = white oak > post oak = hickories > shortleaf pine.

Spring fires typically cause greater damage and mortality of overstory trees than do fall and winter burns, since longer days and direct sunlight accelerate drying of fuels (less humidity, radiation), while warmer air temperatures and hydrated vascular tissues increase cambial heating (Brose and Van Lear 1999a). However, Sparks et al. (2012) explains that the exact timing of the

burn is not as critical for woodland management as achieving fireline intensity parameters (i.e. 500 kW/m) which ensure top-kill of mid-and understory species. Arthur et al. (1998) illustrates how differences in fire intensity can lead to inconsistent levels of mortality, finding that 16% of stems > 10 cm in diameter were dead after two moderate-intensity burns while a single, more intense fire resulted in a 20% dead:live ratio. Also, mortality is generally greater in small diameter stems, but may differ based on the pre-treatment density and composition.

Hutchinson et al. (2012) found that periodic burning over a 13-year interval reduced saplings (3-9.9 cm DBH) by 76%, and midstory trees (10-25 cm DBH) 34%. Dey and Fan (2009) found more modest reductions in the Ozark Highlands, where sapling (4-11 cm DBH) densities were reduced 50% compared to a 4% decline in stems >11 cm DBH after a decade of periodic fire. These and several other studies (Blake and Schuette 2000, Franklin et al. 2003, Hutchinson et al. 2005) show very little effect on overstory (>25 cm DBH) trees. However, Peterson and Reich (2001) found significant overstory reductions (6-8% per year) in oak woodland/savanna communities in Wisconsin where stands burned 4 or more times in 11 years. This study also reported that 32 years of burning (≈3 per decade) virtually eliminated a shade-intolerant sapling layer.

With recurrent fire eliminating most mid- and understory trees, surviving overstory trees remain at lower density than in most forests, and are generally dispersed with less proximity to one another (Nigh 2004, Nelson 2011, Hanberry et al. 2012). As managers target this characteristic open structure, they often describe woodlands by the degree of canopy closure, or percent openness. Bowles et al. (2007) found that 17 years of annual burning increased canopy openness 7% from pre-treatment conditions (12 % total). Increased radiation in open woodlands may further perpetuate spatial arrangements of oak and pine, especially during hot, dry summers

when water is scarce, and mesic or shade-tolerant species often desiccate (Ko and Reich 1993, Iverson and Hutchinson 2002). Belowground cotyledons and allocation of carbon to roots (greater root: shoot ratios) give oaks a competitive advantage when relying on carbohydrate reserves and the ability to resprout after repeated top-kill due to fire, ice, drought, herbivory, flooding, etc. (Brose and Van Lear 1999b, Varner et al. 2005, Dey and Fan 2009, Johnson et al. 2009, Kabrick et al. 2002). Shortleaf pines also have the unique ability to re-sprout from the base, a trait that few coniferous species in the USA possess (Lowery 1968, Guyette and Cutter 1997, Stambaugh et al. 2002).

Basal sprouts may appear as soon as 10-20 days after a prescribed burn (Kruger and Reich 1997), and may not favor oak-hickory and oak-pine associations in cases where pre-burn compositions are dominated by shade-tolerant species which also produce sprouts (Dey and Fan 2009). Arthur et al. (1998) found that while one to two prescribed fires promoted sprouting of red maple and blackgum after eliminating 60-80% of 2-10 cm DBH stems, however, the density of sprouts in twice burned sites was less than after a single fire, as most of the young sprouts were killed during the second fire. Both Alexander et al. (2008) and Hutchinson et al. (2012) found minor increases in canopy openness (<6%) after two controlled burns, and concluded that understory oaks were not more competitive than mesic species such as red maple and blackgum in these shaded conditions. Other studies (Albrecht and McCarthy 2006, Franklin et al. 2003) were able to reduce red maple and oak-competitor species with fire, but could do not show positive increases in oak reproduction. However, Barnes and Van Lear (1998) and Brose and Van Lear (1998) indicate increases in oak establishment following prescribed fire in the piedmont regions of South Carolina and Virginia.

To the author's knowledge, radial growth of surviving overstory oaks after prescribed burning in oak or oak-pine woodlands of the midwest has not been reported. In Arizona, Fule´ et al. (2005) showed that prescribed burning treatments led to significantly lower Basal Area Increment (BAI) of trees in a ponderosa pine (*Pinus ponderosa*)-gambel oak (*Quercus gambelii*) forest, compared to burn+thin and control treatments. Contrarily, Boerner et al. (1988) found 55% growth increase in white oak one year post-burn in an oak-pine forest of the New Jersey pine barrens, attributing this growth to transient nutrient release as indicated by foliar increases in nitrogen, phosphorus, and calcium. Indeed, studies such as Franklin et al. (2003) and Scharenbroch et al. (2012) show increased nutrient availability following controlled burns. Yet, Reich et al. (2001) links fire frequency with decreased Nitrogen cycling and productivity. Scowcroft (1966) found that periodic fires increased nutrient (N, P, K, Mg, Ca) compared to control and annual burns, and shows that radial growth of oak saplings (5-20 cm DBH) was greatest in control plots, while lowest in annually burned plots for red oaks and periodically burned plots for white oaks. Overall, there is not conclusive evidence to explain interactions between radial growth of overstory trees and prescribed fire.

Maintaining partial canopies and open understories is essential to the growth of woodland ground flora (Mottl 2006, Taft 2009, many others). Fire is not only important for sustaining light penetration to the understory, but also consuming dead plant material and exposing growing mediums that favor the germination, establishment, and reproduction of woodland understory physiognomic groups (Mottl 2000, Boerner and Brinkman 2003). For instance, Franklin et al. (2003) finds that lifeforms requiring mineral soil to establish (i.e. grasses [poaceae], sedges [Cyperaceae], mosses [bryales]) responded greatest to burning and thinning treatments, which increased diversity significantly in an oak/maple herbaceous stratum. Kruger and Reich (1997)

also found that although forbs and shrubs were mostly unaffected by two prescribed burns, grasses and sedges became the most abundant understory lifeforms in mesic forest openings. Graminoid dependency on leaf litter removal may be important for developing prescribed burn intervals in woodland management, especially in forest and woodland sites with substantial leaf litter deposition. Stambaugh et al. (2006) evaluated annual forest litter accumulation rates (473 tonnes/ha) in Missouri, and found that 2 years are required to recover 50% of litter after a fire-event, and up to 12 years to achieve 99% re-accumulation. This suggests that periodic fire should fulfill seed germination requirements of most woodland herbs, while sustaining surface organic material important for nutrient cycling.

Sometimes managers conduct fires in early spring to take advantage of wet or frozen litter layers which protect soil surface horizons and surface vegetation during prescribed burns, although these lower-intensity fires are more appropriate in later stages of savanna and woodland development when woody mid- and understory sprouts are less abundant, and herbaceous species have established (McCarty 1998). For example, Ko and Reich (1993) showed that dry weights of herbaceous biomass were as high as 129 g/m^2 outside of oak savanna canopies, but as low as 22 g/m^2 beneath canopies producing less light and depositing allelopathic oak litter. Comparatively, 36 years of annual and periodic fire in the Ozark Highlands produced understory biomass (grasses+forbs) of only 4.5 g/ m^2 in Missouri (Godsey 1988). Moreover, the belowground rhizomes or other propagules of many native perennial herbs and grasses are generally protected during late dormant-season burns (Howe 1994, Iverson and Hutchinson 2002, Knapp et al. 2007). Albeit, Sparks et al. 1998 showed that less than 10% of forb and legume species responded differently to late-dormant season fires than to growing-season fires, which both resulted in overall increased in abundance and frequency, as well as community

richness and diversity. In addition, Bowles et al. (2007) found significant increases in summer herbs at no expense of spring herb abundance, increasing overall understory diversity after 17 years of annual prescribed fire.

Effects of Harvesting in Woodlands

Commercial harvests have also played a historic role in shaping oak and oak-pine woodlands, and are widely advocated for restoring and maintaining partial-canopied ecosystems. However, traditional forest cutting techniques commonly intended for tree regeneration and growth may need to be altered to sustain herbaceous lifeforms in the understory (Zenner et al. 2006). For instance, even-aged shelterwood harvests generally retain considerable overstory (5-7 m^2/ha) compared to seed tree or two-aged harvests (Kruger and Reich 1997). This residual overstory can be used to inhibit advanced reproduction and the development of a dense seedling layer (Dey and Jenson 2002) capable of rapid mid- and overstory recruitment after harvests (Kabrick et al. 2002, Dey and Hartman 2005, Brudvig and Asbjornsen 2007). Failing to restore adequate light to shaded or exhausted seed banks may contribute to further loss of woodland species (Mottl et al. 2006, Zenner et al. 2006), therefore, adjusting traditional shelterwood techniques (i.e. thinning from below) can ensure that understory objectives are met while selectively removing subcanopy and fire-intolerant species from the mid- and overstory (Mount and Horner 2005). Other techniques such as seed-tree harvests offer more drastic canopy reductions, but still reserve the ability to leave overstory trees (25-35 TPH) that 1) create ideal habitat for woodland biota, and 2) improve genetic viability for future tree crops. Clearcuts may also be appropriate when the overstory conditions are not favorable (species composition, disease, etc.) for retention and/or there is difficulty regenerating light-demanding trees, since

shoot growth from root-stocks usually favor oaks and pines (Mount and Horner 2005, Kabrick et al. 2008). Uneven-aged single tree harvests may be applicable where oaks and pines have recruited into overstory positions, if maintained at low densities (Lowenstein 1996, Larsen et al. 1999)

Silvicultural methods aforementioned provide managers the unique capability to target specific species and diameter classes, which may have different costs and returns associated. For example, a clearcut will likely result in the greatest volume of merchantable timber for the time and expenses invested, while shelterwood harvests generally need to take some number of valuable overstory trees when cutting to fund management efforts. In most cases, woodland thinnings may prompt logging companies to "sweeten the pot" with red oaks and white oaks to make a profit. Also, current emergence of biomass industries (i.e. power plants) and other markets (i.e. pulp, charcoal, pellets, firewood, posts) that use small diameter roundwood (5-20 cm DBH) may soon improve the feasibility of woodland restoration harvests (Hansen 2011, Mark Twain National Forest 2011). In addition, several cost-share programs and other financial incentives are offered by federal (Wildlife Habitat Incentive Program, Conservation Reserve Program), state (Missouri Landowner Assistance Program), and private organizations (National Wild Turkey Federation) to restore or "reconstruct" depauperate savanna and woodland communities (Schroeppel 2002). Equipment is also accessible for loan from many of these agencies who encourage landowners to personally create and sustain wildlife habitat, especially with the rising cost of environmental contract labor.

McCarty (1998) reports that nearly 15 years ago, hiring a crew to thin small diameter trees, treat stumps with herbicide (i.e. Garlon), and pile and burn costs up to $3,000 per hectare. Yet, warns that neglecting to chemically treat stumps may result in such dramatic growth of

woody sprouts that early fires (first 4 or 5 burn cycles) may be solely intended for sprout control during ‘phase 1’ of savanna and woodland restoration. This study did not evaluate use of chemical herbicide, but did see potential for application. However, the use of overstory canopy cover to regulate seedling and sapling growth thereby keeping them vulnerable to fire (Brose and Van Lear 1999b), should adequately restore woodland character over time. Eventually, a fire-exclusion period (≈10-30 years) should be implemented to allow regenerating oaks and pines to reach fire-tolerant diameters (avoid top-kill) (Dey and Fan 2009). This interval should also take into consideration that survival probabilities reach 0.5only 14 years after establishment for blackgum, and 23 years for red maple (Harmon 1984). Fire-tolerant size classes of all species may take much longer in poor quality sites, and should be extended if timber value is an objective (Loomis 1974).

II. INITIAL CHANGES TO OAK WOODLAND STAND STRUCTURE AND GROUND FLORA COMPOSITION CAUSED BY THINNING AND BURNING

Introduction

Woodland communities are characterized by open mid- and understories and dense ground flora comprised of forbs, grasses, sedges, and shrubs (Nuzzo 1986, Nelson 2005, Taft 2009). They once were common in the western Central Hardwood Region and prairie-forest transition zone where low-intensity fires occurred frequently (Guyette and others 2002, Taft 2009). In the absence of fire, many of the oak (*Quercus*) woodland ecosystems throughout much of the Midwest have succeeded to compositions and structures resembling those of mature oak forests (Nowacki and Abrams 2008, Johnson and others 2009). In some oak ecosystems, mesophytic vegetation is replacing fire-dependant species at a rapid rate (Ladd 1991, Nowacki and Abrams 2008). Shifts in species composition and structure within woodland communities could jeopardize the biotic diversity, wildlife habitat, and ecosystem processes that occur in each environment (Peterson and Reich 2001, Shifley et al. 2006). Therein, these changes are deemed undesirable by many managers of state and private lands, many of whom have restoration objectives for their forests.

There is increasing interest in restoring the structure and composition of oak woodlands (Ladd 1991, Peterson and Reich 2001). Prescribed fire is considered an important woodland restoration tool (Nowacki and Abrams 2008, Taft 2009), largely because fire was the ecosystem process that maintained oak woodlands in the past and it reduces stand density, particularly by removing fire-sensitive species in the understory, thereby increasing the sunlight reaching the ground (Hutchinson and others 2005, Johnson and others 2009). Much has been written about

how the use of prescribed fire modifies forest structure and favors oak regeneration, particularly in mesic ecosystems where regenerating oaks have remained an important problem (Arthur and others 1998, Dey and Hartman 2005, Hutchinson and others 2005). However, there are relatively fewer studies that measure the effects of both fire and commercial overstory harvests (Brose and others 1999, Albrecht and McCarthy 2006), on forest structure and ground flora composition on upland sites at a landscape-scale.

The objectives of this study were to examine the effects of prescribed fire, thinning, and their interactions with slope position and aspect on forest structure and ground flora species composition one year after treatments were completed. Stand structure was evaluated based on changes in the diameter distribution and overall stocking by diameter. Ground flora species composition was evaluated based on changes in the cover of forbs, legumes, graminoids, shrubs, vines, and woody seedlings in the understory.

Study Area

This project was conducted in southeastern Missouri at Logan Creek Conservation Area (CA) and Clearwater Creek CA managed by the Missouri Department of Conservation. When the study was initiated, the sites were fully stocked and comprised primarily of oak-hickory and oak-pine forest types (Table 1). No management or documented fire had been recorded for at least 40 years. Sites were within the Black River Oak/Pine Woodland/Forest Hills Landtype Association, characterized by steep hillslopes consisting mainly of cherty, low-base soils and occupied by second growth forests (Nigh and Schroeder 2002).

Table 1. –Density, basal area, and stocking (Gingrich 1967) for all treatments.[1]

	Trees Per Hectare			Basal Area (sq. m/hectare)			Stocking Percent		
	2001	2003	2006	2001	2003	2006	2001	2003	2006
Burn	914.8	647.9	527.1	24.1	22.8	22.0	94.1	86.4	81.9
Control	866.4	784.6	750.0	24.6	24.2	25.6	94.7	92.1	95.9
Harvest	836.0	201.9	199.7	25.8	12.0	12.1	93.8	42.9	43.3
Harvest-Burn	916.0	137.6	130.5	24.1	8.7	9.3	94.1	31.1	32.9

[1]Data for 2001 were collected pretreatment, data for 2003 were collected after timber harvests and the first prescribed fire, and data for 2006 were collected after the second prescribed fire.

Methods

Study Design and Treatments

This study is designed so that four treatments are applied across three slope position and aspect combinations: north-facing slopes (aspect 315 to 45 degrees), ridge tops (slopes < 8 percent), and south-facing slopes (aspect 135 to 225 degrees). Treatments included prescribed fire (burn), commercial thinning (harvest), their combination (harvest-burn), and control. Treatments were paired by slope and aspect to create 12, 2-hectare units (hereafter "treatment units") per block. Three complete blocks were initially established; two at Clearwater Creek CA and one at Logan Creek CA, each approximately 24 hectares in area. However, due to unsuitable weather conditions, some of the burn units in block three were not treated timely. Consequently block three was not included in our analyses.

Timber harvests occurred during the summer and early fall 2002, prior to the first burn. Harvesting reduced stand density to 40 percent stocking (Gingrich 1967) by thinning from below. However, to achieve stocking goals, some dominant and codominant trees were removed. Preferred trees for retention were white oak (*Quercus alba*) and shortleaf pine (*Pinus echinata*) because of the fire tolerance of these two species. Prescribed fires were applied during spring for burn and harvest-burn units in 2003 and 2005. Each burn was executed using the ring fire method, while burning the ridges at the same time. Fire behavior parameters are included in Table 2.

Fire Behavior Measurements

Fire behavior was characterized using passive fire behavior sampling techniques, passive flame height sensors and rate of spread clocks. Flame height data was collected using passive flame height sensors which were placed in the three overstory plots within each stand. Passive flame height was measured using 12 strands of cotton string treated with fire-retardant, which were suspended between two wires, one at fuel bed height and the other approximately 2 m above the fuel bed (Kolaks and others 2004). Additionally, trained observers used visual aids to determine flame-tilt angle as the fire front passed through the flame height sensors. Estimated average and maximum flame lengths was derived by averaging flame heights and recording the tallest flame height recorded by the series of sensors and then applying the flame tilt angle. In the same plots, five rate of spread (ROS) clocks were inserted with one at plot center and one at 15 m in each cardinal or sub-cardinal direction (Kolaks and others 2005). ROS and direction were calculated using at least 3 measurements from the buried ROS or the average of all the triangle combinations, if more than three clocks activated and worked properly (Kolaks and others 2004, Simard and others 1984).

Table 2. –Observed parameters for fire behavior averaged across blocks one and two.

Treatment		*Rate of spread*		*Fireline intensity*		*Flame height*	*Total Energy Release*
		Nelson[1]	Byram[2]	Nelson[1]	Byram[2]		
		cm/min		BTU/m/sec		cm	(TER)
1st fire	Burn	73	37	141	72	50	2419
	Harvest-Burn	110	58	249	135	61	3466
2nd fire	Burn	ND[3]	ND	92	46	38	ND
	Harvest-Burn	ND	ND	236	131	46	ND

[1] Based on an equation by Nelson (1986).
[2] Based on an equation by Byram (1959).
[3] ND= Not Determined

Vegetation Sampling

Permanent plots were established during the summer of 2001 along a transect following the contour of the slope that is approximately 215 m long. This transect contained both woody and herbaceous vegetation plots. All trees ≥ 4 cm in diameter at breast height (dbh) were inventoried in three, 0.82-hectare circular plots randomly located along transects within each treatment unit. All trees and shrubs > 1 m tall and < 4 cm dbh were inventoried in 15, 0.0008-hectare circular subplots randomly located along transects within each treatment unit. Trees ≤ 1 m tall, ground flora, and vines were sampled in 30, 1 m X 1 m quadrats that were randomly located along transects within each treatment unit. Within each quadrat all live herbaceous plants and tree seedlings were identified to species and cover was estimated to the nearest percent. Post-treatment understory data were collected in the same plots and quadrats in the summer of 2003 and again in the summer of 2005 (i.e., during the first growing season after harvest and/or prescribed fire was applied).

Data Analysis

We used the general linear models procedure (Proc GLM, SAS version 9.1, SAS Institute, Cary, NC) to examine the effects of treatment (burn, harvest, harvest-burn, or control) and slope position and aspect (north-facing slope, ridge, south-facing slope) and the interaction between treatment and slope aspect. The error term was the block*treatment*slope aspect interaction. Response variables included the change in percent stocking (post treatment – pretreatment) by diameter size class for woody vegetation and the change in percent cover of ground flora by forbs, legumes, graminoids, shrubs, vines, and woody seedlings in the understory. Post treatment included the sampling period following the second burn. We limited our analysis to blocks one and two because they were burned two times during the study period. The vegetation data for 2003 (after one burn) was examined but not included in the analysis. To test for significant effects ($\alpha = 0.05$) we compared means using Fisher's least significant difference test.

Results

Changes in Overstory Structure

Prior to treatment, the average stocking for all stands was greater than 94 percent and sawtimber (> 27 cm dbh) comprised the majority of stocking (Table 1). The two prescribed burns caused only minor reductions in the overall stocking (Table 3). There was a 9 percent stocking reduction in the burn treatment compared to control and only a 6 percent reduction in the harvest-burn treatment compared to the harvest-only treatment. Most of the reductions due to the prescribed fire were in small tree sizes (4 to 14 cm dbh) and greater differences occurred

between the control and the burn treatment than between the harvest and the harvest-burn treatments (Table 3). When sampled in 2006, density (trees per hectare) was reduced twice as much in burn treatments as in control, and five times as much in harvest and harvest-burn treatments than in control (Fig. 1). We found no significant differences among slope position and aspect combinations or with their interactions with treatment.

Table 3. --Percent stocking levels and changes in stocking (post-pretreatment)[1].

Size Class (dbh)	Control	Burn	Harvest	Harvest-Burn
	Post treatment percent stocking			
	Change in percent stocking (pre - post)			
Small Trees (4 – 14 cm)	-2.4 a	-9.3 b	-11.9 bc	-17.7 c
Small Poles (15" - 21")	0.57 a	-4.3 b	-8.0 c	-9.0 c
Large Poles (22" – 26")	-0.89 a	-2.3 a	-5.7 b	-5.7 b
Saw Timber (>27")	7.2 a	6.1 a	-30.2 b	-25.6 b
Total	4.5 a	-9.8 a	-55.9 b	-58.0 b

[1] Post treatment data were collected one year after the last prescribed fire in 2005. Within rows, values followed by a different letter indicate significant differences ($P < 0.05$).

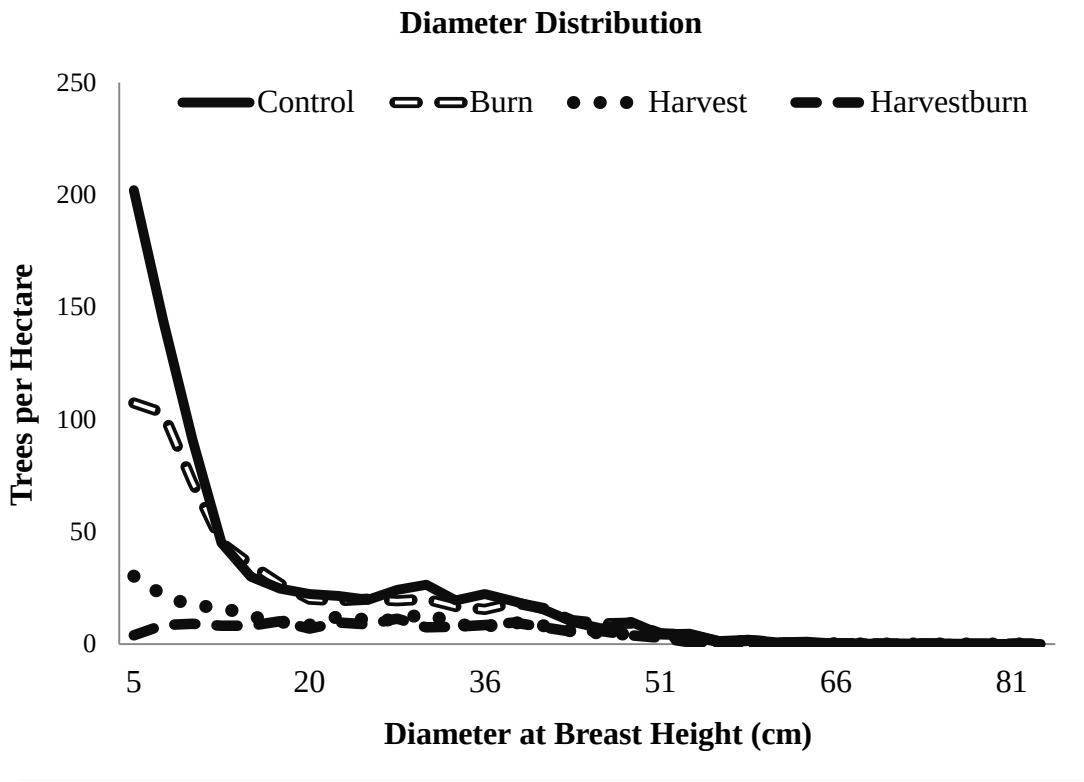


Figure 1. —Diameter distributions for trees one year after second prescribed burn.

Changes in Ground Flora and Understory

Forb cover increased in the burn and harvest-burn treatments (Fig. 2a) indicating that the fire was a primary influence. Harvesting alone had little effect on forb cover and the change in cover was not significantly different from that of the control. Graminoid cover including grasses, sedges, and rushes increased in all treatments except for control (Fig. 2b). As with forbs, the harvest-burn treatment resulted in the greatest increase in percent cover of graminoids (3.6 percent), however, this physiognomic group showed the smallest range of coverage variation between treatments. For legumes, the burn treatment had the greatest nominal increase in percent coverage (3.7 percent), however this increase was not statistically significant (Fig. 2c). The harvest and harvest-burn treatments increased the cover of shrub and vines (Fig. 2d, e) but the

changes were less for these life forms than for the others when compared to their non-burned analogs (i.e., burn vs. control and harvest-burn vs. harvest). The greatest change in percent cover of any physiognomic life form within a treatment was that of woody species in control plots which decreased by 15.9 percent but remained unchanged in the other treatments (Fig. 2f). This decrease in the control was caused by the mortality of seedling cohorts established after heavy seed crops that did not persist under a fully-stocked canopy. As with changes to structure, slope position and aspect had no significant effect on the changes in the coverage of these life forms.

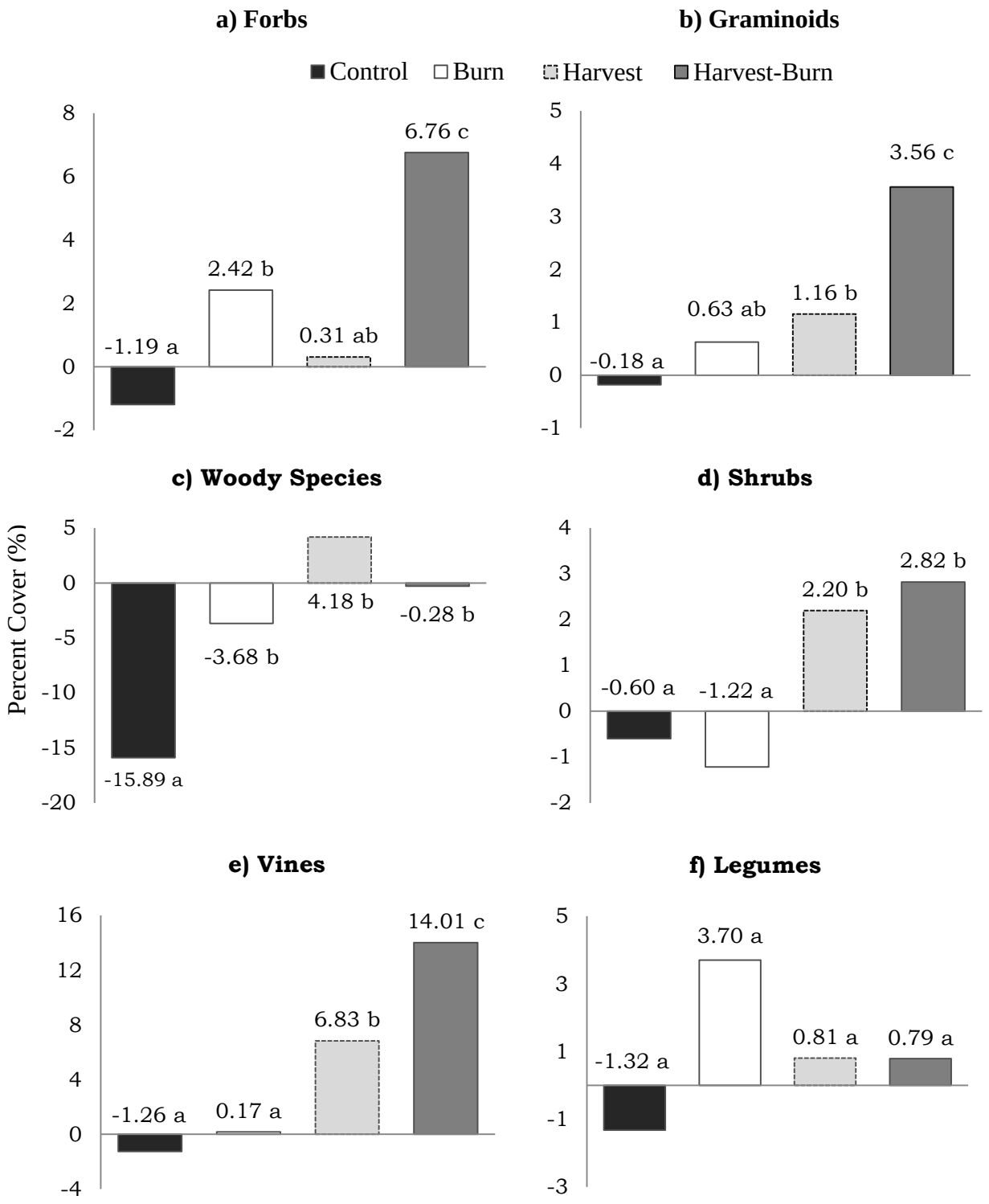


Figure 2. –Changes in percent coverage of six physiognomic life forms. Values followed by a different letter indicate significant differences ($P < 0.05$).

We grouped the ground flora into a "woodland indicators" category using a species list of legumes, forbs, and graminoids common to Ozark Highlands woodlands created by field staff of the Missouri Department of Conservation (Table 4). Percent cover of woodland indicators increased in the burn and harvest treatments compared to control with the greatest increases occurring in the harvest-burn treatment (Fig. 3).

Table 4. –List of woodland indicator species used in understory vegetation sampling.

Andropogon gerardii	*Ionactis linariifolius*	*Silphium terebinthinaceum*
Asclepias tuberosa	*Lespedeza hirta*	*Solidago hispida*
Aureolaria grandiflora	*Lespedeza procumbens*	*Solidago petiolaris*
Babtisia bracteata	*Lespedeza violacea*	*Solidago radula*
Blephilia ciliata	*Lespedeza virginica*	*Solidago rigida*
Ceanothus americanus	*Liatris aspera*	*Solidago speciosa*
Comandra umbellata	*Liatris squarrosa*	*Solidago ulmifolia*
Coreopsis palmata	*Lithospermum canescens*	*Sorghastrum nutans*
Cunila origanoides	*Monarda bradburiana*	*Symphyotrichum anomalum*
Dalea purpurea	*Orbexilum pedunculatum*	*Symphyotrichum oolentangiense*
Desmodium rotundifolium	*Parthenium integrifolium*	*Symphyotrichum patens*
Echinacea pallida	*Phlox pilosa*	*Symphyotrichum turbinellum*
Eryngium yuccifolium	*Pycnanthemum tenuifolium*	*Taenidia integerrima*
Euphorbia corollata	*Schizachyrium scoparium*	*Tephrosia virginiana*
Gentiana alba	*Silene regia*	*Verbesina helianthoides*
Gillenia stipulata	*Silene stellata*	*Violoa pedata*
Helianthus hirsutus	*Silphium integrifolium*	

[1] Woodland indicator species are herbaceous plants that produce flowers and seeds during the summer months, and are adapted to ecosystems where light penetration is relatively high. These species, often associated with prairie and savanna ecosystems, indicate stand density has remained sufficiently low to allow sunlight to reach the ground vegetation.

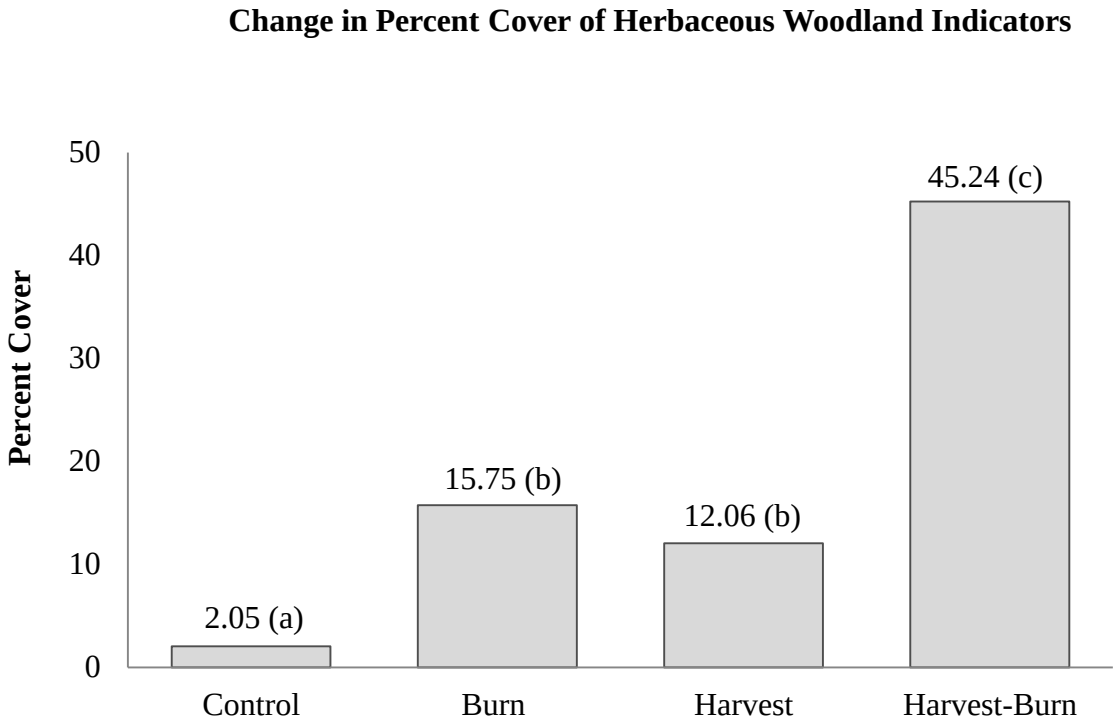


Figure 3. –Percent coverage of woodland indicator species (see Table 4) following treatment. Values followed by a different letter indicate significant differences ($P < 0.05$).

Discussion

The transition from what once were open-canopied oak woodland ecosystems to the present-day dense oak forest throughout the Ozarks is well documented (Ladd 1991, Nuzzo 1986). Fire suppression is thought to have facilitated increases in woody components of a stand, especially in small diameter stems (Albrecht and McCarthy 2006, Arthur et al. 1998, Hutchinson and others 2005, Nowacki and Abrams 2008, Peterson and Reich 2001). Our results showed that the two prescribed fires applied during a three-year period significantly reduced stocking of small trees compared to control conditions although overall stocking decreases attributable to the burning were minor (Table 3). In stands that were harvested prior to burning, stocking reductions attributable to prescribed fire were smaller, mainly because the thinning was applied from below,

targeting first the size classes of trees most vulnerable to mortality caused by fire (Dey and Hartman 2005). The larger size classes of trees in our study were mostly unaffected by the prescribed fire, a result also reported by others (Albrecht and McCarthy 2006, Arthur and others 1998, Hutchinson and others 2005, Johnson and others 2009). Larger diameter trees are much less vulnerable to mortality caused by fire, allowing most of the canopy dominant and codominant trees to persist in the overstory (Brose and others 1998, Dey and Hartman 2005, Taft 2009). This differential susceptibility to fire based upon tree diameter is thought to have led to the development of woodland structure characterized by the presence of "open-grown" large trees with a wide-spreading canopy and a relatively sparse understory and midstory (Nelson 2005, Taft 2009).

Despite only minor reductions in stocking, the two prescribed fires significantly increased the abundance of forbs, legumes, and graminoids (Fig. 2a, b, c), as well as woodland indicator species (Table 4, Fig. 3). The response of woodland indicators to the prescribed fire was about the same as in the harvest treatment where overstory stocking was reduced to about 40 percent. This suggests that the effects of prescribed fire were not limited to simply increasing amount of sunlight reaching the understory. It has been noted that fire reduces competition by woody species and removes some or all of the thick layers of leaf litter that can inhibit the germination of some of the woodland indicator ground flora (Nelson 2005, Stambaugh and others 2006). In our study sites, Kolaks and others (2004) reported that the first prescribed fire consumed more than 97% of the leaf litter, perhaps creating favorable conditions for herbaceous plant germination. Increasing sunlight to the ground layer certainly did play an important role in increasing the cover of woodland indicators. We found that the combination of opening the canopy and applying fire caused the greatest response in the woodland indicators; increasing

their cover more than three times compared to burn only, four times compared to harvest only, and more than 20 times compared to control.

Relative to control and burn treatments, the percent cover of woody species, shrubs, and vines increased where stands were harvested or harvested and burned. This is an important finding considering that a dense layer of woody plants may inhibit the development of a diverse herbaceous layer. In fact, the establishment of woody seedlings, seedling sprouts, and stump sprouts may become a problem for managing the herbaceous ground flora in stands that were thinned to low stocking levels and where the fire-free interval is several years long (Taft 2009). Our data suggested that the high shade from these remaining canopy trees in the burn treatment may have helped to slow or inhibit the cover of understory woody vegetation and promoted the development of herbaceous ground flora.

It is important to recognize that these results represent the changes that have taken place during a relatively short period of time following the application of treatments. Forest and woodland vegetation is dynamic and rapid changes in the cover of the understory are anticipated. Thus woodland management should be considered a continuous process requiring the application of prescribed fire and possibly thinning treatments on a regular basis to prevent the woody cover from becoming dominant in the mid and understory layers (Arthur and others 1998, Albrecht and McCarthy 2006, Brose and others 1998, Johnson and others 2009). Future work in this study will focus on the dynamic nature of restored woodlands and on the changes that take place during fire-free intervals.

It is also important to understand that at some point in time, management actions may be necessary to ensure trees can be recruited into the overstory to replace those that die or that are

harvested. These actions may require creating canopy openings by harvesting some or all of the overstory and by maintaining fire-free intervals to allow seedlings to grow large enough to survive when the prescribed fire regime is reinstituted. Although beyond the scope of this study, recruiting seedlings in managed woodlands is an important consideration if sustaining the composition and structure of the overstories of these unique ecosystems is a management objective.

Conclusion

The application of prescribed fire for restoring and managing woodlands was found to cause minor changes to forest structure primarily by reducing the stocking of trees < 12 cm dbh. Despite these minor changes, the prescribed fire significantly increased the cover of forbs, graminoids, and other plant species considered indicators of woodland composition. The effect of prescribed fire on woodland indicators was about the same as thinning stands to 40 percent stocking, underscoring the important effects of prescribed fire for maintaining woodland composition. However, harvesting alone also increased the cover of woody regeneration, shrubs, and vines – lifeforms that can prohibit the development of other desired woodland herbs. The greatest response of the ground flora, particularly woodland indicator plants, occurred with the combination of harvesting and prescribed burning. This suggests that increased sunlight to the ground layer, the removal of leaf litter, and the reduction of woody competition is the most beneficial to woodland plants. However, it is likely that the density of woody reproduction will rapidly increase under low overstory stocking if fire is not applied on a frequent basis.

III. THINNING AND BURNING TO RESTORE OAK WOODLANDS: EFFECTS ON RESIDUAL OVERSTORY TREES

Introduction

The re-introduction of fire into oak (*Quercus*) ecosystems is well documented (Harmon 1984, Boerner et al. 1988, Huddle and Pallardy 1996, Peterson and Reich 2001, Dey and Hartman 2005, Glascow and Matlack 2007, Johnson et al. 2009, Kinkead et al. 2011 *in press*, Hutchinson et al. 2012, many others). Managers are increasingly using prescribed burns to restore compositions of fire-adapted species and recover habitat structure, especially where fire exclusion has severely altered the functionality of oak savanna, woodland, and forest communities (Abrams 1992, Blake and Schuette 2000, Boerner and Brinkman 2003, Franklin et al. 2003, Johnson et al. 2009, Hutchinson et al. 2012). Evidence suggests that increased fuel loads in fire-suppressed hardwood stands can create hazardous burning conditions (Grabner 1999, Kolaks 2004, Stambaugh et al. 2011), as well as inhibit the establishment of oak, pine (*Pinus spp*), and many herbaceous woodland species (Jenkins et al. 1997, Stambaugh et al. 2006). In some cases of prolonged fire exclusion, increased dominance by mesic hardwood species (i.e. oak competitors) such as red maple (*Acer rubrum*), sugar maple (*A. saccharum*), tulip poplar (*Liriodendron tulipifera*), and blackgum (*Nyssa sylvatica*) may modify fuel bed characteristics and edaphic properties, thereby irreversibly reducing flammability of previously fire-adapted landscapes (Abrams 1992, Stephens and Finney 2002, Boerner and Brinkman 2003, Nowacki and Abrams 2008, Engber et al. 2011, Kitzberger et al. 2012). Consequently, fire disturbance is widely advocated to restore and sustain compositions of native, fire-tolerant

species in each strata of oak ecosystems (Abrams 1992, Packard 1993, Fule´ et al. 2005, Glascow and Matlack 2007, Johnson et al. 2009).

In particular, oak woodlands and savannas require frequent surface fires to control woody vegetation and maintain an open canopy structure (Nuzzo 1986, Packard 1993, Taft 1997). These communities are typically defined by the rich layer of forbs, grasses, shrubs, and trees that inhabit the understory of large, full-crowned oaks and pines (Taft 1997, Nelson 2005). While topo-edaphic factors greatly influence productivity and successive dynamics of open woodlands and savannas, the defining features of these systems are generally ascribed to fire (Bielmann and Brenner 1951, Harmon 1984, Nuzzo 1986, Packard 1993, Jenkins et al 1997, Peterson and Reich 2001, Franklin et al. 2003, Kinkead et al *in press*). Both prairie and forest taxa utilize the ecotonal properties (i.e. structural heterogeneity, quality forage) created by frequent surface fires in savanna and open woodland habitat, placing these ecosystems among the most diverse in North America (Evans and Kirkman 1981, Thompson et al. 1996, Taft 1997, Gaskins 2005, Grundel et al. 2007, Mann and Forbes 2007). The overstory component of open woodlands and savannas not only provides vital structure for songbirds and other wildlife (*Myotis spp., Rana spp. Accipiter spp, Otus spp*.), but also creates heterogeneity in microclimate and resource availability (i.e. water, light, nutrients) above and belowground (Boerner et al. 1988, Peterson and Reich 2001, Boerner and Brinkman 2003). Overstory oaks also provide highly concentrated food energy in the form of acorns, a primary ingredient in the trophic structure of woodlands and savannas (Martin et al. 1951, Burns et al. 1954, Christisen and Korschgen 1955, Block et al. 1991, Johnson et al. 2009).

Previous studies have reported contradictory results concerning the effects of prescribed fire on residual trees in oak woodlands. Some evidence suggest that by preferentially removing

small mid- and understory species with fire, overstory trees potentially benefit from the reallocation of sunlight and water (Parker and Muller 1982, Ko and Reich 1993, Iverson and Hutchinson 2002, Franklin et al. 2003). Transient nutrient releases following prescribed burns have also reportedly lead to increased productivity and tree growth in oak-dominated stands (Boerner et al. 1988, Franklin et al. 2003, Scharenbroch et al. 2012). However, several studies imply that environmental stress caused by burning may in fact reduce growth and vigor of residual trees, accelerate tree mortality, and devalue timber resources (Loomis 1974, Shigo 1984, Brose and Van Lear 1999a, Smith and Sutherland 2001, Marschall 2012). Accelerated mortality and reduced growth may eventually degrade the quality of the habitat in intensively-managed woodlands and savannas. Value loss of merchantable timber can be especially important in cases where harvests are necessary to offset costs associated with woodland management (e.g. removing unmerchantable biomass, prescribed burning). Although much is known about the deleterious effects of wildlfires on overstory trees, the link between tree growth and prescribed fire is not well understood, particularly where low-intensity ground fires are being used to reduce fuels or restore the composition, structure, and function of woodland or savanna communities. This study examines cumulative mortality, scar damage, and growth of trees following low intensity surface fires in an oak-pine woodland restoration site in the Ozark Highlands region.

Methods

Site Description

The study sites are located in southeastern Missouri, in the interior of the Ozark Highlands Region. Sites, primarily comprised of fully stocked oak-hickory and oak-pine forests,

are owned and managed by The Missouri Department of Conservation. Prior to our first treatment in 2002, there was no indication of past management or fire on the landscape for over 40 years. Each site lies within the Black River Oak/Pine Woodland/Forest Hills Landtype Association, characterized by steep, dissected hillslopes and occupied by second growth forests (Nigh and Schroeder 2002). Mean annual precipitation is typically about 100-120 cm, with temperatures averaging between 2.9 - 4.7° C in winter months, and 16.3 - 25.5° C in summer months (NCDC/NOAA 2012). Elevations on each site range from approximately 160-360 m above sea level. Site quality is relatively low, with the most productive sites on protected, north-facing aspects (*Quercus velutina,* site $index_{50}$ = 22-23 m) and the least productive sites on exposed, south-facing aspects (*Quercus velutina,* site $index_{50}$ = 20-21 m) (Kabrick et al. 2008). Soils in the higher slope positions formed in parent material derived from sandstones and cherty dolomites of the Roubidoux and Gasconade formations, and are generally highly weathered Ultisols, including the Series Poynor (loamy-skeletal over clayey, siliceous, semiactive, mesic Typic Paleudults), Scholton (loamy-skeletal, siliceous, active, mesic Typic Fragiudults), and Clarksville (loamy-skeletal, siliceous, semiactive, mesic Typic Paleudults) (Kabrick et al 2002). Soils on the lower slope positions are less weathered Alfisols which formed in clayey residuum of the Gasconade formation, and include the Series Alred (loamy-skeletal over clayey, siliceous, semiactive, mesic Typic Paleudalfs) and Rueter (loamy-skeletal, siliceous, active, mesic Typic Paleudalfs) (Kabrick et al. 2002). *For more detail regarding site characteristics, study design, and treatments see Kinkead et.al in press.*

Study Design and Treatments

A split-split factorial design was used in this study, replicated across three 25-hectare blocks. The experimental design includes two levels of prescribed burn treatments (burn or no

burn), each applied to one of two treatment units. Each treatment unit was split at two levels for commercial harvest treatments (harvest or no harvest), and split again to include three aspects: north-facing slopes (aspect 315 to 45 degrees), ridge tops (slopes < 8 percent), and south-facing slopes (aspect 135 to 225 degrees). In summer and fall of 2002, harvest treatments were implemented to create a woodland structure where large dominant and codominant trees remained scattered across each site. Thinning was conducted from below, removing first the small diameter stems until achieving the residual density goal of 40 percent stocking (*sensu* Gingrich 1967). Retained trees were primarily white oak (*Quercus alba*) and shortleaf pine (*Pinus echinata*), given their fire tolerance and estimated abundance through the presettlement era (Stambaugh et al. 2002).

On April 22-23, 2003, the first prescribed burns were performed using the ring-fire method and strip-fire method along ridge-tops. The second prescribed burn was conducted 2 years later for blocks 1 and 2 (March 12-15, 2005), and 3 years later (April 4, 2006) for block 3. Fire behavior including fireline intensity and flame height (each fire) and the rate-of-spread (first fire only) were estimated (see Kolaks et al. 2004 for protocols) and summarized in Table 5.

Vegetation Sampling

In the summer of 2001, permanent transects containing three randomly placed overstory plots (0.134 hectare) were installed on each aspect (north, south, and ridge) for all treatments in each block. For all trees > 12 cm DBH, species and condition (living, declining, or dead) were recorded. Vegetation was re-inventoried in the summer of 2003, 2006 following prescribed fire treatments, and again in 2012 following a fire-year fire-free interval. In summer 2012, canopy openness was estimated from photos taken with a hemispherical lens in each overstory plot

(Frazer et al. 1999). Fire scars that were present in 2012 were tallied according to type (Fig. 4) and measured dimensionally to calculate the surface area of each scar. No fire scars could be identified in control or harvest-only units, therefore it was assumed that all scars in the burn and harvest-burn treatments resulted from prescribed fire.

Table 5. –Observed parameters for fire behavior averaged across blocks in 2003 (adapted from Kolaks et al 2004), and 2005-2006 (adapted from Kinkead et al *in press*).

	Treatment	***Rate of spread***		***Fireline intensity***		***Flame height***
		Nelson[1]	**Byram[2]**	**Nelson[1]**	**Byram[2]**	
		cm/min		**BTU/m/sec**		**cm**
1st fire (2003)	**Burn**	73	37	141	72	50
	Harvest-burn	110	58	249	135	61
2nd fire (2005/2006)	**Burn**	ND[3]	ND	92	46	38
	Harvest-burn	ND	ND	236	131	46

[1] Based on an equation by Nelson (1986).

[2] Based on an equation by Byram (1959).

[3] ND = Not Determined

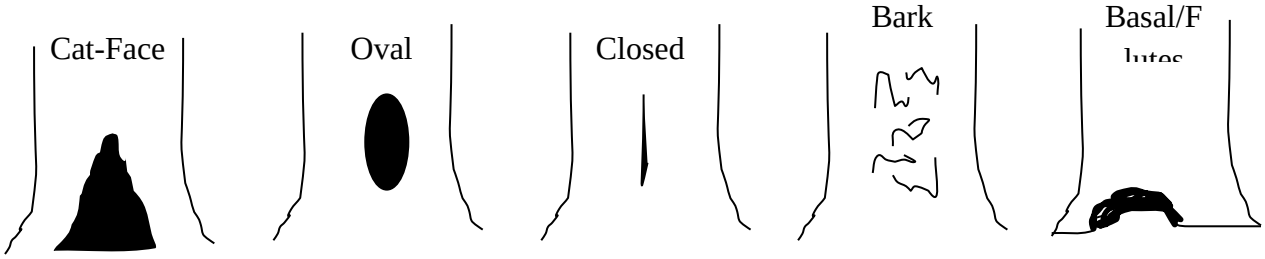


Figure 4. Scar type classifications used to quantify damage from prescribed burning.

Tree Growth

In summer 2011, increment cores were collected within each overstory plot from two canopy dominant or codominant white oaks that appeared most representative of each treatment unit. This species was selected because it was abundant in all treatment units, easily cross-dated, and well-understood with regard to increases in radial growth following commercial harvesting (Rubino and McCarthy 2004). For each cored tree, additional information related to tree growth was collected including diameter at breast height (DBH), and the percent slope, aspect, and basal area (10 BAF prism) immediately surrounding the tree. When possible, we attempted to collect data from one tree with a fire scar, and one without. Where there were fire scars present, size and type of each scar was recorded. All 216 cores were labeled, mounted, and finely sanded (1200 grit). Ring widths were recorded to the nearest 0.01 mm using a microscope and stage micrometer.

From each core, we estimated the ring width change (RWC) (Lorimer and Frelich 1989, Cook and Kairiukstis 1990, Nowacki and Abrams 1997, Rentch et al. 2002). This widely accepted technique referred to as the percent-increase method, compares the average ring width of a fixed time interval (or running mean) to the average ring width of a subsequent interval:

[1] RWC $= \frac{M_2 - M_1}{M_1}$ X 100

Where RWC is percent growth change in ring width, $M2$ is the mean annual radial growth of the period of interest, and $M1$ is the mean annual radial growth of the preceding period (Lorimer and Frelich 1989, Nowacki and Abrams 1997, Rentch et al. 2002). In this case, we used medians ($M1$, $M2$) instead of means for more precise estimates of central tendency to account for the non-normal distribution of ring width (Rubino and McCarthy 2004). Tree ring series were then classified as having either "moderate" (51-100 % RWC) or "major" (> 100 % RWC) growth responses according to Lorimer and Frelich (1989).

In addition, ring width data were also converted to basal area increment (BAI) (Visser 1995). BAI is often preferred in dendrochronology for quantifying wood production, offering accurate estimates of absolute growth when sampled trees vary in age or size in different stands (Rentch et al. 2002, Rubino and McCarthy 2000, Visser 1995). Because BAI accounts for increased tree diameter (circumference), it captures the additional volume added annually by each tree, whereas raw ring-width can mask the increased growth as a tree gets larger and older (Fulè et al. 2005, Johnson and Abrams 2001). As BAI increases with increasing tree diameter, values are usually detrended prior to making pre- and post-event growth comparisons (Cook and Kairiukstis 1990). We detrended our data by fitting a line that was derived from the linear regression [2] of average pre-treatment BAI for all trees:

[2] $Y = -16777 + 9.03131(\text{year}) + E$

Where Y is the predicted BAI for each calendar year and E is the error. Predicted BAI values were subtracted from actual BAI for every living year of each individual tree, similar to

standard detrending techniques applied to raw ring-widths (Cook and Kairiukstis 1990). Residual values for each year then supply a median for pre and post treatment intervals (*M*1 and *M*2). Fraver and Wilson (2005) showed that changes in growth may be overly-sensitive to high or low growth rates prior to a disturbance event when expressed as a percent, and suggest using differences instead. Therefore, rather than calculating percent change ($M_2 - M_1/M_1$), we simply subtracted pre-treatment median annual BAI from post-treatment (*M*2-*M*1).

To find the best fit for our data, we analyzed pretreatment intervals of 8, 10, 15, 25, 50, and 100 years (Fig. 5). There was minimal difference in median BAI's among these intervals, so for the purpose of this study we calculated median BAI values for 10 yr pre- and post-treatment intervals, and used the difference as the response variable. A review of current literature found that 10 year intervals were most commonly used for detection of growth increases due to intermediate canopy disturbance in the central hardwoods region (Lorimer and Frelich 1989, Nowacki and Abrams 1997, Rentch et al. 2002, Rubino and McCarthy 2004).

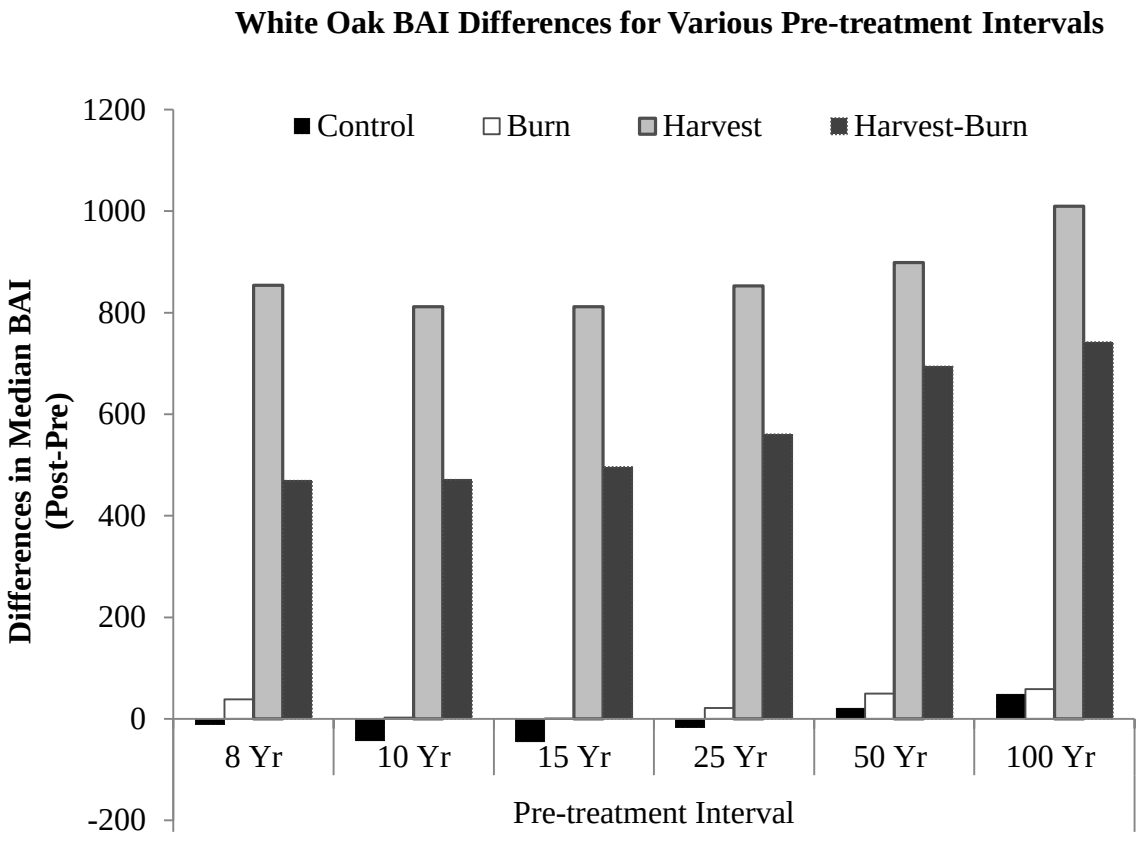


Figure 5. Differences in median BAI for various pre-treatment intervals averaged by treatment.

Analysis

For each plot combination in 2003 and 2006 (after each fire), the proportion of dead standing trees (cumulative mortality) was analyzed using a general linear mixed modeling procedure (Proc GLIMMIX, SAS version 9.3, SAS Institute, Cary, NC). Values were tested in a repeated mixed model where burn treatment, harvest treatment, aspect, and year were fixed effects and block was the random effect. Burn treatment (whole plot) was tested with the burn x block as the error and harvested treatment and aspect (split plots) were tested with the harvest x aspect x burn x block interaction. Year was tested as a repeated effect using the residual error

(year x harvest x aspect x burn x block). Significant ($\alpha = 0.05$) differences in cumulative mortality were determined by comparing Tukey's Least Squares means (LSmeans).

Logistic regression was used to compute the probability of scarring trees as a function of the treatment (burn vs. harvest-burn), and aspect. The binary response (scarred or unscarred) was analyzed using the general linear mixed modeling procedure (Proc GLIMMIX). For this analysis, we used the 2012 data set that includes scar observations from a single year. The model used to test the data was similar to the one described previously except that it did not include the year effect. We also examined the percent scarring by species group using this model. Lastly, scar area was tested as a function of treatment, DBH, and species group.

RWC and differences in median BAI (pre-post treatment) were also analyzed in a similar mixed model for all treatment combinations to assess changes in growth. Burn, harvest, and aspect were fixed effects and block was the random effect. Error for the burn treatment was burn x block interaction, and residual error (tree x harvest x aspect x burn x block) was used for harvest, aspect, and all interactions. These models included the covariates DBH, aspect, percent slope, canopy openness, tree age, surrounding basal area, presence of scars, and scar size (Table 6). All variables included in the growth analysis were also included in a Pearson's correlation matrix (Table 7) and assigned coefficients to detect multicollinearity (Proc CORR, SAS version 9.3, SAS Institute, Cary, NC)

Table 6. –Summary of variables used to model growth

Variable	Mean	Std Dev	Minimum	Maximum
BAI Difference (mm^2)	310.4	738.41	-939.9	4236
Tree Diameter (cm)	37.9	10.46	11.8	85.3
Tree Age (years)	97	32	39	258
Percent Canopy Openness (%)	16.2	7.26	5.1	41.1
Surrounding Basal Area (m^2/ha.)	14.6	8.27	0	39.03
Aspect[1]	1.0	0.69	0.02	1.99
Percent Slope (%)	22.0	10.53	0	54
Number of Scars on Tree	0.4	0.65	0	2
Scar Size (cm^2)	1605.1	1445	18	8352

[1] Aspect is transformed according to Beers (1966); cos (45°-aspect) + 1.

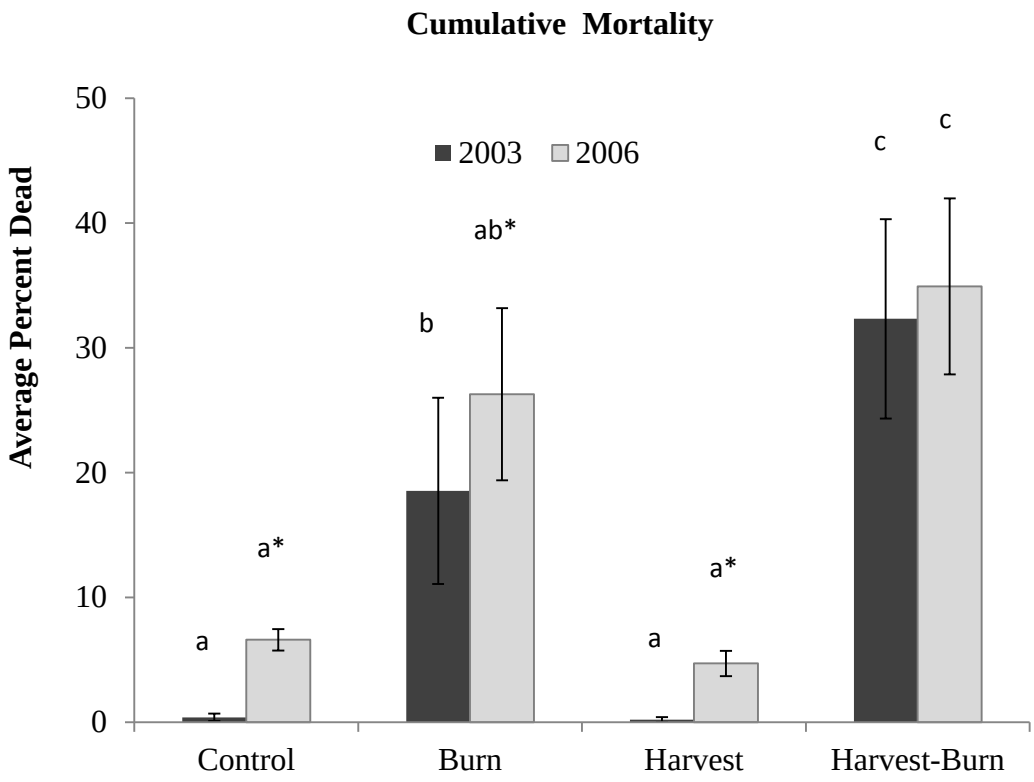
Table 7. – Pearson's correlation coefficients (prob > |r| under h0: rho=0, n = 216) among variables related to tree growth.

	BAI Difference	**Tree Diameter (cm)**	**Tree Age**	**Percent Slope**	**Percent Canopy Openness**	**Aspect**	**Surrounding Basal Area**	**Scar Area (cm^2)**	**Number of Scars**
RW % Change	0.09	-0.07	**-0.19**	**-0.14**	0.08	**0.17**	**-0.19**	0.08	-0.02
	0.19	0.33	**0.01**	**0.03**	0.22	**0.01**	**0.01**	0.22	0.83
BAI Difference		-0.09	**-0.19**	**-0.21**	**-0.14**	0.12	**-0.42**	0.05	-0.1
		0.21	**0.01**	**<.01**	**0.04**	0.08	**<.0001**	0.47	0.15
Tree Diameter (cm)			**0.54**	-0.08	**-0.21**	-0.08	-0.09	**0.15**	-0.03
			<.0001	0.24	**<.0001**	0.26	0.21	**0.03**	0.69
Tree Age				**0.21**	**-0.15**	-0.07	**0.17**	-0.04	-0.09
				<.01	**0.03**	0.32	**0.01**	0.53	0.19
Percent Slope					0.02	0.05	**0.16**	-0.09	-0.07
					0.8	0.48	**0.02**	0.17	0.34
Percent Canopy Openness						-0.13	**-0.17**	0.11	**0.29**
						0.06	**0.01**	0.12	**<.0001**
Aspect							-0.06	0.11	-0.04
							0.35	0.1	0.58
Surrounding Basal Area								**-0.2**	**-0.25**
								<.01	**<.01**
Scar Area (cm^2)									**0.52**
									<.0001

Results

Cumulative Mortality

The two prescribed fires did significantly increase cumulative tree mortality. LSmeans comparisons showed significant mortality differences between burned and unburned stands in 2003 (P=0.002), and 2006 (P=<0.001). Compared to control, the burn-only treatment significantly increased (P=0.042) the cumulative mortality by 18 to 20 times (Fig. 6). Compared to the harvest-only treatment, the harvest-burn treatment significantly increased (P=<0.001) the cumulative mortality by 30 to 35 times. Harvest-burn plots also had a significantly greater mortality than burn-only plots after each prescribed fire was conducted (P=<0.001 and P=0.031, respectively; Fig. 6). For all treatments except harvest-burn, cumulative mortality was significantly greater in 2006 than in 2003. While not significant (P=0.523), mortality on south aspects was nominally greater than on north aspects and ridges.



[1] Letters indicate significant differences between treatments in each year

[2] Asteriks indidicate significant differences between years in each treatment

Figure 6. Cumulative mortality of trees within 1 to 2 years of prescribed fires in woodland restoration sites of the Missouri Ozarks.

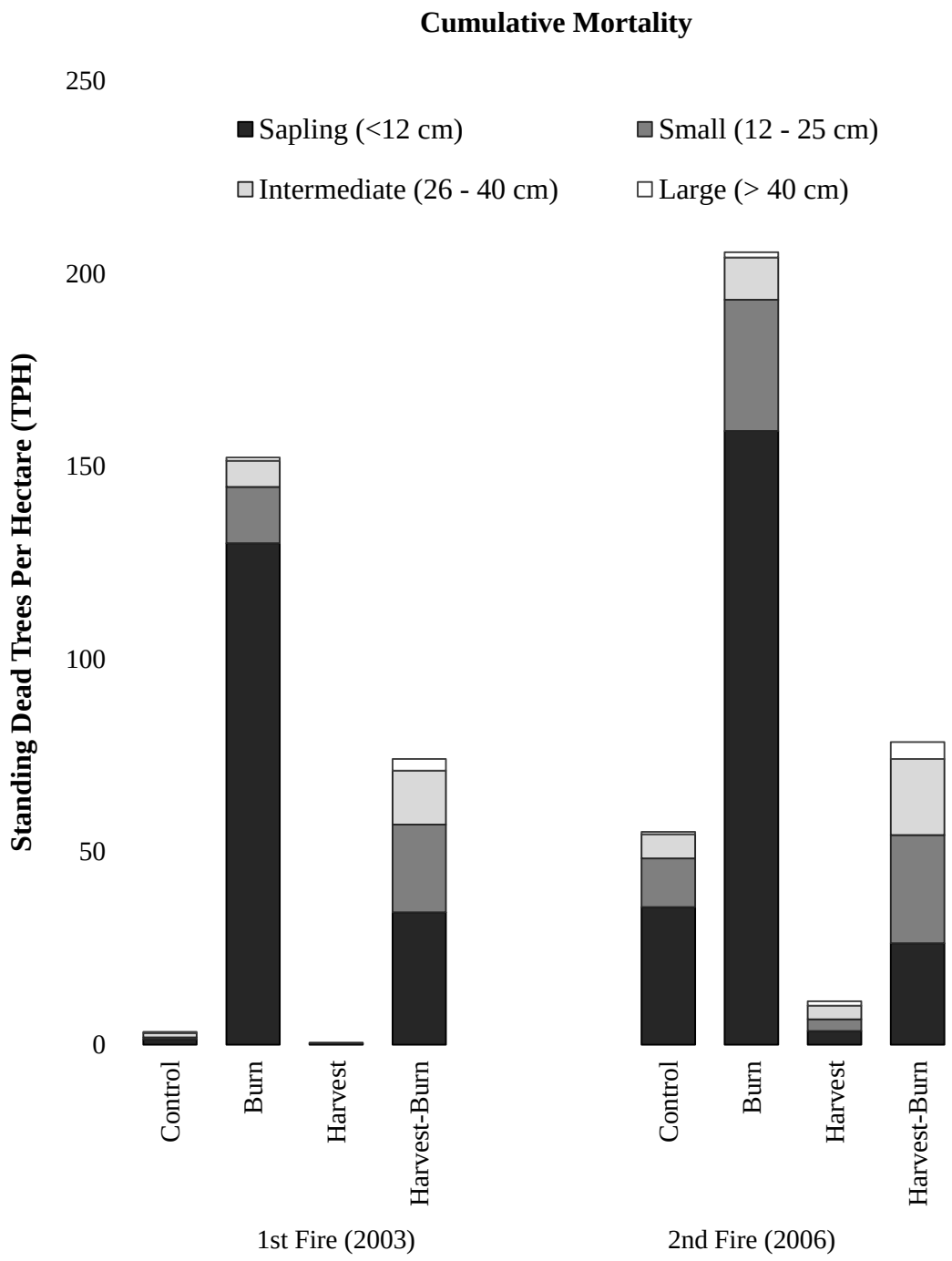


Figure 7. Density (TPH) of standing dead trees in each treatment after each fire.

The only size class with greater mortality in harvest sites than in burn sites was large trees (> 40 cm) in 2006, by a nominal 1%. For all other diameter classes, burn and harvest-burn

treatments resulted in the greatest mortality. The 2003 fire caused 26% mortality of sapling size (< 12 cm) trees in burn-only plots, and 54% in harvest-burn plots. After the 2006 fire, sapling mortality increased to 35% and 58% in burn and harvest-burn treatments, respectively. Cumulative mortality was inversely related to DBH in all burn and harvest-burn treatments, where saplings had an average of 16% greater mortality than small trees (12 - 25 cm). Intermediate (26 - 40 cm) and large trees were the least effected by all treatments. Burning alone caused less than 10% mortality in intermediate trees and less than 5% in large trees. However, harvesting before each fire led to 28% mortality of intermediate trees, and 15% of large trees.

On average, the red oak group (*Quercus* sect. *Lobatae)* was more vulnerable to prescribed burning than the white oak group (*Quercus* sect. *Quercus*), regardless of size (Table 8). Mortality in stems < 12 cm DBH was 12% greater in red oaks than white oaks after one fire, and 23% greater after two fires. Red oaks > 12 cm DBH had 6% more snags after the first burn than the white oak group, and 11% more snags after the second burn. Similar trends were observed in harvest-burn plots, where cumulative mortality of the white oak group increased approximately 9 % following the second prescribed fire. Scarlet oaks >12 cm DBH had more than double the dead:live ratio of black oaks, which were the most abundant oak species. Hickory (*Carya spp*.) and shortleaf pine < 12 cm DBH were the only species with reduced mortality in harvest-burn treatments after the second fire, as there were considerably fewer stems after 2003 (Table 8). Mature size classes of shortleaf pine were least affected by burn and harvest-burn treatments, with the exception of the white oak group in burn plots following the second fire.

Table 8. –Relative percent mortality for selected species after each burn, including the total number of trees (n) for each

	One Prescribed Burn 2003				**Two Prescribed Burns 2006**			
	Control	**Burn**	**Harvest**	**Harvest-Burn**	**Control**	**Burn**	**Harvest**	**Harvest-Burn**
Trees less than 12 cm in Diameter (DBH)								
White oak	0.4%	28.5%	0.8%	43.3%	9.9%	36.5%	2.6%	59.3%
(Quercus alba)	284	323	120	60	283	301	116	59
Post oak	–	25.0%	–	30.0%	8.3%	45.0%	12.5%	62.5%
(Quercus stellata)	45	20	4	10	36	20	8	8
Black oak	2.0%	45.0%	–	65.2%	22.9%	71.4%	–	66.7%
(Quercus velutina)	51	40	14	23	35	28	14	12
Scarlet oak	–	37.5%	–	66.7%	35.3%	65.0%	14.3%	100.0%
(Quercus coccinea)	12	8	3	3	17	20	7	9
Hickory species	–	20.9%	–	52.5%	1.6%	25.9%	1.9%	39.5%
(Carya spp.)	305	449	54	120	305	405	53	81
Shortleaf pine	–	16.2%	–	47.4%	1.4%	17.9%	–	37.5%
(Pinus echinata)	293	296	22	19	289	229	28	8
Blackgum	–	16.7%	–	22.2%	60.0%	50.0%	–	–
(Nyssa sylvatica)	5	6	3	9	5	6	2	4
Elm species	–	17.4%	–	–	4.2%	7.7%	–	100.0%
(Ulmus spp.)	104	46	0	1	96	39	0	1
Red maple	0.7%	47.1%	–	33.3%	1.5%	49.4%	–	100.0%
(Acer rubrum)	146	121	7	3	136	83	6	3
Sassafras	–	56.4%	–	100.0%	17.0%	86.7%	33.3%	100.0%
(Sassafras albidum)	88	78	10	2	94	98	9	3
Flowering Dogwood	0.4%	20.3%	–	63.6%	11.3%	29.6%	6.5%	80.0%
(Cornus florida)	495	538	68	11	512	521	77	10

Table 8. – Cont.

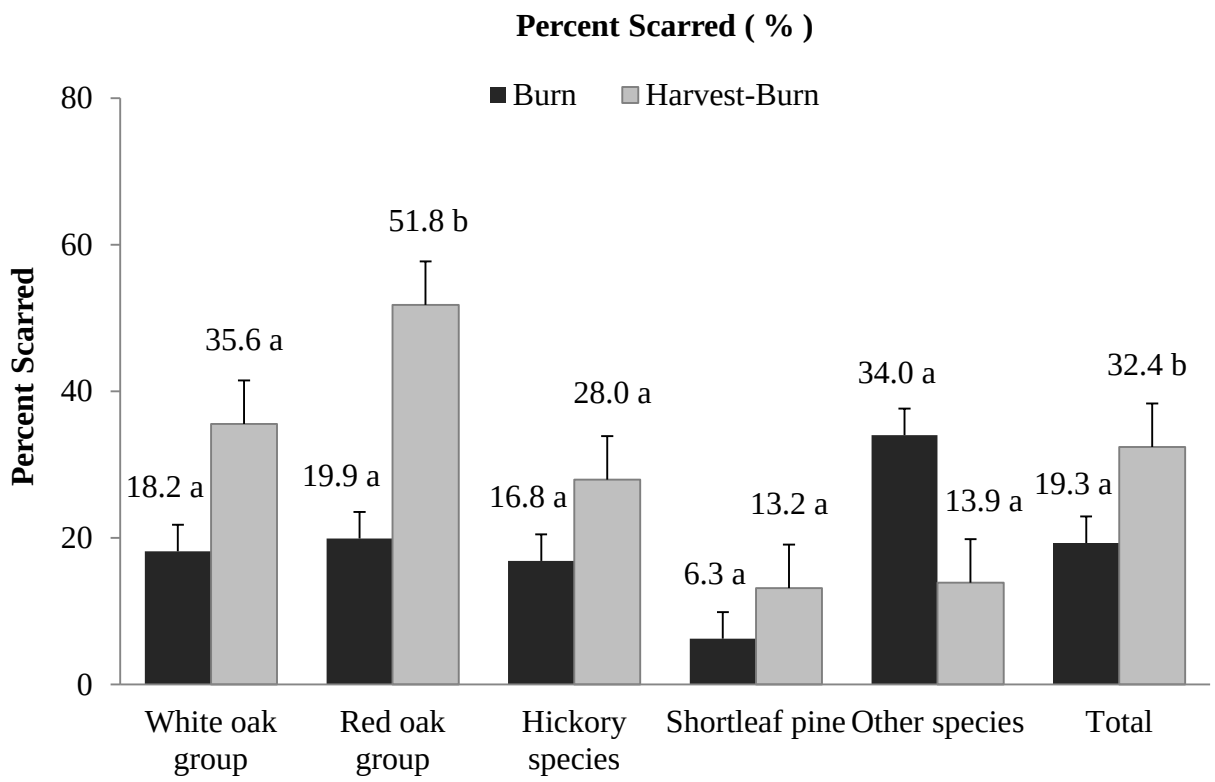
	One Prescribed Burn 2003				Two Prescribed Burns 2006			
	Control	**Burn**	**Harvest**	**Harvest-Burn**	**Control**	**Burn**	**Harvest**	**Harvest-Burn**
Trees greater than 12 cm in Diameter (DBH)								
White oak	0.3%	5.62%	–	20.8%	3.4%	9.0%	2.2%	28.4%
(Quercus alba)	353	338	184	125	357	356	183	141
Post oak	–	5.13%	–	18.6%	3.1%	9.1%	7.3%	27.9%
(Quercus stellata)	76	78	37	59	64	77	41	61
Black oak	0.6%	6.9%	0.7%	27.6%	9.2%	15.2%	9.9%	31.7%
(Quercus velutina)	341	319	139	214	349	309	131	202
Scarlet oak	1.0%	22.61%	–	49.0%	13.1%	32.6%	8.5%	64.0%
(Quercus coccinea)	97	115	58	49	130	144	82	75
Hickory species	–	4.4%	–	38.0%	1.5%	11.6%	1.0%	–
(Carya spp.)	136	250	94	79	131	258	103	5
Shortleaf pine	–	8.77%	–	25.0%	3.3%	7.7%	–	30.3%
(Pinus echinata)	25	57	2	4	30	65	2	76
Blackgum	1.8%	3.51%	–	2.1%	3.5%	11.9%	–	–
(Nyssa sylvatica)	110	57	38	48	113	59	33	5
Elm species	–	–	–	–	4.2%	30.0%	–	4.2%
(Ulmus spp.)	23	11	0	0	24	10	0	48
Red maple	–	12.5%	–	66.7%	–	11.1%	–	–
(Acer rubrum)	19	8	1	3	21	9	1	0
Sassafras	–	50.0%	–	–	20.0%	66.7%	–	–
(Sassafras albidum)	7	4	0	0	10	3	0	0
Flowering Dogwood	4.5%	–	–	–	20.8%	–	–	–
(Cornus florida)	22	12	5	0	24	11	5	0

Scar Damage

Since fire scars may take several years to develop after a cambial injury occurs (Smith and Sutherland 2001), we calculated the probability of fire producing one or more scars approximately six years after the second fire. Probability of scarring was significantly greater (P=<0.001) in the harvest-burn (0.29) treatment than in the burn treatment (0.17). The burn-only treatment resulted in 19.3 % of live overstory trees scarred, compared to 32.4% of trees in the harvest-burn treatment (Fig. 8). Differences in scarring by species group were observed, but did not significantly influence the probability of scarring (P=0.349). In general, species in the red oak group had the greatest percentage of scarred trees, followed by the white oak group, hickory species, and shortleaf pines (Fig. 8). The category "other" species, dominated by blackgum, elm, and dogwood, were the only group with greater scar frequencies in burn treatments than in harvest-burn (Fig. 8). Only the red oak group had significantly greater scar frequencies in the harvest-burn treatment (P=0.001). The scar frequency of shortleaf pine was less than half of any hardwood group in their respective treatments.

Aspect had a significant effect on the likelihood of scarring, where scar frequencies were significantly greater on south-facing slopes than on north-facing slopes (P=0.013). Of 334 total scarred trees in both treatments, only 11 had more than one discernible fire scar (three in the burn-only and eight in the harvest-burn). Without tracking individual trees, it is impossible to know which trees were alive at the time that each fire wound was inflicted. In addition to the data shown, 77 snags were inventoried but not included due to the severe decomposition preventing their identification and collection of scar dimensions.

Analysis of scar area revealed that harvesting before burning significantly increased the surface area (cm^2) of fire scars (P=0.017). On average, scar sizes in harvest-burn plots were twice as large as in burn only plots for live trees, and over six times as large trees killed by fire (Table 9). Tree diameter (P=0.329) and surrounding basal area (P=0.232) were not related to scar size. The only species with significantly different average scar sizes were white oak and dogwood (P=0.020). Aspect did not appear to influence the size of scar wounds on individual trees (P=0.639).



[1] Letters indicate significant differences between treatments for each species group

Figure 8. Percent scarring by species groups in a mixed oak-pine ecosystem.

"Cat-face" scars were the most common scar type recorded, comprising 40% of scars found on live trees in burn only treatments, and 37% in harvest-burn (Table 9). In addition, over half of the scars recorded on dead trees were cat-face. Closed and oval scars were the next most abundant scar types, each comprising approximately one quarter of the scars on live trees in burn plots. Oval scars also had the largest average scar size among live trees in burn plots, despite "bark slough" more than doubling the average size of any other scar type in any other category. For both treatments, the smallest scar type was closed.

Table 9. –Percentage and average wound size (cm^2) for all fire scar types observed in 2012.

	Burn		Harvest-Burn	
	%	cm^2	%	cm^2
Total	19.3 A	63.5 a	32.4 B	142.4 b
Type[1]				
Cat Face	40	85.4	37	165.4
Oval	26	99.2	17	174.6
Closed	24	11.5	30	23.2
Bark Slough	4	55	13	221.4
Basal / Flutes	6	66.2	3	127.4
Total Scarred	*174*		*142*	
Total Observed	*904*		*438*	

[1] Not analyzed by scar type

[2] Significant differences in scar percentages indicated by capital letters

[3] Significant differences in scar area indicated by lowercase letters

Growth Response

Percent-Increase method

In our study, only 5% of white oaks had a moderate growth response (> 50% RWC) to prescribed fire, and there were no major growth releases caused by burning (P=0.864). The control treatment did not have any trees that exhibited a moderate response, although it did have two trees (of 18) with growth increases that qualified as major release events. Harvesting caused significant (P=0.033) releases that were moderate in 22% of trees and major in 28% of trees. However, when fires were applied to harvested areas, only 13% of trees showed a moderate response, and only 9% major responses. Over half of the trees in harvest-only treatments had greater than 50% RWC, more than double the number of trees in harvest-burn plots and 9x more than in burn-only plots (Table 10).

Although control plots had the fewest number of trees with responses >50% RWC, this was not significantly different than burn only treatments (P=0.818). The majority of trees in control and burn treatments had between 0 and -24 % change in growth rate (Table 10). Where burned and harvested, 32 % of trees responded negatively to treatments, while only 15 % of trees showed reduced growth rates in harvest-only stands. The percentage of total trees with decreasing growth rates was 48.6 %. Treatment averages for RWC indicate that burn-only treatments led to a 1.5% reduction in growth rate, compare to 1.9 % decrease in untreated stands (Table 11). Harvest-only treatments resulted in an 84% RWC growth increase; an effect that was reduced to 35% by subsequent burns that were conducted following the thinning (harvest-burn).

Table 10. – Number of trees in each treatment according to growth response classification

		Control	Burn	Harvest	Harvest-burn	Total
RWC	< 0	35	35	8	17	95
	0 - 50	17	16	19	25	77
(% Change)	> 50	2	3	27	12	44
BAI	< 0	36	37	5	15	93
	0 - 50	16	13	15	22	66
(Difference)	> 50	2	4	34	17	57

Growth responses were compared among size classifications using both RWC and BAI differencing methods (Table 11). Both metrics of growth indicate negative responses for each size class in control plots, except RWC in intermediate size trees (1.6%). Prescribed burning caused a negative RWC in all size classes, and a negative BAI difference (post-pre treatment) in all trees below 40 cm DBH. Increased growth observed in harvest and harvest-burn plots was greatest in small and intermediate trees (Table 11). For all size classes, burns applied after harvest reduced radial growth (RWC, P=0.970) (BAI, P=0.013) compared to harvest-only treatments.

Table 11. – Average growth response of three size classifications within each treatment.

		Control	Burn	Harvest	Harvest-burn
RWC (% Change)	Small (12- 25 cm)	-8.0% a	-2.1% a	183.9% a	65.2% a
	Intermediate (26 - 40 cm)	1.6% a	-1.7% a	128.5% a	50.2% a
	Large (> 40 cm)	-6.3% a	-0.6% a	25.8% a	4.1% a
	All	-1.1 % a	-1.4 % a	82.1 % a	34.8 % a
BAI (Difference)	Small (12- 25 cm)	-85.0 a	-18.9 a	1196.0 b	526.9 a
	Intermediate (26 - 40 cm)	-10.8 a	-6.9 b	1012.4 c	639.0 d
	Large (> 40 cm)	-109.7 a	26.7 a	574.2 b	219.1 a
	All	-43.3 a	2.3 ab	811.6 c	471.0 b

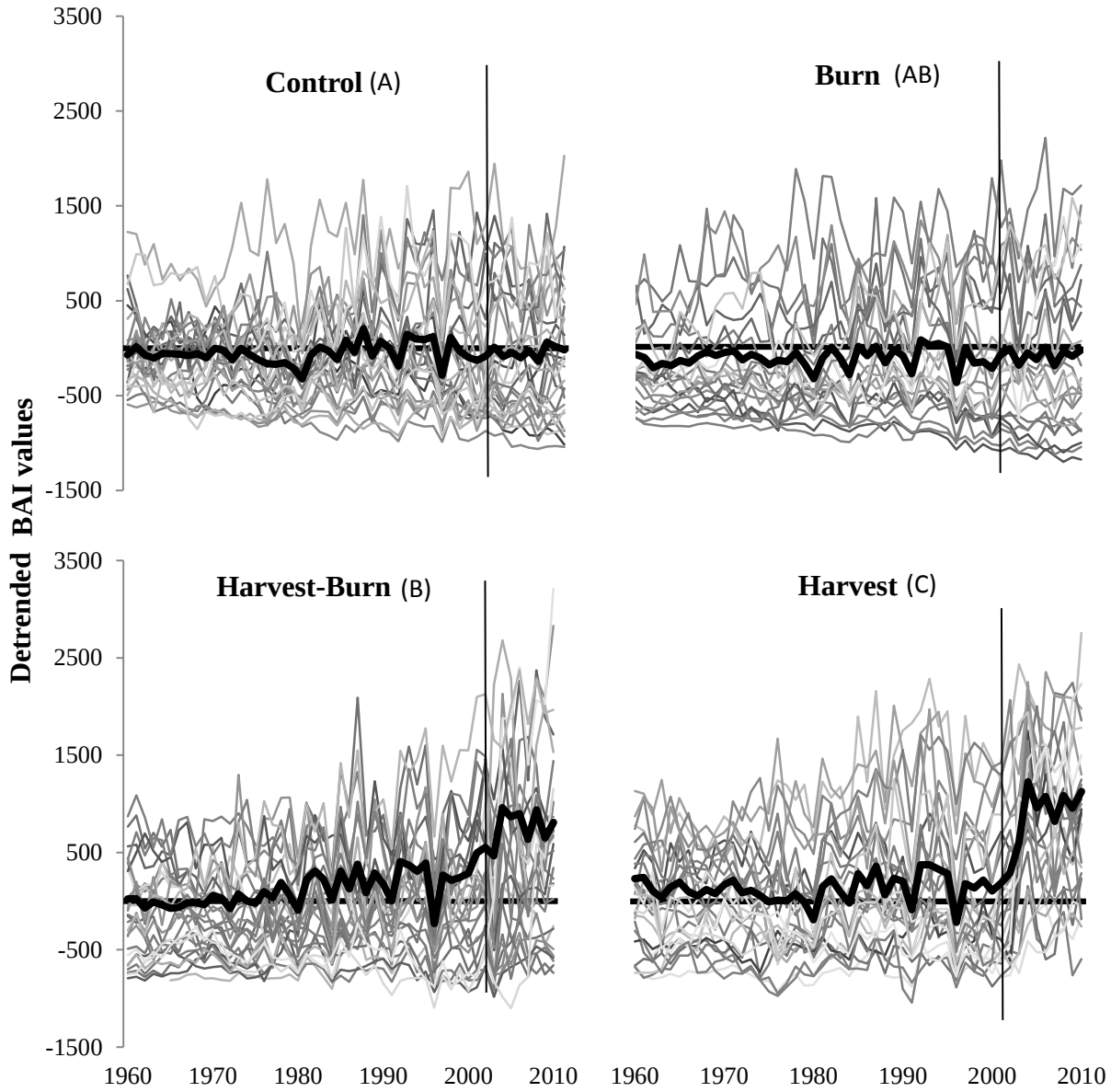
[1] Letters indicate significant differences between treatments for each size class

Statistical analysis of BAI

Overall, two prescribed burns did not significantly alter the growth of mature white oaks retained in this study (P=0.217). Commercial harvests (P=<0.001), and the interaction between burn and harvest treatments (P=0.006) were the only significant effects in the base model (Table 12), which received an AIC value of 311.1 (Akaike 1973). Aspect (i.e. north, ridge, south) nested within each treatment was also not significant (P=0.162). Tukey's LSmeans showed that

both harvest (P=<0.001) and harvest-burn (P=<0.008) treatments had significantly greater radial growth than in control. Growth in harvest-only treatments was also significantly greater than in harvest-burn (P=<0.007) and burn-only (P=<0.001) treatments (Fig. 9).

Several covariates including aspect, percent slope, canopy openness, surrounding basal area, and tree age were also analyzed, but did not explain any differences in radial growth in addition to treatment. Tree diameter (DBH) was the only significant predictor of growth response in the model (P=0.003) (Table 12), although it was not correlated to BAI differences in a Pearson's test (Table 7). The relationship between DBH and growth indicates that, on average, small-diameter trees had greater increases in BAI than large-diameter trees. Pearson's correlation analysis showed significant relationships between BAI differences and percent slope, tree age, canopy openness, and surrounding basal area (Table 7). Except for surrounding basal area, the same covariates were correlated with RWC, as well as aspect which was not correlated with any other variables in this analysis. Overall, the two metrics for growth (BAI differences and percent change in RWC) were not significantly related (P=0.19) .



[1] Letters indicate significant differences in mean BAI between treatments

Figure 9. Detrended BAI values (population averages in bold). Dotted line delineates pre- and post-treatment periods.

No significant difference in growth occurred between trees with fire scars or without (P=0.303). Scarred trees had slightly lower average BAI values before and after prescribed fire in burn-only treatments, whereas in harvest-burn plots the average growth of scarred trees was below that of unscarred trees after the first burn in 2003 (P=0.904) (Fig. 10). Although the average scar size was three times greater in harvest-burn plots than in burn-only plots (Table 9), the effects of scar area (P=0.698) and scar type (P=0.640, Fig. 4) were unrelated to tree growth.

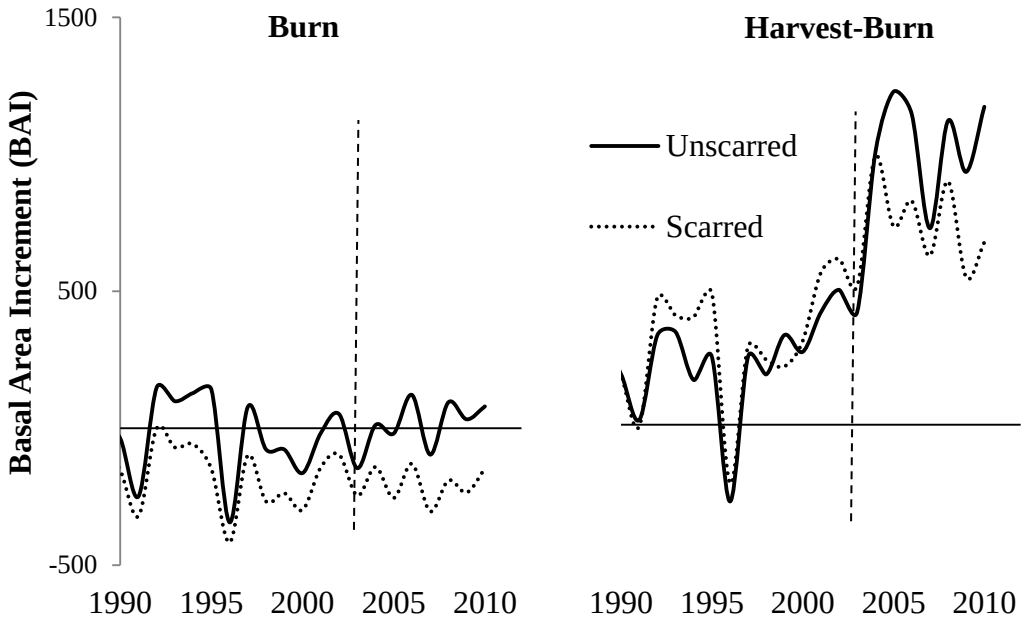


Figure 10. Detrended BAI averages for scarred vs. unscarred trees in each treatment.

Discussion

Cumulative Mortality

In woodland management, prescribed fire is applied to reduce the density of small diameter stems from mid and understory strata while retaining widely spaced dominant and codominant trees. Our results confirm that trees < 25 cm DBH accounted for 95% of mortality after moderate to high intensity (Relative Humidity ≈ 22.4, Mid Flame Winds ≈ 2.5 mph) controlled burns, and only 3% of large overstory trees became snags after two prescribed fires, compared to 1% in control (Table 8, Fig. 7). These findings are consistent with other studies regarding fire-induced mortality in oak-pine (Arthur et al. 1998) and mixed hardwood stands (Brose and Van Lear 1999a).

Commercial harvests (thinning from below) caused no additional mortality to residual trees in a fully stocked oak woodland; however, when applied prior to prescribed burns significantly increased the ratio of dead:live trees. Even after mechanical removal of fire-intolerant size classes and species in the mid- and understory, cumulative mortality and the basal area of snag trees was greater in harvest-burn treatments than in burn-only treatments (Fig. 7). This is likely due to increased fuel loads (up to 300%) which lead to greater fire-intensity (Kolaks et al. 2004, Kinkead et al. *in press*).

In addition to diameter, adaptations such as bark thickness, ability to sprout from dormant buds, and investment in chemical defense compounds also permit some species, especially in the white oak group, to withstand fire disturbances and maintain a competitive advantage (Hare 1965, Scrowcroft 1966, Hengst and Dawson 1993, Brose and Van Lear 1999a, Black et al. 2008,

Johnson et al. 2009). In this study, post oaks had the lowest cumulative mortality in burn and harvest-burn treatments (Table 8), followed by white oaks, which were four times more abundant in these treatments. Scarlet oak had the greatest overall mortality in burned plots, followed by black oaks which were the most abundant overstory species. Similar patterns have been reported by Paulsell (1957), Kabrick et al. (2004), and Dey and Hartman (2005).

Scar Damage

The most susceptible trees to scarring were species of the red oak group followed by the white oak group and hickories. Shortleaf pines were the least scarred trees with 6.3% and 13.2% in burn and harvest-burn plots, respectively. The fact that oak-competitors ("other species") appear more prone to scarring in burn-only treatments reflects the low survival of those species in harvest-burn plots (Table 8), probably as a function of smaller average diameters in species like red maple, elm, and blackgum. Although DBH was not related to scar size in this study (P=0.329), smaller diameter stems generally have greater cambial damage than do large trees due to lesser bark thickness and lower surface to mass ratio (Hare 1965, Hengst and Dawson 1993, Brose and Van Lear 1999a, Guyette and Stambaugh 2004). Hence, repeatedly burned stands may be void of healthy overstory trees in future rotations via long-term susceptibility to pathogens and decay (Burns 1955, Loomis 1974, Shigo 1984, Smith and Sutherland 2001). In another Missouri study, prescribed fire resulted in approximately a 10% value loss in the butt log of merchantable red oaks (Marschall 2012). If this is assumed applicable to other oak stands in the Ozarks Highlands, the cost of scar damage due to prescribed burning is relatively minor, especially since the probability of scarring a tree is only 0.17 in burn-only plots. However, this probability nearly doubles if a stand is harvested prior to burning (P=0.004), which causes greater fireline intensity and average wound size (Tables 5, 9). Surprisingly, scar size could not

be attributed to aspect, surrounding basal area, or tree diameter (see Guyette and Stambaugh 2004). Stevenson (2004) also found that although aspect was not related to scar size, wound dimensions did parallel fire severity which was much greater on south-facing aspects in both studies.

Growth

The application of two prescribed fires did not significantly improve or diminish growth of overstory white oaks examined in this study, except where harvested. This suggests that adaptations to fire disturbance are more strictly limited to tolerance, rather than a superior ability to capitalize on newly available resources. Moreover, marginal decreases in radial growth (-1.4% RWC) might indicate that fire-induced stress can potentially reduce diameter growth. A number of studies have also reported minor reductions in radial growth rates of overstory trees following periodic controlled burns (Scowcroft 1966, Huddle and Pallardy 1996, Fule´ et al. 2005). Although effects on tree growth are not conclusive, there is evidence to suggest that physical changes such as increased soil hydrophobicity and the removal of moisture-trapping leaf litter may decrease available water to plants (Scowcroft 1966, Stambaugh et al. 2006, Ponder et al. 2009). In addition, Boerner and Brinkman (2003) showed that prescribed burning lead to extensive loss of recalcitrant organic compounds and reduced enzyme activity in Ohio.

Mechanical thinning of mesic and other late successional species is often coupled with prescribed burning to achieve desired woodland structure and composition (Dey and Fan 2009, Kinkead et al. *in press*). While the increased growth and other benefits to the residual stand following commercial thinning have long been documented (Gingrich 1971, Schlesinger 1978), few studies have reported growth responses to the combination of prescribed fire and thinning. Fule´ et al. (2005) showed that thinning increased the growth of the residual trees in stands that

were burned in Arizona, but did not compare thin to thin+burn treatments. Our data suggested that significant reductions in radial growth were associated with burning in harvested stands (Fig. 9, Table 11). On average, RWC was 47% lower in stands that were burned 1 and 3 years after harvests. Similarly, BAI differences suggest that the magnitude of growth increases were two times greater in the harvest-only treatment than in harvest-burn treatment (Peterson et al. 1994, Stephens and Finney 2002**)**. We suspect that the added severity caused by residual fuels after harvests exacerbated stresses on overstory trees caused by burning, therefore reducing growth. Increased stresses may have included loss of fine-root mass and slowed nutrient mineralization or leaching (Reich et al. 2001, Varner et al. 2005)

Using both RWC and BAI differencing methods to measure growth response, a few generalizations can be made about how various size classes respond (Table 11). First, burning has the most negative effect on growth of small and intermediate size trees. Second, the response of these size classes to commercial thinning is much greater than in large trees. Overall, the combination of harvest and burn treatments increased growth of trees < 40 cm DBH. This may be favorable where mechanical removal is necessary to meet woodland objectives for two reasons, 1) harvests accelerate recruitment of small trees < 25 cm into more fire tolerant size classes 2) subsequent burns may be used to kill basal sprouts and ruderal sapling species < 12 cm DBH that respond to mechanical canopy removal (Fig. 7).

Additional site variables (Tables 6, 7) intended to reveal relationships between disturbance mechanisms and specific treatment effects were mostly unimportant in determining tree growth patterns (Table 12). The hypothesis that fire scar size, frequency, and type could be used to predict residual growth following prescribed burning and harvesting was not confirmed in this analysis, perhaps due to the overriding effect of other treatment induced changes. Guyette

and Kabrick (2002) report similar findings, asserting that while some growth reductions begin in years that fire scars were formed, no direct association can be made between decreased growth and scarring. Jemison (1943) also concluded that scarring had no effect on diameter growth in oak stands. Despite having the greatest scar sizes, long-lived white oaks show only slight reductions in growth compared to unburned or unscarred trees (Figs. 9,10).

As mixed hardwood stands age, crown architecture and species composition can vary substantially, creating a diverse canopy structure with numerous stratified layers (Rentch et al. 2002). Therefore it is difficult to predict the magnitude in growth response to thinning and burning without information about competition and light conditions for residual trees (Harrison et al. 1986). Many studies, including Rubino and McCarthy (2004), report slowing radial growth as a result of increased canopy closure. Unfortunately, our canopy data was collected nearly a decade after harvests occurred in our study, and did not accurately represent light availability to overstory trees used to analyze growth. Consequently, canopy openness was not helpful (P=0.953) in attributing growth increases to canopy disturbances that reduce competition for light (Table 12). Basal area around each stem was also recorded with the intention of explaining radial growth as a function of competition, but decreases in growth due to increased volume around each stem was only marginal (P=0.155).

Table 12.-- Recorded P>F values for each variable when added to the base growth model (top).

β0 (Intercept)	**β1 (Burn)**	**β2 (Harvest)**	**β3 (Burn*Harvest)**	**β4 (aspect)**	**βx (dbh)**	**Bx (surrounding basal area)**	**βx (age)**	**βx (slope)**	**βx (aspect)**	**βx (canopy openness)**	**AIC**
0.0005	0.217	**<0.0001**	**0.0055**	0.162							311.1
0.0006	0.160	**<0.0001**	**0.006**	0.103	**0.003**						311.5
0.0005	0.184	**0.002**	**0.008**	0.132		0.248					321.1
0.0005	0.194	**<0.0001**	**0.005**	0.202			0.256				321.8
0.0007	0.264	**<0.0001**	**0.005**	0.133				0.389			319.5
0.0005	0.217	**<0.0001**	**0.006**	0.162					0.133		311.1
0.0008	0.241	**<0.0001**	**0.006**	0.164						0.956	319.7

Many other studies also concluded that older, often larger, trees respond more conservatively to canopy disturbances such as thinning and burning (Lorimer and Frelich 1989, Nowacki and Abrams 1997, Rentch et al. 2002, Fule´ et al. 2005, Thorpe et al. 2007, Black et al. 2008). Although age was not significant (P=0.158), similar patterns were captured in our data for both age and DBH (Fig. 11). Most likely, significance of DBH (P=0.001) reflects the importance of expressing growth in terms of BAI (diameter-dependent metric), rather than the actual influence of diameter on tree growth. In other words, BAI values are inherently greater for trees in treated stands due to higher average DBH, potentially biasing treatment effects on tree growth (See also Fule´ et al. 2005, Voelker et al. 2008).

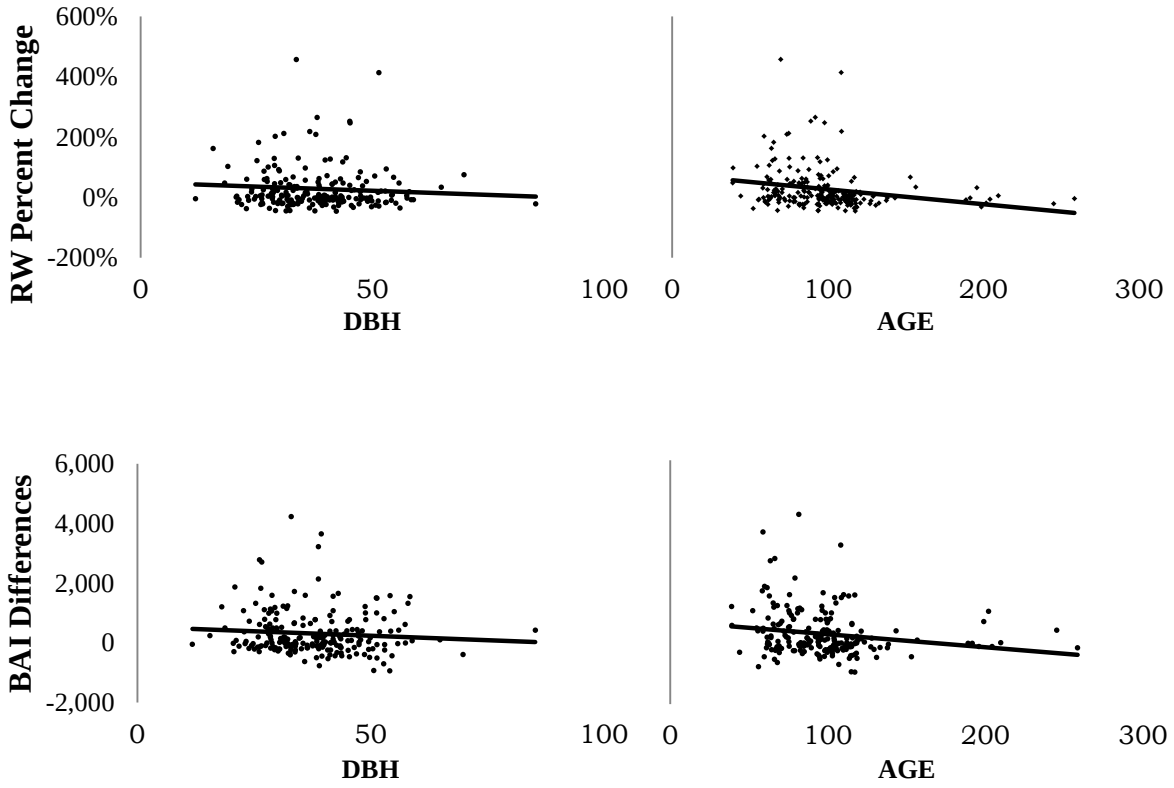


Figure 11. Growth responses averaged across all treatments to portray size and age dynamics over time.

Management Implications

Prescribed fire is a primary tool for restoring and maintaining oak woodlands, shown to improve herbaceous diversity and consistently remove small diameter woody vegetation. However, results presented here shows that mortality and scarring were significantly increased following two moderate to low-intensity controlled burns. These findings represent a fundamental challenge that managers face when targeting partial stocking levels of open woodlands; that is finding a balanced disturbance regime capable of re-opening canopy structure without causing severe damage to residual trees. Our data indicate that burning has minimal impacts on most overstory trees (> 25 cm DBH), especially white oak and shortleaf pine. Nonetheless, damage to smaller trees may reduce recruitment potential and cause substantial shifts in species composition or loss of structural diversity. Although burning may eliminate individuals with less tolerance to fire, it does not ensure that growing stock which is best suited for woodland habitat will be retained in the conversion from close-canopied stands. Therefore, mechanical harvests are also frequently conducted to meet structural objectives in the short-term (mitigating the need to risk higher intensity burns), and to manipulate composition more precisely. In addition, revenue from commercial thinning operations can often be used to fund management practices. However, a subsequent flush of woody re-growth is common in harvested stands of the central hardwood region (Dey and Jenson 2002, Kabrick et al. 2002), including in this study. Our data show that smaller size classes typically respond much greater to thinning than do larger trees in terms of radial growth, potentially making it even more difficult to setback woody vegetation in open woodlands. Yet, burns conducted after harvests cause significantly greater cumulative mortality than burning alone, especially of saplings and small diameter trees. Therefore, repeated prescribed fires should adequately offset the response of

sprouts and young trees. Where thinning is necessary to re-establish open conditions, managers should account for increased mortality during prescribed burns that follow. Likewise, where timber production is a central objective, burning should only be applied when it is needed to eliminate competitive mesic or early successional (r-strategist) species.

Conclusion

Residual overstory trees > 25 cm DBH were generally unaffected by prescribed fire. Red oaks tended to have greater mortality and scarring rates than white oaks, hickories, and shortleaf pine, especially where commercial thinning took place. Harvest treatments significantly increased tree growth but burning appears to greatly reduce the added growth benefits of those harvests. Even in stands that were not harvested, burning nominally reduced the growth of residual trees. Although trends in mortality or tree growth could not be attributed to scarring, likelihood estimates of scarring should guide managers with concerns about bole damage. Cumulative mortality rates of various diameter classifications and species are also applicable to prescription burn plans with specific restoration targets.

IV. TEMPORAL CHANGES FOLLOWING THINNING AND BURNING IN OAK-PINE WOODLANDS

Introduction

Prior to Euro-American settlement, frequent fires and other disturbances (grazing, drought, disease, etc.) mediated a continuum of forest, woodland, savanna, and prairie communities in the central US (Nuzzo 1986, Abrams 1992, Taft 1997, Faber-Langednoen 1999). Open woodlands were especially prevalent along the margins of the central Great Plains and eastern deciduous forest biome, where perennial prairie grasses and forbs grew beneath the partial canopies of fire-adapted tree species, e.g. oak (*Quercus* spp.) and pine (*Pinus* spp.) (Schoolcraft 1821, Houck 1908, Beilmann and Brenner 1951, Howell and Kucera 1956, Steyermark 1959, Schroeder 1981, Ladd 1991, Jenkins et al. 1997, Batek et al. 1999). Spatial and temporal distributions of woodland communities reflect several geologic, climatic, and anthropogenic conditions, especially in relation to historic fire regimes (Borchert 1950, Nuzzo 1986, Kabrick et al. 2002, Packard 1993, Taft 1997, Nigh 1999, Grabner et al. 2002, Guyette et al. 2002, Stambuagh and Guyette 2006).

Fire history information in the Midwestern USA has shown that mean fire return intervals (MFI) between 2-24 years have maintained oak-dominated ecosystems for several centuries, while MFI's of 3-6 years have been associated with open woodland character (Guyette and Cutter 1991, Packard 1993, Cutter and Guyette 1994, Guyette et al. 2002, Guyette et al. 2006). However, large-scale fragmentation and cessation of periodic fires has imperiled several plant and animal species that rely on structural and compositional attributes of woodland ecosystems (Callahan 1996, Churchwell and Mierzwa 2004, Missouri's Comprehensive Wildlife

Conservation Strategy 2005, Nelson 2005, Yates and Muzika 2006, Mann and Forbes 2007, Masters 2007, Mark Twain National Forest 2011).

In the absence of fire, woodland succession is influenced by other local disturbance events (i.e. wind, ice) and topo-edaphic qualities (i.e. slope, soil fertility, depth, acidity, etc.) (Grabner et al. 2002, Guyette and Kabrick 2002, Hutchinson et al. 2005, Dey and Fan 2009). Typically, productive sites exhibit greater densities and growth rates of shrub and tree species, while nutrient-poor sites often sustain canopy openness for longer periods (Pallardy et al. 1991 *in* Nelson 2004 and 2005). Eventually, shade-tolerant vegetation may increase at the expense of light-demanding (heliophytic) forbs and graminoids, permanently modifying the chemical and physical properties of the seed bank and fuel bed (Hutchinson et al. 2005, Stambaugh et al. 2006, Nowacki and Abrams 2008, Johnson et al. 2009). As litter accumulates and canopy cover increases, woodland herbs are often unable to germinate in fire-excluded stands (Jenkins et al. 1997, Mottl 2000, Iverson and Hutchinson 2002, Stambaugh et al. 2006).

In recent years, several studies have focused on reversing successional trends (Nigh et al. 1985, Pallardy et al. 1988, Pallardy et al. 1991, Kruger and Reich 1997, Varner et al. 2005, Glascow and Matlock 2006, Stan et al. 2006, Hutchinson et al. 2012). To restore the composition of fire-adapted flora and fauna in oak-pine systems, managers have increased the use of prescribed fire and mechanical thinning in woodlands throughout the Midwestern U.S. (Arthur et al. 1998, Sparks et al. 1998, Hartman and Heumann 2003, Cunningham 2007, Dey and Fan 2009, Mark Twain National Forest 2011). However, the effectiveness of these treatments for meeting woodland management objectives in the long-term remains unclear, in part due to the lack of quantitative information regarding the succession of plant communities in oak-pine stands following these silvicultural practices. As baseline restoration targets evolve from

qualitative descriptions to precise structural and compositional thresholds, data that describes vegetative responses to woodland management techniques become increasingly important. This study describes findings from woodland restoration sites over 10 years after silvicultural treatments began. Included are analyses of (1) structural overstory changes resulting from prescribed burning and mechanical harvests (2) relative abundance and diversity of understory species following treatments and (3) temporal patterns of tree species and herbaceous physiognomic (i.e. functional) groups such as forbs, graminoids, vines, and shrubs.

Methods

Study Area

This study was conducted in Reynolds County, MO, on Conservation Areas (CA) owned and managed by The Missouri Department of Conservation. Nested in the interior of the Ozark Highlands Region, study sites are fully stocked oak-hickory (*Carya*) and oak-pine forests with no prior management in the last four decades (Fig. 12). Historically occupied by open woodlands, second growth stands now dominate the steep, dissected hillslopes (<90 m relief) that characterize the Black River Oak/Pine Woodland/Forest Hills Landtype Association (Black River Hills LTA) and Current River Pine-Oak Woodland Dissected Plain LTA (Meinert et al. 1997, Nigh and Schroeder 2002). Soils are formed in clay residuum derived from cherty dolostone, as well as some hillslope sediment. Highly weathered Ultisols, formed in sandstones and cherty dolomites of the Roubidoux and Gasonade formations, dominate higher slope positions on the landscape. These soils were mapped as Poynor (loamy-skeletal over clayey, siliceous, semiactive, mesic Typic Paleudults), Scholton (loamy-skeletal, siliceous, active, mesic

Typic Fragiudults), and Clarksville (loamy-skeletal, siliceous, semiactive, mesic Typic Paleudults). Less weathered Alfisols are prevalent on lower slope positions, including the Series Alred (loamy-skeletal over clayey, siliceous, semiactive, mesic Typic Paleudalfs) and Rueter (loamy-skeletal, siliceous, active, mesic Typic Paleudalfs).

Experimental Design and Treatments

In 2001, three blocks (two at Clearwater Creek CA and one at Logan Creek CA, each approximately 25 hectares) were established, each containing four randomly assigned treatments; control, burn, harvest, harvest-burn (Table 13, Fig. 12). Within each treatment, permanent transects were installed on each of three aspects; north-facing slopes (aspect 315 to 45 degrees), ridge tops (slopes < 8 percent), and south-facing slopes (aspect 135 to 225 degrees). An inventory of all trees ≥ 4 cm diameter at breast height (DBH) was done in three randomly located 0.14 hectare plots on each transect. Along the same transect, 30 quadrats (1 m X 1 m) were also randomly located to record ground flora composition, including woody species, vines, and shrubs < 1 m tall. Percent cover of all herbaceous species and tree seedlings was estimated to the nearest percent. Certain genera such as *Carex*, *Desmodium*, and *Panicum* were only identified to species if included as 'woodland indicators'. This classification was assigned to several summer-flowering plant species known to inhabit open oak woodlands (Missouri Department of Conservation personnel, 2010). Ground flora inventories were repeated in summers of 2003, 2005, and 2011. Overstory was sampled again in 2003, 2006, and 2012.

Table 13. –Diagram illustrating treatment and aspect combinations.

	No-Burn		**Burn**	
No-Harvest	North	"Control"	North	"Burn"
	Ridge		Ridge	
	South		South	
Harvest	North	"Harvest"	North	"Harvest-Burn"
	Ridge		Ridge	
	South		South	

Commercial timber harvests were conducted in summer and fall of 2002, reducing stocking to about 40 percent (Gingrich 1967) by mechanically thinning from below. Nearly all small diameter stems were removed, as well as some dominant and co-dominant trees. Most white oak (*Quercus alba*) and shortleaf pine (*Pinus echinata*) were retained based on fire-tolerance and estimated pre-settlement abundance (Stambaugh et al. 2002, Cunningham 2007). Using the ring-fire method, the first prescribed burns were applied in 2003. Burn conditions recorded by Kolaks et al. (2004) indicate average air temperature of 1˚C, relative humidity of 22.4%, and wind speed at 4 kph. Fuel moisture was approximately 5.4% in 1-hour fuels, 9.8% in 10-hour fuels, 13.7% in 100-hour fuels, and 17.6% in 1000-hour fuels. One additional burn, under the same prescription, was conducted 2 years later for blocks 1 and 2, and three years later for block 3. Using passive flame height sensors and rate of spread (ROS) clocks, fire behavior

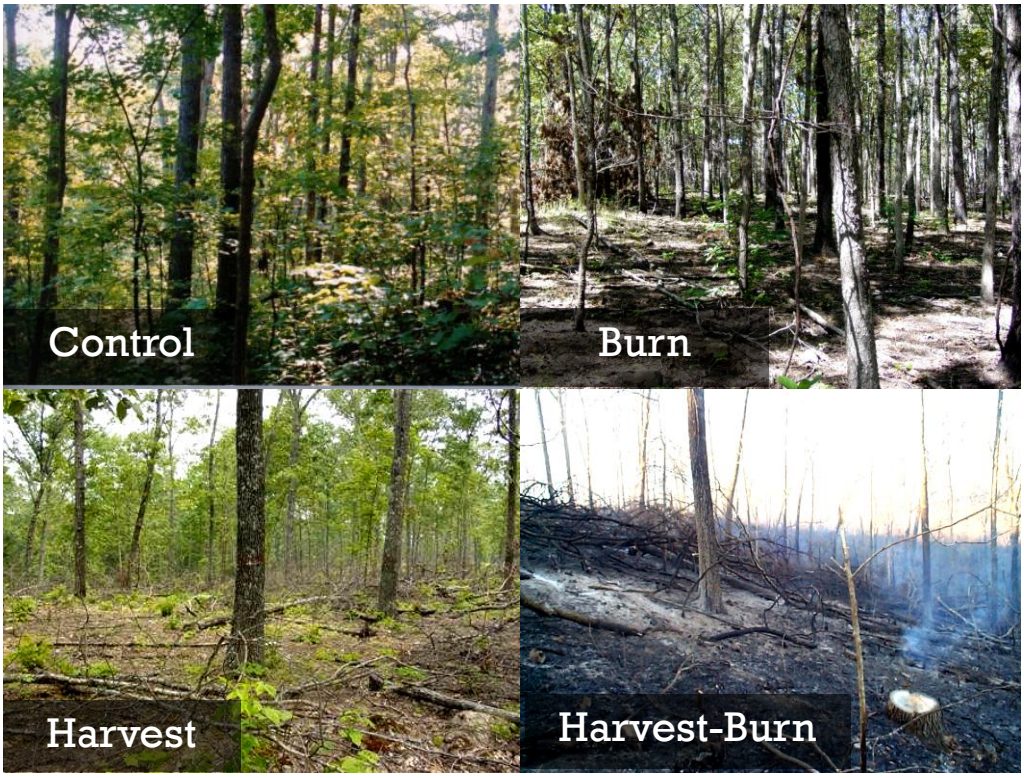
was characterized in 2003 (Table 14, Kolaks et al. 2004). In general, harvest-burn treatments had significantly greater total energy release (TER) than burn treatments. ROS was over 3 times faster on slopes than on ridges. Fireline intensity (FLI) was greatest on south slopes in particular, where flame lengths were significantly greater than on protected and ridge slopes (*For more fire behavior information see Kolaks et al. 2004*).

Table 14. – Observed parameters for fire behavior averaged across blocks in 2003 (adapted from Kolaks et al 2004), and 2005-2006 (adapted from Kinkead et al *in press*).

Treatment		*Rate of spread*		*Fireline intensity*		*Flame height*	*Total Energy Release*
		Nelson[1]	Byram[2]	Nelson[1]	Byram[2]		
		cm/min		BTU/m/sec		cm	(TER)
1st fire	Burn	73	37	141	72	50	2419
	Harvest-Burn	110	58	249	135	61	3466
2nd fire	Burn	ND[3]	ND	92	46	38	ND
	Harvest-Burn	ND	ND	236	131	46	ND

[1] Based on an equation by Nelson (1986)
[2] Based on an equation by Byram (1959)
[3] ND= Not Determined

A) Two years after harvest, one year after burn



B) Ten years after harvest, 6 year after burn

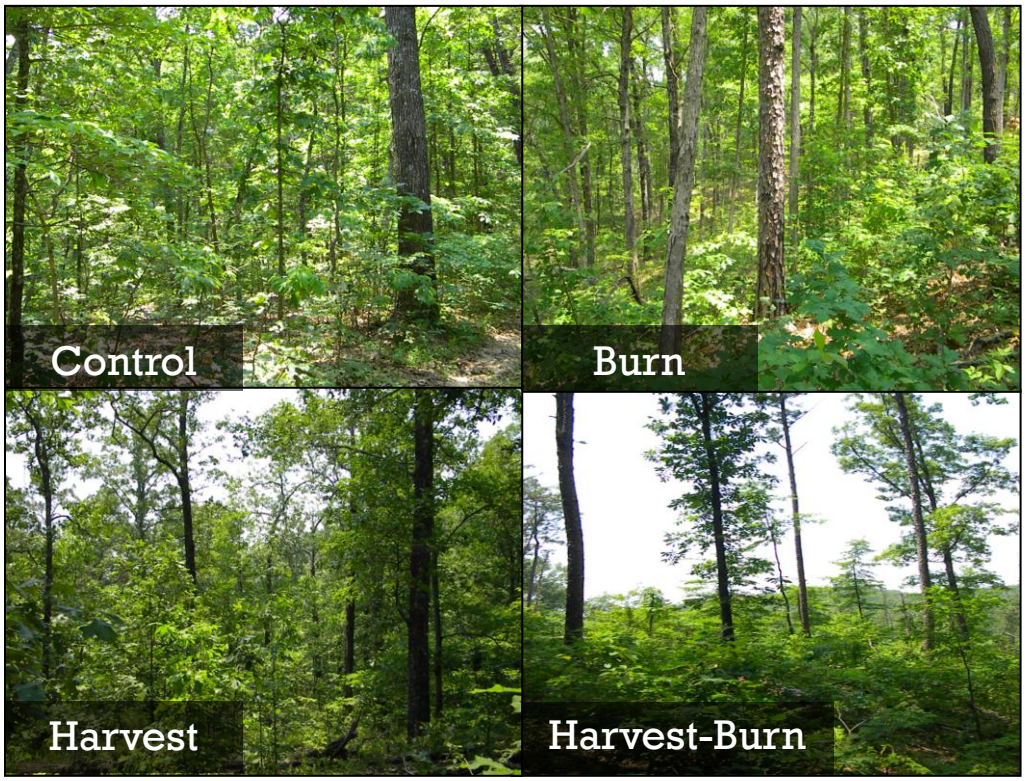


Figure 12. Treatment units (a) and (b) 10 years after harvests, and 6 years since last burn (2012).

In 2012, canopy photographs were collected using a hemispherical, or fish-eye lens, at 1 m above each overstory plot center. Gap Light Analyzer software was then used to calculate canopy openness percentages for each image by dividing pixels into sky and non-sky (version 2.0, Frazer et al. 1999). To estimate canopy openness prior to 2012, estimates based on stocking values were used according to Blizzard et al. (2013) [eq. 1.0].

[1.0] Canopy Openness = 41.8335 log10 (1.2077 + Percent Stocking)

Analysis

Generalized linear mixed models were used to determine significant (α =0.05) changes in canopy openness, stand density, percent stocking, understory diversity, and percent cover of various functional groups (Proc GLIMMIX, SAS version 9.3, SAS Institute, Inc., Cary, NC). Each model first ensured that regression slopes were linear and significant, and then tested if slopes for all treatments were parallel. Gamma distributions (default link function = log) were specified for Trees Per Hectare (TPH), percent stocking, percent cover of woodland indicators, percent cover of tree seedling species. Other physiognomic groups (i.e. forbs, shrubs, etc.) were analyzed using exponential distributions (default link function = log), and diversity was assigned a lognormal distribution (default link function = identity). Models (adjusted for parallel or unequal slopes) then simultaneously examined fixed effects including treatment (burn, harvest), aspect, year, and all interactions. With block as a random effect, burn treatments (whole plot) were tested with burn x block interaction as the error, while harvest and aspect (split-plot) effects were determined using block x burn x harvest x aspect as the error. For temporal data, year was

testes as a repeated effect using residual error (block x burn x harvest x aspect x year). Significant differences in fixed effects were determined using Tukey's Least Squares Means (LSmeans). The general form of the model is:

[2.0] Response Variable = B_0 + Burn (Harvest (Aspect)) + Year + E

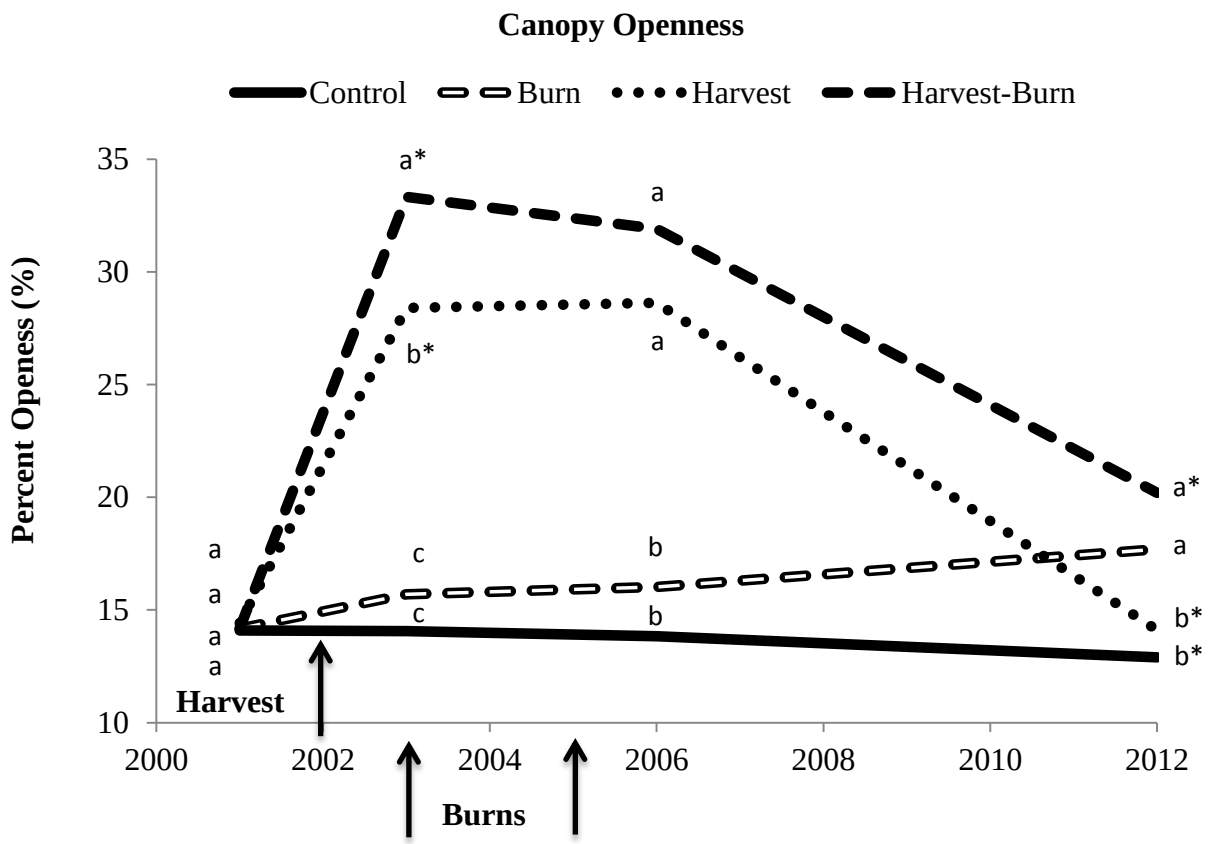
Results

Overstory Structure

Canopy Openness

Stocking based estimates of canopy structure prior to 2012 found that burning significantly (P=0.057) increased percent openness, as well as indicate that harvests more than doubled canopy openness above the regeneration strata (P=<0.001; Fig. 13). The additional openness in stands that were burned after harvests was significant after one fire (P=<0.001; 2003). Photographs taken in 2012 also show that on average, stands with two prescribed fires had significantly greater canopy openness than control stands (P=0.002; Fig. 13). Canopy openness was only marginally greater in harvest-burn treatments than in burn-only treatments, and no significant differences between north, ridge, or south aspects were detected. Low values in harvested stands (P=0.430) resulted from advanced reproduction and early seral vegetation that grew above 1 m tall in the 10 years after the thinning, often covering the camera lens aimed at capturing light to understory species. Therefore, canopy openness estimates based on stocking percentages [Eq. 1.0] were used to infer changes in canopy structure prior to 2012. Both photographs and model estimates indicate that prescribed fire increased canopy openness approximately 5% when compared to control and harvest treatments. Overall, the combination of

prescribed fire and commercial thinning resulted in the greatest percent canopy openness (Fig. 13).



[1] Letters indicate significant differences between treatments for a given year

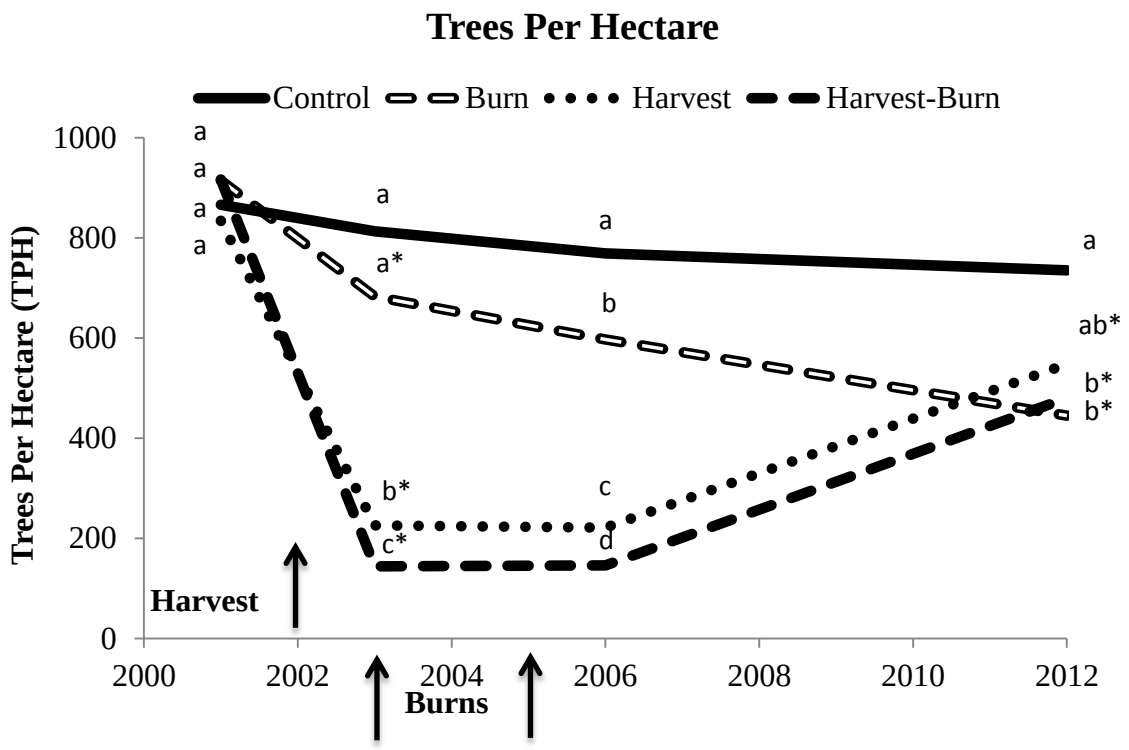
[2] Asterisks indicate significant differences between years

Figure 13. Canopy openness estimations based on stocking models by Blizzard et al. (2013) for years 2001-2006, and hemispherical photographs taken in 2012. Openess in harvested stands in 2012 was masked by vegetation that had grown above 1 m (lens ht) exactly one decade after commercial thinning took place.

Trees Per Hectare

The combined effect of two controlled burns significantly reduced overstory tree density (P=0.041). On average, the single prescribed fire decreased tree density by approximately 232 trees per hectare (TPH), and a second burn caused an additional 86 TPH reduction (Fig. 14). In

burn-only plots, TPH became significantly less than in control in 2006 (P=0.012). However, greater decreases in tree density of burned plots occurred during the fire-free interval from 2006 to 2012 (P=0.010) than immediately after the second fire. This decrease left burn-only plots with less than half of the stems recorded in pretreatment sampling (P=<0.001). Similar TPH declines were observed during this time period in control plots, though not significant (P=0.568). As expected, commercial harvest treatments had the greatest impact on stem density, decreasing TPH 73% (P=<0.001). The additional reduction of TPH in harvested stand caused by two subsequent burns (≈ 80 TPH) was also significant one year after each fire (P=0.005 and P=0.010, respectively; Fig. 14). Harvest-burn plots sustained the lowest tree density of all treatments until inventories in 2012 revealed that burn-only treatments had declined to 445 TPH. Aspect was not important in explaining differences in tree density (P=0.835).



[1] Letters indicate significant differences between treatments
[2] Asterisks indicate significant differences between years

Figure 14. Trees densities following silvicultural treatments in fully stocked oak woodland sites.

Percent Stocking

Prescribed fire caused only a marginal 7% decrease in stocking after one burn, and an additional 2% after the second. However, 6 years after the last fire was conducted, stocking values had declined an additional 14% in burn-only treatments. Nearly a decade after harvests were conducted, average stocking in harvest treatments was still 45% lower than pretreatment levels (P=<0.001), and 57% lower in harvest-burn treatments (P=<0.001). Overall, harvest-burn treatments maintained significantly lower stocking than burn-only (P=<0.001) and harvest-only treatments (P=0.002). Gradual stocking increases after 2006 occurred in both treatments, although this effect was only significant in harvested stands where fire was excluded (P=0.035; Fig. 15). Aspect did not have a significant effect on percent stocking (P=0.397).

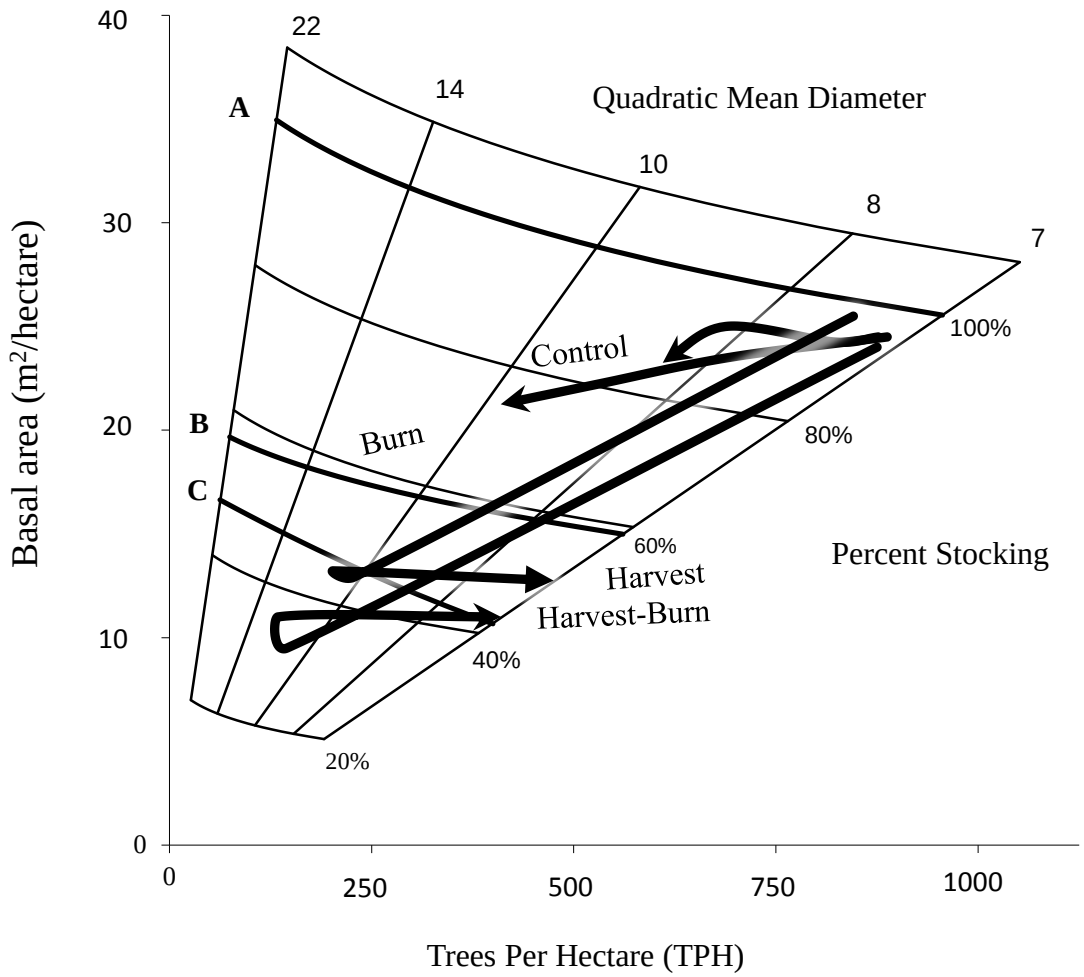


Figure 15. Gingrich (1967) diagram portraying stocking trajectories of four woodland management treatments over a ten year period.

Sapling Response

Pretreatment diameter distributions in 2001 resemble a traditional reverse J-curve, common to most undisturbed mixed hardwood stands in the central hardwoods region (Johnson et al. 2009; Fig. 16). Prescribed fire reduced the density of sapling (4-12 cm DBH) stems 33% after one burn (P=0.046) and an additional 20% after the second burn (61% total by 2012; P=<0.001). Approximately 16% of saplings size stems (approx. 85 TPH) were retained after harvests, which stayed constant as of 2006 (P=<0.001; Figs. 14, 16). However, by 2012 sapling

size stems were 5x more dense than immediately post-harvest TPH (P=<0.001). Burning in harvested plots reduced saplings 94% after one burn (P=<0.001), but only an additional 18% after a second burn (P=<0.562). Compared to harvest-only, harvest-burn treatments had significantly fewer saplings after in 2003 (P=0.006) and 2006 (P=0.004). After 6 fire-free years, sapling density in harvest-burn plots became 13x more abundant (P=<0.001), although still 38 % less dense (232 TPH) than pre-treatment (Figs. 14, 16).

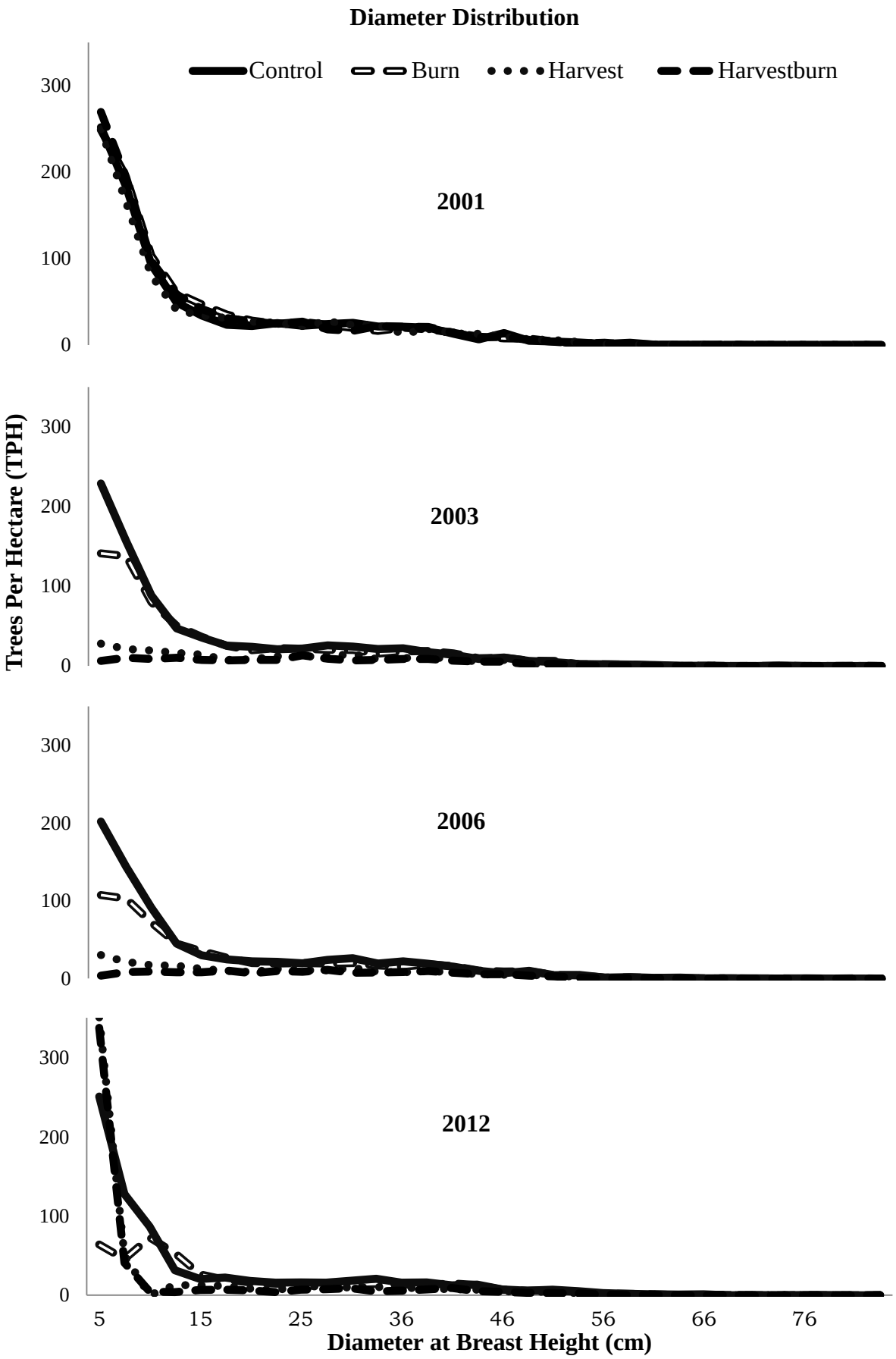

Diameter Distribution
Control
Burn
Harvest
Harvestburn
2001
2003
2006
2012
Trees Per Hectare (TPH)
0
100
200
300
5
15
25
36
46
56
66
76
Diameter at Breast Height (cm)

Figure 16. Diameter distribution of woodland management sites in the Ozark Highlands Ecoregion during sample inventories.

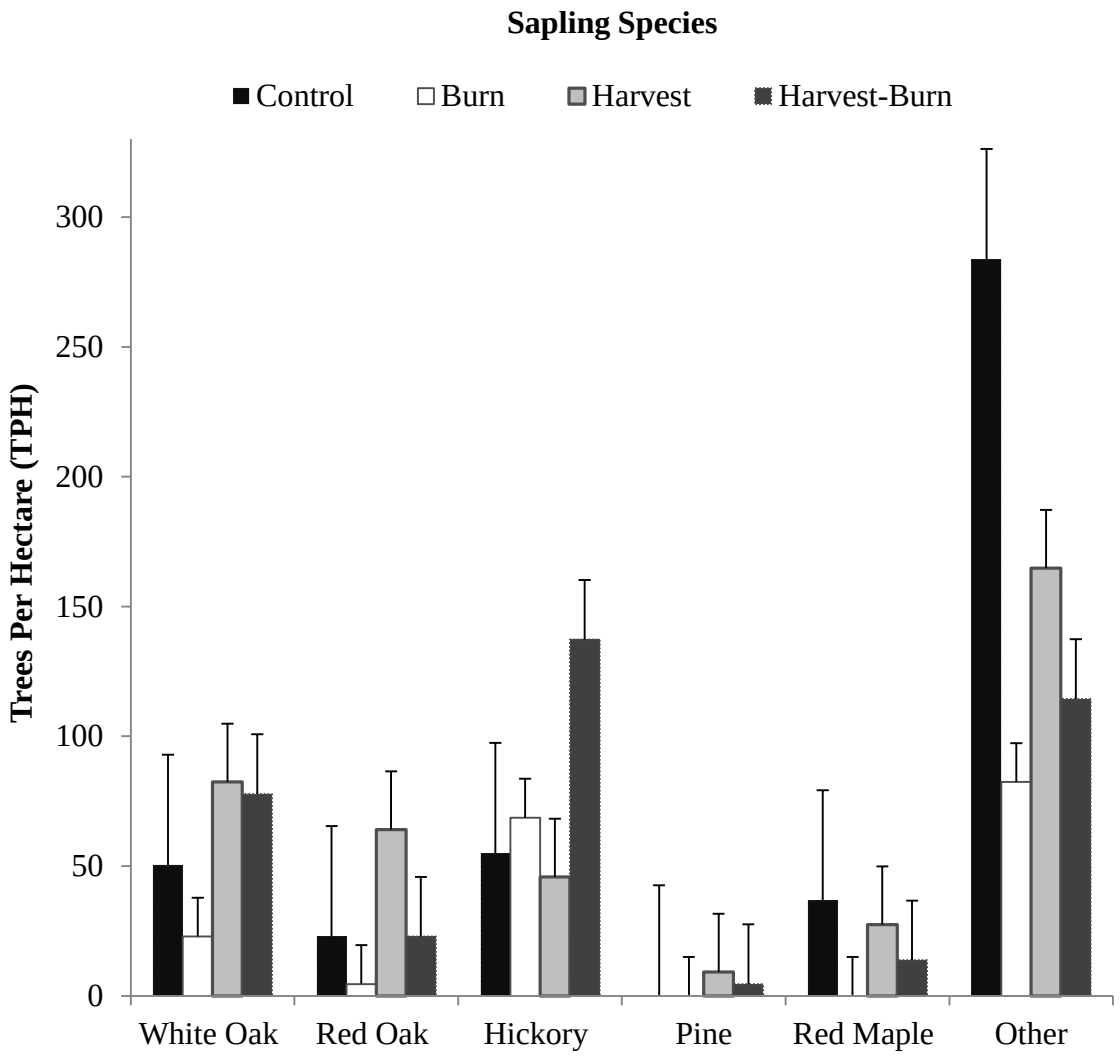


Figure 17. Abundance of sapling-size (4-12 cm) trees in common species groups ten years after first treatments were initiated. 'Other' species include blackgum (*Nyssa sylvatica*), elm spp. (*Ulmus spp*.), flowering dogwood (*Cornus florida*), sassafras (*Sassafras albidum*), and black cherry (*Prunus serotina*). Bars indicate standard error (±1).

Dominant species groups of the sapling strata were primarily oak and hickory, although mesic oak-competitor species (e.g. "other"), including blackgum and black cherry were collectively more abundant (Fig. 17). Both white and red oak saplings were most abundant in

harvest-only treatments, and lowest in burn-only treatments. In fact, burned plots had fewer saplings than any other treatment for all species except hickory, which was most abundant in harvest-burn plots. The greatest abundance of oak competitor species was in control plots. Shortleaf pine and red maple were relatively less frequent, and completely absent from burn-only treatments in 2012. Overall, "other" species outnumbered oaks nearly 4 to 1 in control and burn plots, but were less abundant than oaks in harvest-only sites.

Understory Composition

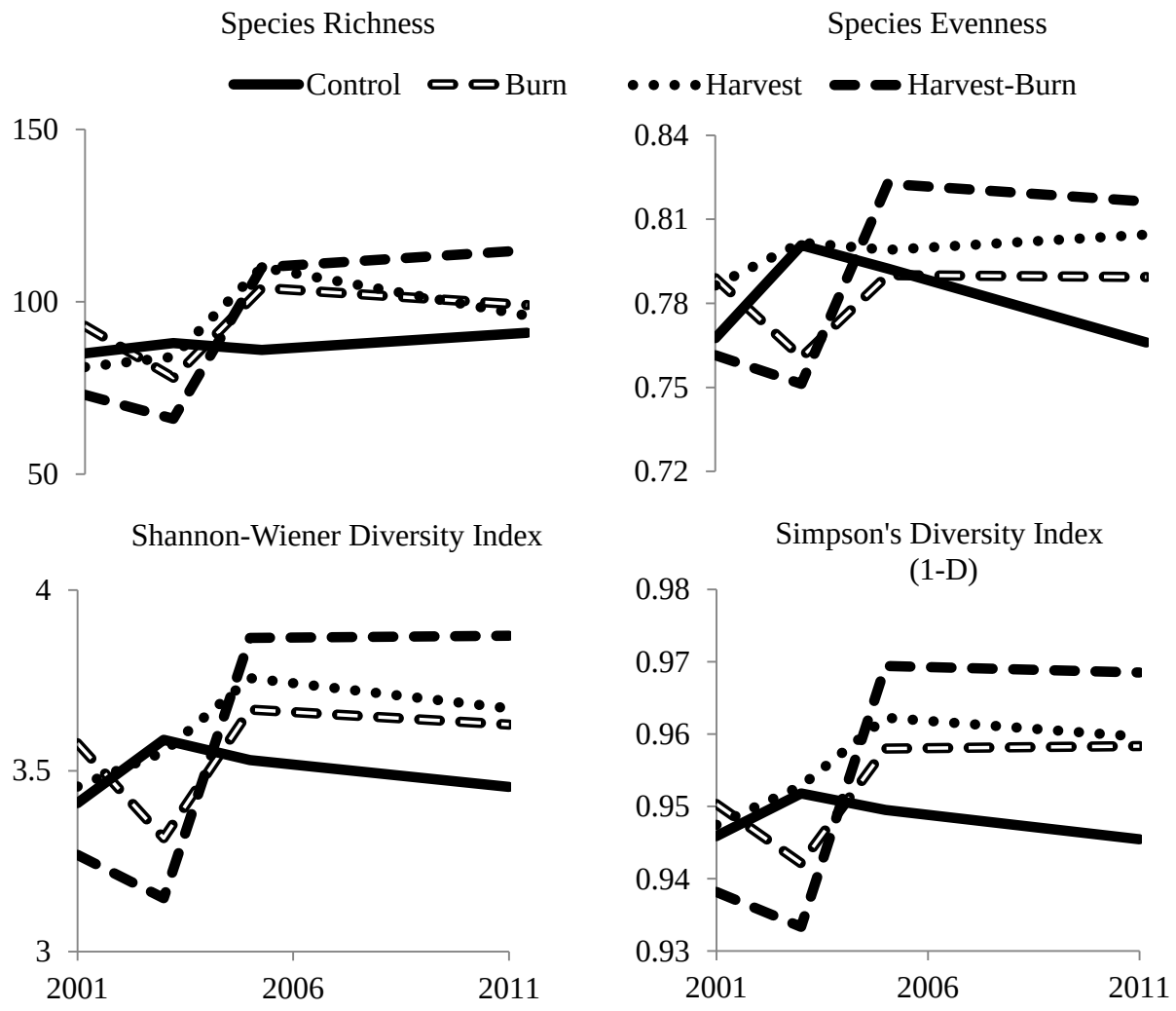


Figure 18. Understory characteristics of each treatment during a 10-year period.

Although no significant changes in diversity could be attributed prescribed burning (P=0.671) or commercial thinning (P=0.363), some trends were evident (Fig. 18). By 2012, the harvest-burn led all other treatments in species richness, evenness, and diversity, despite having the lowest values for each in 2003. Harvest-only plots had slightly greater post-treatment evenness and diversity than burn-only plots, and recent inventories showed declining richness and diversity in both treatments (Fig. 18). Control plots remained relatively consistent throughout a decade of monitoring, and had the least amount of species diversity, richness, and evenness after all treatments were applied. Aspect effects were also not significant (P=0.837).

Our analysis also indicated no significant treatment effects on coverage of any physiognomic groups, except for graminoids (i.e. sedges, rushes, and grasses) in response to commercial harvests (P=0.002; Fig. 19). With the exception of trees, harvest-burn treatments resulted in greater coverages of each lifeform than any other treatment. Aspect significantly influenced coverages of each physiognomic groups, where graminoid (P=0.002), shrub (P=0.034), and vine (P=0.006) species were favored on drier south slopes, and forbs (P=<0.001) and trees (P=0.024) were favored on north slopes.

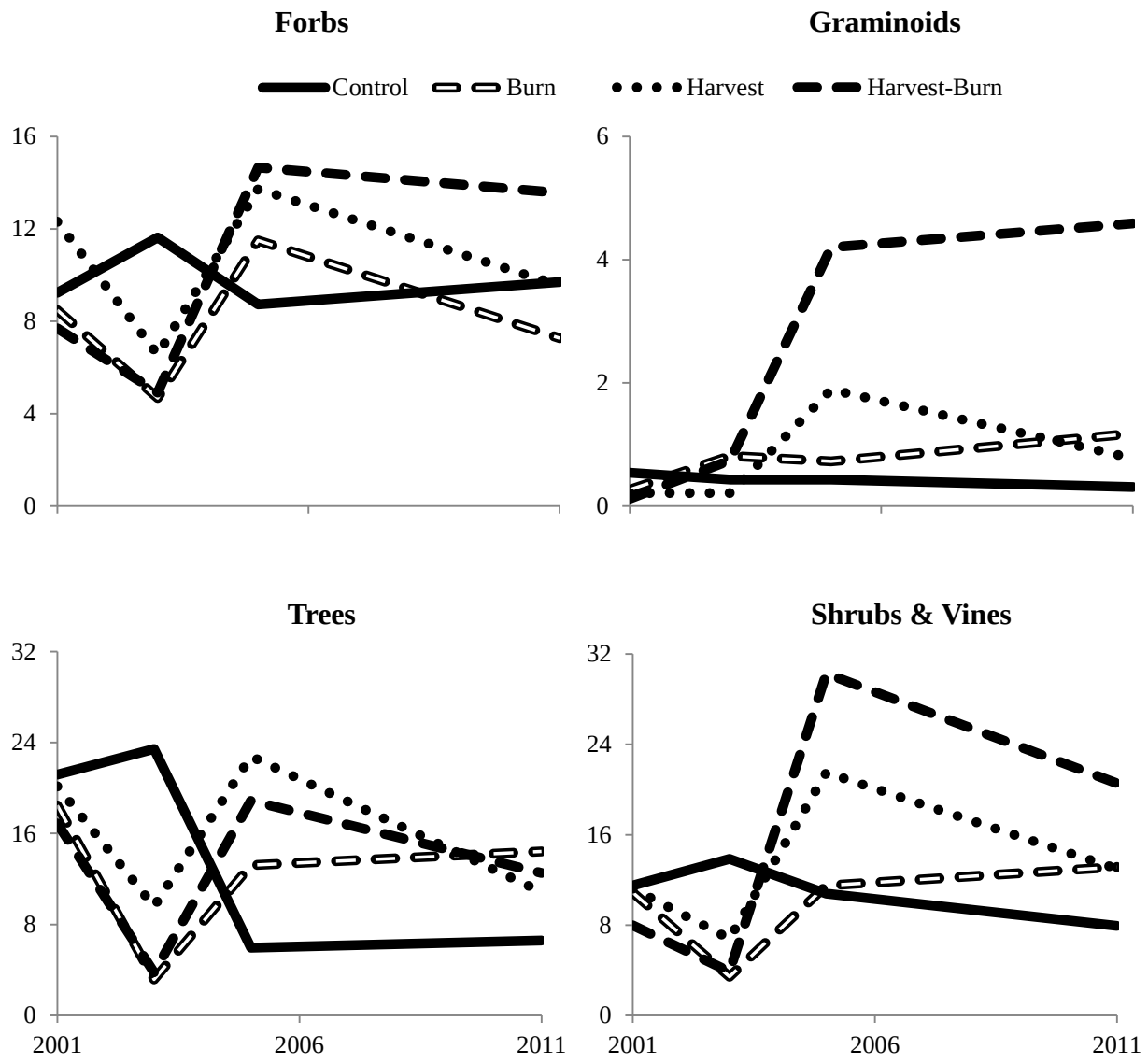
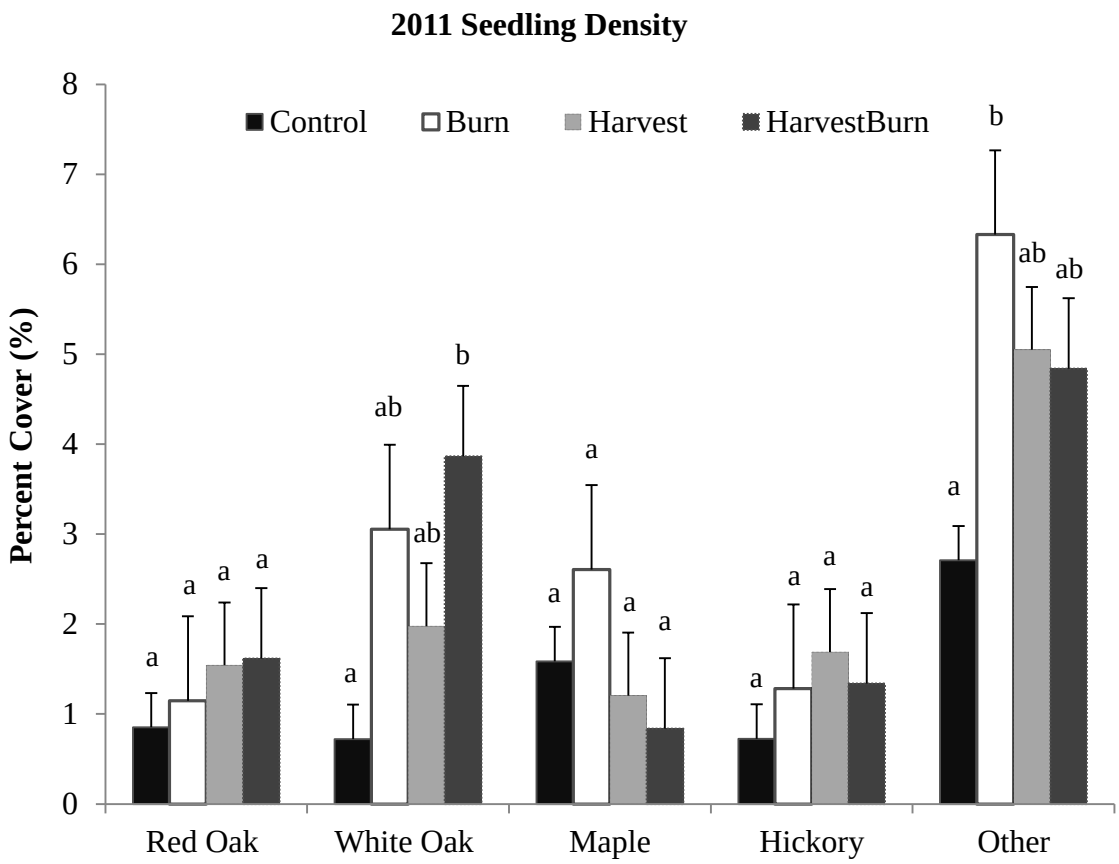


Figure 19. Average percent cover of each vegetative physiognomic (i.e. functional) group over ten years.

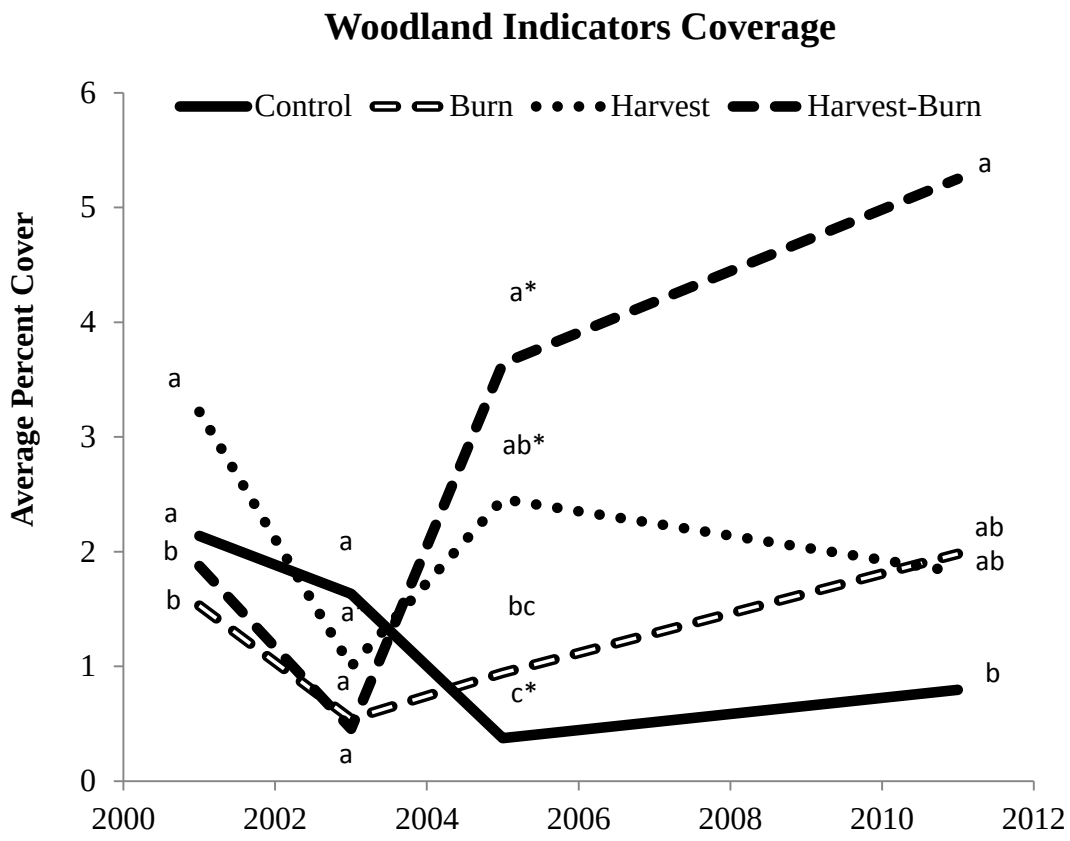
Decreases in the coverage of tree seedlings in 2003 following treatments were similar to subsequent decreases of woody vegetation in control plots around 2005. Although tree regeneration in harvest and harvest-burn stands appear to decrease after 2005, increased abundance of sapling size trees in overstory sampling data (Fig. 16) suggests that most stems had

simply surpassed size requirements for the regeneration layer inventory (> 1 m tall and > 4 cm DBH). Average percent cover of white oak in the regeneration strata was significantly greater in harvest-burn plots than in control (P=0.037; Fig. 20). Red oak and hickory species also had slightly greater coverage in treated stands than in control, although only marginal. Interestingly, burn-only treatments resulted in greater coverages of maple and oak-competitor seedlings than any other treatments.



[1] Letters indicate significant differences between treatments

Figure 20. Composition of the understory seedling/regeneration layer. Other species include blackgum (*Nyssa sylvatica*), elm spp. (*Ulmus spp*.), flowering dogwood (*Cornus florida*), sassafras (*Sassafras albidum*), and black cherry (*Prunus serotina*). Bars indicate standard error (±1).



[1] Letters indicate significant differences between treatments for a given year
[2] Asterisks indicate significant differences between years

Figure 21. Changes in percent coverage of woodland indicator species within each applied treatment.

Analysis of woodland indicator species shows that the effect of prescribed burning (P=0.532) is less important than that of commercial harvest treatments (P=<0.003). However, the response of woodland indicators in harvest plots did not persist compared to burn plots, which had greater average percent cover by 2011 (Fig. 21). The combination of these management techniques resulted in the greatest coverage of woodland indicators in the understory strata, where coverage was over 6x greater than in control plots, 3x greater than in harvest-only, and nearly twice that of burn-only plots (Fig. 21). Prescribed fire alone caused coverage of this

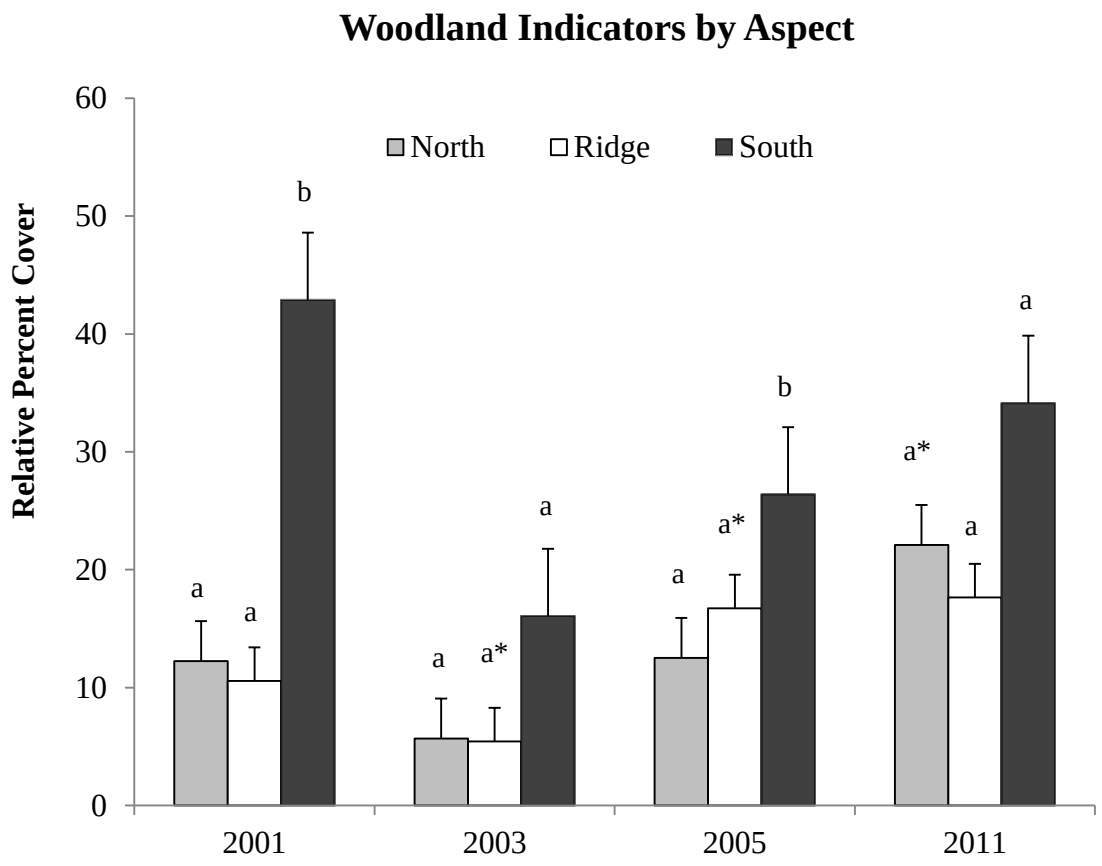
functional group to more than double from pretreatment levels, and become 4x greater than in the control by 2011. Aspect was a significant predictor of woodland indicator coverages, where south slopes favored these species more than north and ridge slopes (P=<0.001; Fig. 22).

Overall, the maximum number of woodland indicator species found any a single treatment was 20, observed in harvest sites in 2005 and harvest-burn sites in 2011 (Table 15). In both of those years, harvest-burn treatments afforded the greatest occurrence of indicator species, followed by harvest, burn, and control plots. After all treatments were conducted, the most frequent woodland indicators were trailing lespedeza (*Lespedeza procumbens)*, common dittany *(Cunila origanoides)*, and round-leaved tick trefoil *(Desmodium rotundifolium)*, respectively. Less dominant, more conservative species included wild quinine (*Parthenium integrifolium*) and yellow pimpernel (*Taenidia integerrima)*.

Table 15. Sums of woodland indicator data for each sample year.

	2001		**2003**		**2005**		**2011**	
	# Spp[1]	n	# Spp[2]	n	# Spp	n	# Spp	n
Control	12	119	12	88	10	38	15	58
Burn	12	92	13	75	17	82	15	96
Harvest	15	133	13	62	20	112	15	106
Harvest-Burn	13	93	14	59	17	200	20	223

[1] Number of species per plot
[2] Number of plots with woodland indicators



[1] Letters indicate significant differences between aspects
[2] Asterisks indicate significant differences between years

Figure 22. Woodland indicator coverages averaged for all treatments by aspect and year. Bars indicate standard error.

Discussion

Overstory Structure

Canopy Openness

Classification of woodland communities is often based on the extent of canopy openness (25-60%, Leach and Ross 1955, Faber-Langednoen 1999) or closure (30-100%, Nelson 2005) in a particular stand. Although stocking-based estimates of canopy openness indicate that harvest and harvest-burn treatments achieved > 30 percent openness, photographs taken in 2012 show

that actual openness was much less (Fig. 13). This can be attributed to the substantial layer of basal sprouts (oak, hickory) and *Rubus* species in harvested areas which caused heavy shading of the ground layer. Brudvig and Asbjornsen (2007) also report heavy woody encroachment of open oak systems only 3 years after similar harvests (> 48% overstory removal) were conducted in an Iowa, USA, while Hutchinson et. al. (2012) found that modest reductions of sapling and midstory trees after prescribed fire caused significant increases in canopy openness. Likewise, our data suggest that burning alone significantly decreased small diameter stems (Fig. 16), therefore increasing available light to the understory (Fig. 13). However, only the combination of harvest and burn operations resulted in greater than 20 percent canopy openness, which is the expected threshold for compositional shifts toward woodland and savanna ground flora (Bray 1958, Bowles et al. 2007).

Tree Density and Stocking

Nelson (1997) used original surveys (1815) from the General Land Office (GLO) to estimate presettlement tree densities of the Ozark Plateau in Missouri. Although more than 87% of the region was 'timbered', Nelson concluded that open woodlands were the dominant community type of upland sites, averaging approximately 79 TPH. By this standard, two prescribed fires did not achieve desired reference condition, reaching only 445 TPH (Fig. 14). In fact, even harvest-burn treatments had nearly double Nelson's estimated presettlement density. However, Hanberry et al. (2012) shows that GLO survey records likely underestimate presettlement tree densities by as much as 25- 55% of the actual density, due to non-random selection of reference trees (not) nearest to section corners. In this case, a 51% decline in overstory TPH due to prescribed burning would have reduced densities very near average

presettlement estimations, as well as densities found in similar studies (Franklin et al. 2003, Stan et al. 2006, Hutchinson et al. 2012).

A separate report evaluating cumulative mortality indicates that prescribed burns conducted in this study had a significant effect (P=<0.001) on the percent of trees killed by each fire, but caused < 10% mortality in intermediate trees (26-40 cm) and < 5% in large (>40 cm) trees (Kinkead et al. *in prep*; see also Dey and Fan 2009). Although fire severity may have resulted in slightly greater overall mortality than expected, impacts were generally confined to small diameter stems (Fig. 16), at least initially. Our data show that burning only reduced stocking 9 percent after the second fire, but an additional 14 percent decrease was observed 6 years post-treatment, compared to 6 percent in control. Pre- to post-treatment reductions in stocking observed in control stands suggest a natural dynamic presumed to influence all stands, while lag reductions in burned stands were likely due to fire-induced damage and changes in micro-climate (e.g. increased radiation, windspeed) occurring after 2006 (Kinkead et al. *in prep*, Nauertz et al. 2004, Nowacki and Abrams 2008). Varner et al. (2005) shows that delayed mortality and tree stress (> 2 years post-fire) may be explained by loss of root carbohydrates due to heating of mineral soil.

Commercial harvests reduced stocking to less than 50 percent (< 225 TPH) in all stands. However, seedling and sapling densities nearly tripled from 2006 to 2012, even where burns followed. Basal sprouts accounted for the majority of woody re-growth, where harvested stands had 7x more stems between 4-6 cm DBH than saplings within 7-12 cm (Fig 16). Many speculate that onset of such dense woody regrowth is characteristic of early succession in oak woodlands, as depicted in GLO records describing “scrub oak” thickets of fire-stunted, multi-stemmed sprout clumps (Schroeder 1981, Ladd 1991, Abrams 1992, Johnson et al. 2009). Yet, our data

suggest that such prolific sprouting is only triggered by disturbances severe enough to remove ≥ 50% of the canopy, and that moderate severity surface fires significantly *reduced* sapling densities. Therefore, given the limited frequency of stand-replacing crown fires in the Midwestern U.S. (Westin 1992), most historic fire events likely contributed to the open woodland character illustrated in our burn-only treatments.

In fact, the two fires conducted in this study (burn-only) damaged fewer trees with the potential to sprout (Kinkead et al. *in prep*) and retained adequate cover of residual overstory trees (22 m^2/hectare) to limit sprout growth. Both Dey and Jenson (2002) and Dey and Hartman (2005) show that average overstory basal area between 14-18 m^2/hectare (≈70 percent stocking) substantially inhibits growth of oak stump sprouts in the Missouri Ozarks. Basal re-sprout of oak and hickory species comprised most of the woody growth in harvested stands, and combined for greater abundance than all 'other' competitors (sassafras, blackgum, elm, dogwood, black cherry) (Fig. 17). Similar to Hartman and Heumann (2003), we found that white oak decreased in smaller percentages than red oak, while one of the most abundant midstory species, dogwood, was reduced approximately 1/3 in after repeated burns.

Understory Composition

Understory composition showed a clear response to burn-only and harvest-only treatments, although only harvest-burn sites had significantly greater evenness, richness, and diversity than control (P=<0.001, Shannon 1948, Simpson 1949). Nearly equivalent increases in diversity were recorded in burn-only and harvest-only stands in 2005, but each eventually began to decline without continued management. We found that two prescribed fires increased ground flora richness by 14 species, although values returned to pretreatment levels six years after the last burn. Thinnings from below afforded a 29 species increase, but richness was reduced to half

by 2011. Phillips et al. (2007) also reports transient increases in species richness following burn-only and thin-only treatments in Ohio, USA, and notes that thin-only stands had fewer species than in control and burn plots after only 3 years, while stands that were thinned and burned had the greatest overall richness. Likewise, harvest-burn treatments in our study had 42 more species present a decade after this study was initiated (115 total, not including taxa only identified to genus), and 16 more species than in control (Fig. 17).

Low coverages of each physiognomic group in 2003 may be attributed to the intensity and timing of prescribed burns stands (mid-April). Although rhizomes of perennial herbs and grasses are generally protected in late dormant season burns (Howe 1994, Iverson and Hutchinson et al. 2002, Knapp et al. 2007), fire behavior data and decreased frequencies of nearly all seed-banking species suggest that germination and establishment was inhibited immediately after the first burn. Thinning from below caused similar decreases in percent cover of most lifeforms, where residual fuel loads (Kolaks et al. 2004) likely prevented woodland species from achieving light and growing space requirements (Bowles et al. 2007). By 2005, desired treatment effects were more apparent on harvest-only and burn-only sites, though did not persist toward the end of our 10-year evaluation. Hartman and Heumann (2003) point out the resilience of certain herbaceous perennial species (e.g. *Helianthus hirsutus*) in burned sites, especially legumes such as *Desmodium* spp. and *Amphicarpa* spp. which were the two most common genera of legumes in our study. After all treatments, legume abundance increased an average of 28.2 % in the burn-only treatment, 16.9 % in harvest, 102.1 % in harvest-burn, and 3.1 % in control.

In total, forbs exhibited minor coverage increases (3.01%) following the second burn, but returned to pretreatment levels after six fire-free growing seasons. The same effect was observed in harvest-only plots, with each physiognomic group except graminoids declining from 2005 to present. Shading by small diameter woody re-growth is most likely the cause of diminishing frequencies and coverages after 2005 (Figs. 16, 19) (Kabrick et al. 2002, Franklin et al. 2003, Brudvig and asbjornsen 2007, Dey and Fan 2009). Burning in harvested stands significantly reduced sapling response (Figs. 16, 17), allowing harvest-burn treatments to sustain the greatest abundance of forbs and graminoids. Despite declining coverages, forb cover in harvest-burn treatments remained 5% greater in 2011, and graminoids were actually more abundant in 2011 than any other year (Fig. 19). This fits accordingly with the observed lag in declining overstory strucuture and subsequent increases in light penetration 6 years after the last fire.

Shrub and vine coverages were considerably greater in harvest-only treatments than in burn-only treatments in 2003 and 2005. The dominant taxa, including *Vaccinium* spp., *Rubus* spp., *Rhus* spp., and *Vitis* spp., each had the greatest coverage in harvest-burn plots and the least in control, contrary to Albrecht and McCarthy (2006). The appearance of declining tree species in regeneration layer (Fig. 20) is more clearly understood when examining diameter distributions after 2005 (Fig. 16), which show the increase in stems <12 cm DBH. Although burn-only plots contained the greatest percent cover of woody species in 2011, ground flora inventories no longer included most sprouts and woody re-growth in harvested sites (Fig. 16). Of reproductive stems, percent cover of the white oak group (primarily *Quercus alba* and *Quercus stellata*) was significantly greater in harvest-burn plots than in control (P=0.037; Fig. 20). Dey and Hartman 2005 show that after three prescribed burns, seedling (5 cm) survival probabilities were greatest in black oak (88%), post oak (88%), and white oak (80%), while blackgum (2%) was severely

affected by repeated fire. Interestingly, our data show that blackgum coverages in the herbaceous layer decreased 82% after one prescribed burn, but increased after a second burn to more than 2x pretreatment coverage. In fact, "other" species, including blackgum and red maple, were more abundant in the regeneration strata of burn-only plots than any other treatment (Hutchinson et al. 2012; Fig. 20), contradicting reports of similar studies (Kruger and Reich 1997, Glascow and Matlock 2006). Stan et al. (2006) also found that white oak seedlings were less responsive than competitive hardwood species such as black cherry, slippery elm, white ash, and bitternut hickory following two prescribed burns in a woodland community in northern Illinois. The pretreatment cover of non-oaks is probably an important factor in compositional patterns after burning, especially when a lack of disturbance has allowed shade tolerant species to store carbohydrate reserves adequate for post-fire re-sprouting (Brose and Van Lear 1998b, Dey and Fan 2009, Johnson et al. 2009). Kruger and Reich (1997) note that basal sprouts appeared on woody vegetation as early as 10 to 20 days post-fire in Wisconsin, highlighting the necessity of multiple fires to exhaust carbohydrate reserves of oak competitors (black cherry, sassafras, hickory spp.), thus favoring species with greater root: shoot ratios (Brose and Van Lear 1999b, Varner et al. 2005, Johnson et al. 2009, Hutchinson et al. 2012).

The last functional group analyzed in this study is comprised of woodland indicator species, which are generally heliophytic summer herbs that require relatively high light in order to produce flowers and seeds. Typically, these herbaceous plants are associated with prairie and grassland ecosystems, but are also indicative of open canopied savanna and woodland communities where sufficient light reaches the ground-layer (Taft 1997). Restoring species in a depauperate woodland ground flora is challenging because most individuals produce small quantities of low-viability seed, with specific germination requirements and long periods to reach

reproductive maturity (Mottl et al 2006). In this study, the combination of harvesting and burning lead to significantly greater woodland indicator coverages than control (P=<0.001), while burn-only and harvest-only treatments resulted in nearly identical responses (Fig. 21). These findings support the notion that fire promotes woodland species via different ecological processes (i.e. nutrient cycling, seed scarification, litter removal, etc.) than does the environmental heterogeneity caused by structural alteration (i.e. available light, soil condition, species dominance) (Zenner et al. 2006, Bowles et al. 2007). Although commercial thinning had a greater effect (p=0.003) on woodland indicator coverages than burning (P=0.532), prescribed fire appears to sustain compositions of woodland species for a greater length of time (Fig. 21). Importance of aspect (P=<0.001) suggests that drier, south-facing slopes facilitate greater disturbance intensities and favor shade-intolerant understory vegetation (Fig. 22).

Management Implications and Conclusions

Two fundamental differences between fire and harvests emerge from our data as important linkages for overstory structure and understory composition. First, the *magnitude* of overstory removal is potentially much greater in commercially thinned stands than following prescribed fire. Although woodland ground flora are considerably more receptive to the light availability after thinning, seedling and sapling species rapidly intercept this light from desired herbs and graminoids, shading them out in only a few years. The second difference is the *immediacy* of mid- and overstory removal. While prescribed fire can instantly girdle small-diameter stems, an apparent lag time exists for the reduction of intermediate and large trees. Although this effect may have been exacerbated by drought during 2012 inventories, adjacent control plots had significantly greater stocking than burn-only plots. This subtle decrease in

overstory structure as opposed to immediate mechanical removal also resulted in far fewer basal sprouts and woody growth, ultimately sustaining desired ground cover for a longer interval. Albeit, seedling and sapling compositions in burn-only treatments were primarily shade-tolerant, but we expect that continual spring burning will eventually exhaust these species from the regeneration layer (Brose and Van Lear 1999b, Bowles et al. 2007, and others). Burning in harvested stands not only favored oak and hickory regeneration, but also sustained the greatest understory diversity and richness. The trade-off for these benefits is the continued growth of basal sprouts and r-strategists, which also necessitates additional fire if benefits are to be maintained.

V. SUMMARY & CONCLUSIONS

The purposes of this study were to 1) examine changes in overstory structure and woody composition 2) quantify tree mortality, growth, and damage, and 3) evaluate changes in the ground layer vegetation, all in response to silvicultural treatments used for woodland management. Analyses of thinning and burning techniques in this context are increasingly important for developing sound management regimes capable of sustaining oak and oak-pine community attributes. This investigation used measures of canopy cover, ground cover, species composition, scar dimensions, and annual diameter growth to formulate understanding of the successive dynamics in oak woodlands in the Midwest USA. Pre-treatment character of each fully stocked, second growth site represents forests and woodland condition across the Central Hardwoods Region and the Ozark Highlands Ecoregion.

Post-treatment analysis conducted immediately after the last prescribed confirmed the hypothesis that tree density and percent stocking would decrease in all treatments except control. Stocking decreases due to prescribed burning were marginally greater (3%) in un-thinned stands (burn-only) than in stands that were thinned from below prior to burning. This result is not surprising given that thinning occurred before any fire capable of eliminating fire-sensitive stems was introduced; however, fire intensity was much greater where slash and additional fuels remained after harvests. Therefore, although stocking reductions attributed to fire were greater in burn-only treatments, fire-induced mortality was significantly greater in harvest-burn plots as a function of increased fire severity. Even fire-tolerant size classes (intermediate and large trees) had nearly 3x greater mortality where harvests preceded burning.

Patterns in tree density and mortality also confirmed that smaller diameter stems are more susceptible to thinning and burning treatments than larger diameter stems. Again, harvesting before applying fire increased sapling mortality (54%) more than double the burn-only (26%). Although additional removal of this size class is often intended for woodland management, a 10-year evaluation shows that sapling size stems (primarily basal sprouts) are rapidly increasing where thinning removed over half the basal area, even after two fires. Sapling composition in harvest and harvest-burn treatments were dominated by oak and hickory, while burn-only and control treatments retained greater ratios of mesic oak-competitors such as blackgum, sassafras, and black cherry. Although there were not many small diameter pines present in any of the plots, this study did find that overstory pines were more resistant to fire than any other species, followed by species of the white oak group.

While radial growth of white oaks was not significantly altered by burning alone, significant reductions did occur between burned and unburned stands that were harvested. Although mortality of dominant and co-dominant trees was minimal, radial growth rates in harvest-burn treatments may signify environmental stresses that burning imposes on open-growing trees. This analysis did not find any correlation between fire-induced damage (i.e. scaring) and tree growth, however, did show that harvest-burn combinations led to significantly greater scar percentages than burning alone, especially in red oaks. Future research will focus on broader changes in productivity of managed woodlands.

Burn-only and harvest-only treatments led to nearly equivalent coverage responses of all understory life-forms. Although aspect had almost no statistical importance in the majority of this analysis, understory composition was particularly influenced by slope aspect. Graminoid,

shrub, and vine species were favored on drier south slopes, while forbs and trees were favored on shadier north slopes. Woodland indicator species were more responsive to commercial harvest treatments than to prescribed burning, however, coverages in harvest-only plots did not persist compared to burn-only plots due to the vigorous growth of woody sprouts and saplings. Lagged reductions in overstory stocking of burn-only plots strongly support use this management technique to subtly or slowly open the canopy as opposed to immediate thinnings in order to suppress seedling and sapling response. Shading by small diameter stems in the harvest-burn resulted in only 20.2% canopy openness above ground-layer vegetation. Albeit, harvest-burn treatments still resulted in the greatest overall canopy openness, understory richness and diversity, and coverage of desired physiognomic groups (forbs, graminoids, woodland indicators, legumes). While initial post-treatment analysis (ch. II) found significantly greater coverages of desired physiognomic groups in harvest-burn plots, a longer-term evaluation (ch. IV) could not conclude that treatments were significantly different. Therefore, this assessment should highlight the trade-off between seedling and sapling promotion via overstory removal > 50%, in return for statistically nominal, yet perhaps ecologically significant, increases in understory richness, diversity, and desired physiognomic groups.

VI. LITERATURE CITED

Abrams, M.D. 1992. Fire and the development of oak forests. Bioscience 42(5): 346–353.

Adams, A.S. and L.K. Rieske. 2001. Herbivory and fire influence white oak (*Quercus alba L.*) seedling vigor. Forest Science 47, 331–337.

Akaike, H. 1973. Information theory and an extension of the maximum likelihood principle. *In* Petrov, B. N. and F. Csaki. (Eds.). Proceedings of the Second International Symposium on Information Theory. Kademiai Kiado, Budapest. pp. 267-281.

Albrecht, M. A. and B.C. McCarthy. 2006. Effects of prescribed fire and thinning on tree recruitment patterns in central hardwood forests. Forest Ecology and Management 226:88–103.

Alexander, H.D., M.A. Arthur, D.L. Loftis, and S.R. Green. 2008. Survival and growth of upland oak and co-occurring competitor seedlings following single and repeated prescribed fires. Forest Ecology and Management 226, 88-103.

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

Atwood, E.L. and J.A. Steyermark. 1937. The white-tailed deer in Missouri. (Preliminary report). Clark National Forest, U.S. Department of Agriculture, Forest Service. 34 p.

Barnes, T.A. and D.H. Van Lear. 1998. Prescribed fire effects on hardwood advance regeneration in mixed hardwood stands. Southern Journal of Applied Forestry 22, 138-142.

Batek, M.J., A.J. Rebertus, W.A. Schroeder, T.L. Haithcoat, E. Compas, and R.P. Guyette. 1999. Reconstruction of early nineteenth century vegetation and fire regimes in the Missouri Ozarks. Journal of Biogeography 26, 397-412.

Beers, T.W., P.E. Dress and L.C. Wensel. 1966. Aspect transformation in site productivity research. Journal of Forestry 64, 691-692.

Bielmann, A.P. and L.G. Brenner. 1951. The recent intrusion of forests in the Ozarks. Annals of the Missouri Botanical Gardens 38, 261-282.

Black, B. A., J.J. Colbert, and N. Pedersen. 2008. Relationships between radial growth rates and lifespan within North American tree species. Ecoscience 15, 349–357.

Blake, J.G. and B. Schuette. 2000. Restoration of an oak forest in east-central Missouri: early effects of prescribed burning on woody vegetation. Forest Ecology and Management 139, 109-126.

Blizzard, E.M., J.M. Kabrick, D.C. Dey, D.R Larsen, S.G. Pallardy, and D.P. Gwaze. 2013. Light, canopy closure and overstory retention in upland Ozark forests. *In* Guldin, J.M., ed. Proceedings of the 15th biennial southern silvicultural research conference; 2008 November 17-20; Hot Springs, AR. Gen. Tech. Rep. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. pp. 73-79.

Block, W.M. and M.L. Morrison. 1991. Influence of scale on the management of wildlife in California oak woodlands. *In* Standiford, R.B. (Tech. Coord). Proceedings of a symposium on oak woodlands and hardwood rangeland management, Davis, CA. October 31-November 2, 1990. USDA Forest Service, Pacific Southwest Research Station, Gen Tech. Rep. PSW-126. pp. 96-104.

Boerner, R. E. J., T. R. Lord, and J. C. Peterson. 1988. Prescribed burning in the oak-pine forest of the New Jersey Pine Barrens: effects on growth and nutrient dynamics of two *Quercus* species. American Midland Naturalist 120, 108-119.

Boerner, R. E. J. and J. A. Brinkman. 2003. Fire frequency and soil enzyme activity in southern Ohio oak–hickory forests. Applied Soil Ecology 23, 137–146.

Borchert, J. R. 1950. The climate of the central North American grassland. Annals of the Association of American Geographers 40, 1-39.

Bowles, M.L., K.A. Jacobs, and J.L. Mengler. 2007. Long-term changes in an oak forest's woody understory and herb layer with repeated burning. Journal of Torrey Botanical Society 134, 223-237.

Boyles, J.G. and D.P. Aubrey. 2006. Managing forests with prescribed fire: implications for a cavity-dwelling bat species. Forest Ecology and Management 222, 108-115.

Brose, P.H. and D.H. Van Lear. 1999a. Effects of seasonal prescribed fires on residual overstory trees in oak-dominated shelterwood stands. Southern Journal of Applied Forestry 23(2), 88-93.

Brose, P. H., D.H. Van Lear, and R. Cooper. 1999b. Using shelterwood harvests and prescribed fire to regenerate oak stands on productive upland sites. Forest Ecology and Management. 113,125-14.

Brudvig, L.A. and H. Asbjornsen. 2007. Stand structure, composition, and regeneration dynamics following removal of encroaching woody vegetation from Midwestern oak savannas. Forest Ecology and Management 244, 112–121.

Burns, P.Y. 1955. Fire scars and decay in Missouri oaks. Bulletin 642. Columbia, MO:University of Missouri, College of Agriculture, Agricultural Experiment Station: 8 p.

Burns, P.Y., D.M. Christisen, and J.M. Nichols. 1954. Acorn production in the Missouri Ozarks. Bulletin 611. University of Missouri, College of Agriculture Experiment Station: 8 p.

Byram, G. M. 1959. Combustion of forest fuels. *In* Davis, K.P. (Ed.). Forest Fire Control and Use. McGraw Hill, New York. pp. 61-89.

Callahan, T.R. 1996. Avian community structure within restored oak-savanna and burned oak-woodland in Missouri. M.S. thesis, University of Missouri, Columbia. 124 p.

Christisen, D. M. and L. Korschgen. 1955. Acorn yields and wildlife management in Missouri. Transactions of North American Wildlife Conference 20, 337-357.

Colbert, J. J., M. Schuckers, D. Fekedulegn, J. Rentch, M. MacSiurtain, and K. Gottschalk. 2004. Individual tree basal-area growth parameter estimates for four models. Ecological Applications 174, 115–126.

Cook, E.R. and L.A. Kairiukstis (Eds.). 1990. Methods of Dendrochronology: Kluwer Academic Publishers, Dordrecht. 393 p.

Cunningham, R.J. 2007. Historical and social factors affecting pine management in the Ozarks during the late 1800's through 1940. *In* Kabrick, J.M., D.C. Dey and D. Gwaze (Eds.). Shortleaf pine restoration and ecology in the Ozarks: Proceedings of a symposium. November 7-9, 2006, Springfield, MO. USDA Forest Service, Northern Research Station, Gen. Tech. Rep. NRS-P-Volume 15.

Cutter, B. E. and R. P. Guyette. 1994. Fire frequency on an oak–hickory ridgetop in the Missouri Ozarks. American Midland Naturalist 132, 393–398.

Delcourt, P.A., G.R. Wilkins, and E.N. Smith. 1986.Vegetational history of the cedar glades regions of Tennessee, Kentucky and Missouri during the past 30,000 years. A.S.B. Bulletin 33 (4), 128-137.

Dey, D.C. and Z. Fan. 2009. A review of fire and oak regeneration and overstory recruitment. *In* Hutchinson, T.F. (Ed.). Proceedings of the 3rd Fire in Eastern Oak Forests Conference. USDA Forest Service, Northern Research Station, Gen. Tech. Rep. NRS-P-46. Newtown Square, PA, pp. 2-20.

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

Dey, D.C. and R.G. Jensen. 2002. Stump sprouting potential of oaks in Missouri Ozark forests managed by even- and uneven-aged silviculture. *In* Shifley, S.R. and J.M. Kabrick. (Eds.).

USDA Forest Service, North Central Research Station, Gen. Tech. Rep NC-227. St. Paul, MN. pp. 102-113.

Engber, E., J.M.Varner, L.A. Arguello, and N.G. Sugihara. 2011. The effects of conifer encroachment and overstory structure on fuels and fire in an oak woodland landscape. Fire Ecology 7(2), 32-50.

Ericksen, B. 1997. Small mammal response to restoration in two Missouri savannas. M.S.thesis, Central Missouri State University, Warrensburg. MO.

Evans, K.E. and R.A. Kirkman. 1981. Guide to bird-habitats of the Ozark Plateau. Gen. Tech. Rep. NC-68. USDA Forest Service, North Central Forest Experiment Station, St. Paul, MN.

Faber-Langednoen, D. (Ed.). 1999. International classification of ecological communities; terrestrial vegetation of the Midwestern U.S. The Nature Conservancy, Midwest Conservation Science Department, Minneapolis, MN.

Fralish, J. S., F. B. Crooks, J. L. Chambers, and F. M. Harty. 1991. Comparison of presettlement, second-growth and old-growth forest on six site types in the Illinois Shawnee Hills. American Midland Naturalist 125, 294-309.

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

Frazer, G.W., C.D. Canham, and K.P Lertzman.1999. Gap Light Analyzer GLA, Version 2.0: Imaging software to extract canopy structure and gap light transmission indices from true-colour fisheye photographs, users manual and program documentation. Copyright © 1999. Simon Fraser University, Burnaby, British Columbia, and the Institute of Ecosystem Studies, Millbrook, New York. 36 p.

Fule´, P.Z., D.C. Laughlin, and W.W. Covington. 2005. Pine-oak forest dynamics five years after ecological restoration treatments, Arizona, USA. Forest Ecology and Management 218, 129–145.

Gaskins, M., 2005. Impacts of Fire on Wildlife. 2005. *In* Groninger, J. W., L.A. Horner, J.L. Nelson and C.M. Ruffner (Eds.). Prescribed Fire and Oak Ecosystem Maintenance: A Primer for Land Managers. Southern Illinois University Carbondale, Department of Forestry.

Gingrich, S. F. 1967. Measuring and evaluating stocking and stand density in upland hardwood forests in the Central States. Forest Science 13, 38–53.

Gingrich, S. F. 1971. Management of young and intermediate stands of upland hardwoods. Research Paper NE-195. USDA Forest Service, Northeastern Forest Experiment Station, Broomall, PA. 26 p.

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

Godsey, K.W. 1988. Effect of fire on an oak-hickory forest. Columbia, MO: University of Missouri. MS Thesis. 125 p.

Grabner, J.K., D.R. Larsen, and J.M. Kabrick. 1997. An analysis of MOFEP ground flora: pre-treatment conditions. *In* Brookshire, B.L. and S.R. Shifley. (Eds.). Proceedings of the Missouri Ozark Forest Ecosystem Project Symposium: an experimental approach to landscape research; 1997 June 3-5; St. Louis, MO. Gen. Tech. Rep. NC-193. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. pp. 169-197.

Grabner, K.W., G. Hartman, J. Dwyer, and B.E. Cutter. 1999. Characterizing fuel loading and structure using ecological landscapes (ELT's) in the Missouri Ozarks. *In* The Joint Fire Science Conference and Workshop. University of Idaho, The Grove Hotel, Boise, ID. pp. 278-282

Grundel, R. and N. Pavlovic. 2007. Distinctiveness, use, and value of Midwestern oak savannas and woodlands as avian habitats. The Auk 124, 969-985.

Guyette, R.P. 1999. Litter accumulation and fire intervals in the Ozark forest, a report prepared for the Missouri Department of Conservation MOFEP Project. The School of Natural Resources, University of Missouri - Columbia. 9 p.

Guyette, R.P. and B.E. Cutter. 1991. Tree-ring analysis of fire history of a post oak savanna in the Missouri Ozarks. Natural Areas Journal 11(2), 93-99.

Guyette, R.P. and B.E. Cutter. 1997. Fire history, population, and calcium cycling in the Current River watershed. *In* Pallardy, S.G., R.A. Cecich, H.G. Garrett and P.S. Johnson. (Eds.). Proceedings of the 11th Central Hardwood Forest Conference. USDA Forest Service, GTR-NE-188, Columbia, MO. pp. 354-372.

Guyette, R.P. and D.C. Dey. 1997. Historic shortleaf pine (Pinus echinata Mill.) abundance and fire frequency in a mixed oak - pine forest (MOFEP, Site 8). *In* Brookshire, B.L. and S.R. Shifley. (Eds.). Proceedings of the Missouri Ozark Forest Ecosystem Project Symposium: an experimental approach to landscape research; 1997 June 3-5; St. Louis, MO. Gen. Tech. Rep. NC-193. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. pp. 136-149

Guyette, R.P., M. Dey and D. Dey. 1999. An Ozark fire history. Missouri Conservationist 60(3), 4-7.

Guyette, R.P and J.M. Kabrick. 2002. The legacy and continuity of forest disturbance, succession, and species at the MOFEP sites. *In* Shifley, S. R. and J.M. Kabrick. (Eds.). Proceedings of the Second Missouri Ozark Forest Ecosystem Project Symposium: Post-treatment Results of the Landscape Experiment. Gen. Tech. Rep. NC-227. St. Paul, MN: U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station. pp. 26-44.

Guyette, R. and D. Larsen. 2000. A history of anthropogenic and natural disturbances in the area of the Missouri Ozark Forest Ecosystem Project. *In* Shifley, S.R. and B.L. Brookshire. (Eds.). Missouri Ozark Forest Ecosystem Project: site history, soils, landforms, woody and herbaceous vegetation, down wood, and inventory methods for the landscape experiment. Gen. Tech. Rep. NC-208. St. Paul, MN: U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station. pp. 19-40.

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

Guyette, R.P. and M.C. Stambaugh. 2004. Post-oak fire scars as a function of diameter, growth, and tree age. Forest Ecology and Management 198, 183-192.

Hanberry B.B., J. Yang, J.M. Kabrick, and H.S. He. 2012. Adjusting forest density estimates for surveyor bias in historical tree surveys. American Midland Naturalist 167, 285-306.

Hansen, C. L. 2011. The effects of silvicultural treatments on oak height and basal diameter growth and oak regeneration abundance following a woody biomass removal during harvest in the Missouri Ozarks. M.S. Thesis. University of Missouri- Columbia. 92 p.

Hare, R.C. 1965. Contribution of bark to fire resistance of southern trees. Journal of Forestry 63, 248-251.

Harmon, M.E. 1984. Survival of trees after low-intensity surface fires in Great Smoky Mountains National Park. Ecology 65, 796–802.

Harrison, W., T. Burk, and D. Beck. 1986. Individual tree basal area increment and total height equations for Appalachian mixed hardwoods after thinning. Southern Journal of Applied Forestry 10, 99–104.

Hartman, G.W. and B. Heumann. 2003. Prescribed fire effects in the Ozarks of Missouri: the Chilton Creek Project 1996-2001. In Proceedings of the second international wildland fire ecology and fire management congress. November 16-22, 2003. Orlando,FL.

Hengst, G.E. and J.O. Dawson. 1993. Bark and thermal properties of selected hardwood species. In Proceedings of the 9th Central Hardwoods Conference. U.S. Dept. of Agriculture, Gen. Tech. Rep. NC-161. pp. 55-75.

Houck, L. 1908. A history of Missouri, from the earliest explorations and settlements until the admission of the state into the union. 3 volumes. Chicago, IL. R.R. Donelly & Sons Co.

Howe, H.F. 1994. Response of early- and late-flowering plants to fire season in experimental prairies. Ecological Applications 4, 121-133.

Howell, D. L. and C. L. Kucera. 1956. Composition of pre-settlement forests in three counties of Missouri. Bulletin of the Torrey Botanical Club 83, 207-217.

Huddle, J.A. and S.G. Pallardy. 1996. Effects of long-term annual and periodic burning on tree survival and growth in a Missouri Ozark oak-hickory forest. Forest Ecology and Management 82, 1-9.

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

Hutchinson, T. F., D.A. Yaussy, R.P. Long, J. Rebbeck, and E.K. Sutherland. 2012. Long-term (13-year) effects of repeated prescribed fires on stand structure and tree regeneration in mixed-oak forests. Forest Ecology and Management 286, 87-100.

Iverson, L.R. and T.F. Hutchinson. 2002. Soil temperature and moisture fluctuations during and after prescribed fire in mixed-oak forests, USA. Natural Areas Journal 22, 296–304.

Jemison, G.M., 1943. Effect of Single Fires on the Diameter Growth of Shortleaf Pine in the Southern Appalachians. Journal of Forestry 41, 574-576.

Jenkins, S.E., R.P. Guyette, and A.J. Rebertus. 1997. Vegetation-site relationships and fire history of a savanna-glade-woodland mosaic in the Ozarks. *In* Pallardy, S.G., R.A. Cecich, H.G. Garret, and P.S. Johnson (Eds.). Proceedings: 11th Central Hardwood forest conference; 1997 March 23-26; Columbia, MO. Gen Tech. Rep. NC-188. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. pp.184-201.

Johnson, P.S., S.R. Shifley, and R. Rogers. 2009. The ecology and silviculture of oaks. CABI Publishing, New York, 580 p.

Johnson, S. E. and M.D. Abrams. 2009. Age class, longevity and growth rate relationships: protracted growth increases in old trees in the eastern United States. Tree Physiology 29, 1317–1328.

Kabrick, J. M., R.G. Jensen, S.R. Shifley, and D.R. Larsen. 2002. Woody vegetation following even-aged, uneven-aged, and no-harvest treatments on the Missouri Ozark Forest Ecosystem Project Sites. *In* Shifley, S. R. and J.M. Kabrick. (Eds.). Proceedings of the Second Missouri Ozark Forest Ecosystem Project Symposium: Post-treatment Results of the Landscape Experiment. USDA Forest Service, North Central Forest Experiment Station, Gen. Tech. Rep. NC-227. St. Paul, MN. pp. 84-101.

Kabrick, J.M., S.R. Shifley, R.G. Jensen, Z. Fan, and D.R. Larsen. 2004. Factors associated with oak mortality in Missouri Ozark Forests. *In* Yaussy, D.A., D.M. Hix, R.P. Long, and P.C. Goebel (Eds.). Proceedings of the 14th Central Hardwood Forest Conference, Wooster, Ohio. March 16-19, 2004. USDA Forest Service, Gen. Tech. Rep. GTR-NE-316. pp. 27–35.

Kabrick, J.M., E.K. Zenner, D.C. Dey, D. Gwaze, and R.G. Jensen. 2008. Using ecological land types to examine landscape-scale oak regeneration dynamics. Forest Ecology and Management 255, 3051–3062.

Kinkead, C.O., J.M. Kabrick, M.C. Stambaugh, and K.W. Grabner. in press. Changes to Oak woodland stand structure and ground flora composition caused by thinning and burning. *In* Gottschalk, K., and J. Brooks. (Eds.). Proceedings of the 18th Central hardwoods Conference. USDA For. Serv. North Central Research Station. Gen. Tech. Rep. xx-xxx.

Kitzberger, T., E. Aráoz, J. Gowda, M. Mermoz, and J. Morales. 2012. Decreases in fire spread probability with forest age promotes alternative community states, reduced resilience to climate variability and large fire regime shifts. Ecosystems 15, 97–112.

Knapp, E.E., D.W. Schwilk, J.M. Kane, and J.E. Keeley. 2007. Role of burning season on initial understory response to prescribed fire in a mixed conifer forest. Canadian Journal of Forest Research 37, 11-22.

Kolaks, J.J., B.E. Cutter, E.F. Loewenstein, K.W. Grabner, G.W. Hartman, and J.M. Kabrick. 2004. The effect of thinning and prescribed fire on fuel loading in the Central Hardwood Region of Missouri. *In* Yaussy, D.A., D.M. Hix, R.P. Long and P.C. Goebel (Eds.). Proceedings of the 14th Central Hardwood Forest Conference, Wooster, Ohio. March 16-19, 2004. USDA Forest Service, Gen. Tech. Rep. GTR-NE-316. pp. 168-177.

Ko, L.J. and P.B. Reich. 1993. Oak effects on soil and herbaceous vegetation in savannas and pastures in Wisconsin. American Midland Naturalist 130, 31-42.

Kruger, E.L and P.B. Reich. 1997. Responses of hardwood regeneration to fire in mesic forest openings I. post-fire community dynamics. Canadian Journal of Forest Research 27, 1822-1831.

Ladd, D. 1991. Re-examination of the role of fire in Missouri oak woodlands. *In* Burger, G.V., J.E. Ebinger and G.S. Wilhelm. (Eds.). Proceedings: Oak Woods Management Workshop. Eastern Illinois University. Charelston, IL. pp. 67-80.

Larsen, D.R., M.A. Metzger, and P.S. Johnson. 1997. Oak regeneration and overstory density in the Missouri Ozarks. Canadian Journal of Forest Research 27, 869–875.

Larsen, D.R., E.F. Loewenstein, and P.S. Johnson. 1999. Sustaining recruitment of oak reproduction in uneven-aged stands in the Ozark highlands. U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. Gen. Tech. Rep. NRS-P-203.

Loomis, R.M. 1973. Estimating fire-caused mortality and injury in oak-hickory forests. Res. Pap. NC-94. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 6 p.

Loomis, R.M. 1974. Predicting the losses in sawtimber volume and quality from fires in oak-hickory forests. Res. Pap. NC-104. St. Paul, MN. U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station.

Lorimer, C.G. and L.E. Frelich. 1989. A methodology for estimating canopy disturbance frequency and intensity in dense temperate forests. Canadian Journal of Forest Research 19, 651–663.

Loewenstein, E.F., P.S. Johnson, and H.E. Garrett. 2000. Age and diameter structure of a managed uneven-aged forest. Canadian Journal of Forest Research 28, 1060-1070.

Lowery, R.F. 1968. The effect of initial stem temperature on time required to kill cambium of shortleaf pine. M.S. Thesis, University of Missouri, Columbia, MO. 52 p.

Mann, C. S. and A.R. Forbes. 2007. Wildlife diversity of restored shortleaf pine-oak woodlands in the northern ozarks. *In* Shortleaf pine restoration and ecology in the Ozarks: proceedings of a symposium, Springfield, MO. November 7-9, 2006. USDA Forest Service, Northern Research Station. 167 p.

Mark Twain National Forest. 2011. Proposal for Collaborative Forest Landscape Restoration Program. Missouri Pine-Oak Woodlands Restoration Project. 32 p.

Marschall, J. M. 2013. Timber product value loss due to prescribed fire caused injuries in red oak trees. M.S. Thesis. University of Missouri- Columbia. 60 p.

Martin, A.C., H.S. Zim, and A.L. Nelson. 1951. American wildlife and plants: a guide to wildlife food habits. New York: Dover Publications, Inc. 500 p.

Master, R.E. 2007. The importance of shortleaf pine for wildlife and diversity in mixed oak-pine forests and in pine-grassland woodlands. *In* Kabrick, J.M., D.C. Dey, and D. Gwaze. (Eds.). Shortleaf pine restoration and ecology in the Ozarks: Proceedings of a symposium, 7-9 November 2006, Springfield, MO, USA. US Department of Agriculture, Forest Service, Gen. Tech. Rep. NRS-P-15, Newtown Square, PA, USA.

McMurry, E.R., R. Muzika, E.F. Lowenstein, K.W. Grabner, and G.W. Hartman. 2007.Initial effects of prescribed burning and thinning on plant communities in the southeast Missouri Ozarks. *In* Buckley, D.S. and W.L. Clatterbuck. (Eds.). Proceedings: 15th Central Hardwood forest conference; 2006 February 27 – March 1; Knoxville, TN. Gen. Tech. Rep. SRS-101. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: pp. 241-249.

Meinert, D., T. Nigh, and J. Kabrick. 1997. Landforms, geology, and soils of the MOFEP study area. *In* Proceedings of the Missouri Forest Ecosystem Project Symposium: an experimental approach to landscape research. Brookshire, B.L. and S.R. Shifley (Eds.). USDA Forest Service, North Central Research Station, GTR-NC-193, St. Louis, MO.

Missouri's Comprehensive Wildlife Conservation Strategy. 2005. Conserving all wildlife in Missouri: A directory of conservation opportunities. The Conservation Commission of the State of Missouri.

Missouri Natural Resources Conservation Service. 2004. Prescribed Burning. Conservation Practice Information Sheet: IS-MO338. 8 p.

Mottl, L. M., 2000. Woodland herbaceous layer restoration in Iowa. M.S. thesis. Iowa State University, Ames.

Mottl, L.M., C.M. Mabry, and D.R. Farrar. 2006. Seven-year survival of perennial herbaceous transplants in temperate woodland restoration. Restoration Ecology 14 (3), 330-338.

Mount, J. and L. Horner. 2005. Silvicultural alternatives to Fire. *In* Groninger, J.W., L.A. Horner, J.L Nelson, and C.M. Ruffner (Eds.). Prescribed Fire in Oak Ecosystem Maintenance: A primer for land managers. Department of Forestry, Southern Illinois University- Carbondale.

National Climatic Data Center and National Oceanic and Atmospheric Administration (NCDC/NOAA). 2012. Selection criteria for displaying period of record. Available at <http://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp?divId=2305>.

Nelson, J.C. 1997. Presettlement vegetation patterns along the 5th principal meridian, Missouri Territory, 1815. American Midland Naturalist 137(1), 79-94.

Nelson, P.W. 2004. Classification and characterization of savannas and woodlands in Missouri. *In* Hartman, G., S. Holst, and B. Palmer. (Eds.). Proceedings of SRM 2002: Savanna/Woodland Symposium. Missouri Department of Conservation Press, Jefferson City, MO. pp. 9-25.

Nelson, P.W. 2005. The Terrestrial Natural Communities of Missouri. Jefferson City, MO: Missouri Natural Areas Committee. 550 p.

Nelson, P.W. 2011. Fire-adapted natural communities of the Ozark highlands at the time of European settlement and now. *In* Dey, D.C., M.C. Stambaugh, S.L. Clark and C.J. Schweitzer

(Eds.). Proceedings of the 4th fire in eastern oak forests conference. USDA Forest Service, Northern Research Station, Gen. Tech. Rep. NRS-P-102. pp.92-102

Nigh, T.A. 1992. The forests prior to European settlement. *In* Journet, A.R.P. and H. G. Spratt (Eds). Proceedings: Towards a vision for Missouri's public forest. Cape Girardeau, MO. Southeast Missouri State University. pp. 6-13.

Nigh, T.A. 2004. Missouri's forests: an ecological perspective. *In* S.L. Flader (Ed.). Proceedings: Toward sustainability of Missouri forests; 1999 March 4-5; Columbia, MO. Gen. Tech. Rep. NC-239. St. Paul MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. pp. 6-19.

Nigh, T.A., S.G. Pallardy, and H.E. Garrett. 1985. Sugar maple environment relationships in the River Hills and central Ozark Mountains of Missouri. American Midland Naturalist 114, 235-251.

Nigh, T.A. and W.A. Schroeder. 2002. Atlas of Missouri Ecoregions. Jefferson City, MO. Missouri Department of Conservation. 212 p.

Nowacki, G.J. and M.D. Abrams. 1997. Radial-growth averaging criteria for reconstructing disturbance histories from presettlement-origin oaks. Ecological Monographs 67, 225–249.

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

Natural Resource Conservation Service (NRCS). 2008. Open woodland information sheet: conservation practice information sheet (IS-MO643w). Available online at https://www.forestandwoodland.org/pdfs/woodland%20information%20sheet.pdf.

Nuzzo, V.A. 1986. Extent and status of Midwest oak savanna: pre-settlement and 1985. Natural Areas Journal 6(2), 6-36.

Packard, S. 1993. Restoring oak ecosystems. Restoration and Management Notes 11, 5-16.

Pallardy, S.G., T.A. Nigh, and H.E. Garrett. 1988. Changes in forest composition in central Missouri: 1968-1982. American Midland Naturalist 120, 380-390.

Pallardy, S.G., T.A. Nigh, and H.E. Garrett. 1991. Sugar maple invasion in oak forests of Missouri. In G.V. Burger, J.E. Ebinger and G.S. Wilhelm, (Eds.). Proceedings: Oak Woods Management Workshop. Eastern Illinois University. Charelston, IL. pp. 21-30.

Paulsell, L.K. 1957. Effects of burning on Ozark hardwood timberlands. University of Missouri College of Agriculture, Agricultural Experiment Station. Research Bulletin 640. 24 p.

Parker, V.T. and C.H. Muller. 1982. Vegetational and environmental changes beneath isolated live oak trees (Quercus agrifolia) in a California annual grassland. American Midland Naturalist 107, 69-81.

Peterson, D.W and P.B. Reich. 2001. Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. Journal of Applied Ecology 11, 914–92.

Peterson, D.L., S.S. Sackett, L.J. Robinson, and S.M. Haase. 1994. The effects of repeated prescribed burning on Pinus ponderosa growth. International Journal of Wildland Fire 4, 239-247.

Phillips, R., T. Hutchinson, L. Brudnak, and T. Waldrop. 2007. Fire and fire surrogate treatments in mixed-oak forests: effects on herbaceous layer vegetation. In Butler, B.W. and W. Cook. (Eds.). Proceedings: The fire environment—innovations, management, and policy. March 26-30, 2007; Destin, FL. RMRS-P-46CD. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. pp. 475-485.

Ponder, F., M. Tadros, and E.F. Loewenstein. 2009. Microbial properties and litter on soil nutrients after two prescribed fires in developing savannas in an upland Missouri Ozark forest. Forest Ecology and Management 257, 755-763.

Pyne, S.J., P.L. Andrews, and R.D. Laven. 1996. Introduction to wildland fire. New York, NY: John Wiley & Sons. 769 p.

Reich, P.B., D.W. Peterson, D.A. Wedin and K. Wrage. 2001. Fire and vegetation effects on productivity and nitrogen cycling across a forest-grassland continuum. Ecology 82, 1703-1719.

Rentch, J. S., F. Desta, and G.W. Miller. 2002. Climate, canopy disturbance, and radial growth averaging in a second-growth mixed-oak forest in West Virginia, USA. Canadian Journal of Forest Research 32, 915–27.

Rubino, D.L. and B.C. McCarthy. 2004. Comparative analysis of dendroecological methods used to assess disturbance events. Dendrochronologia 21, 97–115.

SAS Institute Inc. 2012. What's New in SAS ® 9.3. Cary, NC: SAS Institute Inc., 2012.

Society of American Foresters (SAF). 2008. The Dictionary of Forestry. Available online at http://www.dictionaryofforestry.org.

Scharenbroch, B.C., B. Nix, K.A. Jacobs, and M.L. Bowles 2012. Two decades of low-severity Prescribed fire increases soil nutrient availability in a Midwestern, USA oak (Quercus) forest. Geoderma 183-184, 80-91.

Schlesinger, R.C. 1978. Increased growth of released white oak poles continues through two decades. Journal of Forestry 76, 726–727.

Schoolcraft, H.R. 1821. Journal of a tour into the interior of Missouri and Arkansas in 1818 and 1819. London. Reprinted in 1955 by Press-Argus Printers, Van Buren, AK.

Schroeder, W. A. 1981. Presettlement prairie of Missouri. Missouri Department of. Conservation. Jefferson City. Natural History Series No. 2. 40 p.

Schroeppel, B. 2004. Federal and state programs to offset costs of savanna restoration. *In* Proceedings of SRM 2002: Savanna/Woodland Symposium, ed. G. Hartman, S. Holst, and B. Palmer, 9-25. Missouri Department of Conservation Press, Jefferson City, MO.

Shannon, C.E. and W. Weaver. 1949. The mathematical theory of communication. Illinois: Universityof Illinois Press.

Shifley, S.R., F.R. Thompson III, W.D. Dijak, M.A. Larson, and J.J. Millspaugh. 2006. Simulated effects of forest management alternatives on landscape structure and habitat suitability in the Midwestern United States. Forest Ecology and management 229, 361-377.

Scowcroft, P.G. 1965. The effects of fire on the hardwood forest of the Missouri Ozarks. Masters Thesis. University of Missouri.126 p.

Shigo, A. L. 1984. Compartmentalization: a conceptual framework for understanding how trees grow and defend themselves. Annual Review of Phytopathology 22, 189-214.

Simard, A. J., J.E. Eenigenburg, K.B. Adams, J. Nissen, L. Roger, and A. G. Deacon. 1984. A general proceedure for sampling and analyzing wildland fire spread. Forest Science. 30, 51-64.

Simpson, E. H. 1949. Measurement of diversity. Nature, 163, 688.

Smith, K.T. and E.K. Sutherland. 2001. Terminology and biology of fire scars in selected central hardwoods. Tree-Ring Research 57(2), 141-147.

Smith, K.T. and E.K. Sutherland. 2006. Resistance of eastern hardwood stems to fire injury and damage. *In* Dickinson, M.B. (Ed.). Proceedings: Fire in eastern oak forests: delivering science to land managers; 2005 November 15-17; Columbus, OH. Gen. Tech. Rep. NRS-P-1. Newton Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: pp. 210-217.

Sparks, J.C., R.E. Masters, D.M. Engle, M.W. Palmer, and G.A. Bukenhofer. 1998. Effects of late growing-season and late dormant-season prescribed fire on herbaceous vegetation in restored pine-grassland communities. Journal of Vegetation Science 9: 133-142.

Sparks, J.C., M.C. Stambaugh, and E.L. Keith. 2012. Restoring fire suppressed Texas oak woodlands to historic conditions using prescribed fire. *In* Dey, D.C., M.C. Stambaugh, S.L. Clark and C.J. Schweitzer. (Eds.). Proceedings of the 4th fire in eastern oak forests conference. USDA Forest Service, Northern Research Station, Gen. Tech. Rep. NRS-P-102. pp.127-141.

Stambaugh, M.C. and R.P. Guyette. 2006. Fire regime of an Ozark wilderness area, Arkansas. American Midland Naturalist 156, 237-251.

Stambaugh, M.C., R.P. Guyette, K. Grabner, and J. Kolaks. 2006. Understanding Ozark forest litter variability through a synthesis of accumulation rates and fire events. *In* Butler, B.W., Andrews, P.L., (Comps.), Fuels management- how to measure success: Conference Proceedings. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-41.

Stambaugh, M.C., D.C. Dey, R.P. Guyette, H. He, and J.M. Marschall. 2011. Spatial and temporal patterning of fuel loading and fire hazard across a deciduous forest landscape, USA. Landscape Ecology 26(7), 923-935.

Stambaugh, M.C. and R.P. Guyette. 2006. Fire regime of an Ozark wilderness area, Arkansas. American Midland Naturalist 156, 237–251.

Stambaugh, M.C., R.P Guyette, and D.C. Dey. 2007. What fire frequency is appropriate for shortleaf pine regeneration and survival? *In* Kabrick, J.M., Dey, D.C., Gwaze, D. (Eds.). Shortleaf Pine Restoration and Ecology in the Ozarks: Proceedings of a Symposium, Gen. Tech. Rep. NRS-P-15. Newtown, PA.

Stambaugh, M.C., R.M. Muzika, and R.P. Guyette. 2002. Disturbance characteristics and overstory composition of an oldgrowth shortleaf pine (*Pinus echinata* Mill.) forest in the Ozark Highlands, Missouri. U.S.A. Natural Areas Journal 22, 108–119.

Stan, A., B. Rigg, and L.S. Jones. 2006. Dynamics of a managed oak woodland in northeastern Illinois. Natural Areas Journal 26, 187–197.

Stephens, S.L. and M.A. Finney. 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. Forest Ecology and Management 162, 261–271.

Steyermark, J.A. 1959. Vegetational history of the Ozark forest. Univeristy of Missouri Studies no. 31, University of Missouri, Columbia.

Sauer, C.O. 1920. The Geography of the Ozark Highland of Missouri. Geographical Society. Bulletin 7. Chicago, Ill.

Stevenson, A. P., R.M. Muzika, and R.P. Guyette. 2008. Fire scars and tree vigor following prescribed fires in Missouri Ozark upland forests. *In* Jacobs, D. F. and C.H. Michler. (Eds.), Proceedings of 16th central hardwood forest conference. USDA Forest Service, Northern Research Station, Gen. Tech. Rep. NRS-P-24. pp. 525-534.

Taft, J.B. 1997. Savanna and open-woodland communities. In Schwartz, M.W. (Ed.). Conservation in highly fragmented landscapes. Chapman and Hall, New York. pp. 24-54.

Taft, J. B. 2009. Effects of overstory stand density and fire on ground layer vegetation in oak woodland and savanna habitats. *In* Hutchinson, T. F. (Ed.). Proceedings of the 3rd fire in eastern oak forests conference; 2008 May 20-22; Carbondale, IL. Gen. Tech. Rep. NRS-P-46. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: pp. 21-39.

Thompson, F.R. III, S.K. Robinson, D.L. Whitehead and J.D. Brawn.1996. Management of central hardwood landscapes for the conservation of migratory birds. *In* Thompson, F.R. III (Ed.). Management of midwestem landscapes for the conservation of neotropical migratory birds, 1995. USDA Forest Service, North Central Forest Experiment Station, Gen. Tech. Rep. NC-187, St Paul, MN. 207 p.

Varner, J.M., D.R. Gordon, F.E. Putz, and J.K. Hiers. 2005. Restoring fire to long-unburned Pinus Palustris ecosystems: novel fire effects and consequences for long-unburned ecosystems. Restoration Ecology 13, 536-544.

Visser, H., 1995. Note on the relation between ring widths and basal area increments. Forest Science 41, 297–304.

Voelker, S.L., R.M. Muzika, and R.P. Guyette. 2008. Individual tree and stand level influences on the growth, vigor, and decline of Red Oaks in the Ozarks. Forest Science 54, 8–20.

Vose, J.M. 2000. Perspectives on using prescribed fire to achieve desired ecosystem conditions. Pages 12-17. *In* Moser, W.K and C.F. Moser. (Eds.). Fire and forest ecology: innovative silviculture and vegetation management. Tall Timbers Fire Ecology Conference Proceedings, No. 21. Tall Timbers Research Station, Tallahassee, FL.

Westin, S. 1992. Wildfire in Missouri. Missouri Department of Conservation. 161 p.

Wyckoff, P.H. and J.S. Clark. 2005. Tree growth prediction using size and exposed crown area. Canadian Journal of Forest Research 35, 13–20.

Yates, M.D and R.M. Muzika. 2006. Effect of forest structure and fragmentation on site occupancy of bat species in Missouri Ozark forests. Journal of Wildlife Management 70, 1238-1248.