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Proceedings of the 4th Fire in Eastern Oak Forests Conference

Abstract

Contains 14 full-length papers and 40 abstracts of posters that were presented at the 4th Fire in Eastern Oak Forests conference, held in Springfield, MO, May 17-19, 2011. The conference was attended by over 250 people from 65 different organizations and entities, representing 22 states and 1 Canadian province.

Acknowledgments

The steering committee thanks the Missouri University Conference Office and in particular Angela Freemyer, Conference Coordinator, Lindsay Kilgore, Administrative Assistant, and Sharon Rodes, Graphic Designer. We also extend our thanks to the management and staff of the University Plaza Hotel for our conference and hotel accommodations. Special thanks go to Nancy Bastin, Southern Research Station, for formatting all papers and abstracts in this proceedings. Special thanks also go to Reggie Bray, Jody Eberly, Paul Nelson, and other staff of the U.S. Forest Service Mark Twain National Forests for their leadership and contributions in the Glade Top Trail and shortleaf pine restoration field trips, to Ken McCarty, Allison Vaughn, and Tim Smith of the Missouri Department of Natural Resources, State Parks for their contributions to the field trip at Roaring River State Park, and to Sherry Leis, Missouri State University Biology Department for leading a tour on ecosystem restoration at the National Park Service Wilson's Creek National Battlefield Park.

This conference was dedicated to the memory of Alan Zentz (1953-2010), who was a long-standing supporter of this gathering.

> Alan joined the U.S. Forest Service in 1999 as the Cooperative Fire Program Specialist for the Northeastern Area. Throughout his career he worked to improve forest fire protection for millions of people in the Northeast. He provided response assistance with the Federal Emergency Management Agency at the World Trade Center following the attack of 9/11, and on hurricanes Katrina and Rita. His work in the arena of emergency response helped provide a foundation for incident management implementation throughout the nation. Alan will be missed by all who knew him for his wit, humor, and dedication to his family and the fire program he served so well.

Cover Photo

Prescribed burn to restore a Missouri Oak savanna and woodland by Lee Wilbeck, Missouri Department of Natural Resources, State Parks; used with permission.

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Proceedings of the 4th Fire in Eastern Oak Forests Conference

May 17-19, 2011

University Plaza Hotel
Springfield, MO

Edited by

Daniel C. Dey
U.S. Forest Service, Northern Research Station

Michael C. Stambaugh
University of Missouri-Columbia, The School of Natural Resources, Dept. Forestry

Stacy L. Clark
U.S. Forest Service, Southern Research Station

Callie J. Schweitzer
U.S. Forest Service, Southern Research Station

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In cooperation with the MU Conference Office

Angela Freemyer, *Conference Coordinator*
Lindsay Kilgore, *Administrative Assistant*
Sharon Rodes, *Graphic Designer for program*

Review Process

Each paper was peer-reviewed by two reviewers. The authors worked with the editors to revise their manuscripts, based on the reviewer's comments. Revised manuscripts were edited for formatting by Nancy Bastin, U.S. Forest Service, Southern Research Station, and then submitted to the Northern Research Station Production Services for technical editing and design. Reviewers were: Christopher Webster, Michael Jenkins, Jesse Burton, Terry Bidwell, Scott Nielsen, Charles Lafon, Justin Hart, Richard Guyette, Lyndia Hammer, Mike Leahy, Ryan McEwan, Joe Marschall, Mike Stambaugh, David Engle, Nathan Klaus, David King, Martin Spetich, Susan Loeb, Joy O'Keefe, William Sutton, David Steen, Greg Spyreas, and Greg Nowacki.

WELCOME

This conference was a major symposium focused on fire in oak forests, woodlands, and savannas where noted experts in research and management gathered to present state-of-the-art information, perspectives and synthesis on key issues. The conference has been held previously in:

- Carbondale, IL (2008): Forest and fire management, effects and modeling in the Central Hardwood Region
- Columbus, OH (2005): Delivering science to land managers
- Richmond, KY (2000): a workshop on fire, people and the Central Hardwoods landscape

This symposium took place in the most western location of the eastern deciduous forests in the conference's history. This is a landscape that is really different than those that have been represented at the previous conferences, a land that has a rich history of human settlement and fire use, and where managers have been using prescribed fire for more than 30 years on large-scale to landscape scale projects. The theme of this conference was "Managing Oak Woodlands & Savannas in the Forest-Prairie Region". The conference featured 15 invited presentations and 45 posters by scientists and managers from around the United States, who provided information on fire use and effects on a wide range of topics including:

- Human, fire and natural history of the region – foundations of understanding and basis for restoration
- Examples of restoring oak-pine woodlands and savannas from Texas, Arkansas, Missouri, and Oklahoma
- The history and current use of fire in natural resource public agencies in the Midwest
- Grazing and fire interactions in grasslands
- The role of fire in the southern Appalachian Region during the Holocene
- Managing woodlands and savannas for songbirds, bats, and reptiles
- Managing invasive species in woodland and savanna restoration
- Tools for assessing disturbance risk and management options
- Introduction of the Midwest Oak Woodland and Forest Fire Consortium

Many of the presentation topics were reinforced by the four field tours that featured the rich and relatively long-term history of using prescribed fire to manage for a wide range of natural resource and conservation objectives. There was a mix of topics on using fire to manage vegetation and wildlife and restore native ecosystems presented by scientists and managers who are noted leaders in the use of prescribed fire in this region.

More than 250 people attended the conference representing 22 states, 1 Canadian province, and 65 different organizations, institutions, agencies, and private entities.

We were glad to host you in Missouri. I hope your time here was richly rewarded by the fine presentations and field tours, by good fellowship with each other, and by your active participation in all our discussions.

Dan Dey

Photo by http://armedwithvisions.com

Cloak of Darkness

By Dan Dey
May 9, 2011

Campfire kindled for evening meal,
burns bright to turn back the night intruder.
We stoke the flames with one more branch,
taken from yonder wood.
With deafness to the call for slumber,
we dance as flames in merry delight.

Now in silence, we gaze in eternal wonder,
into the depths of fiery magic.
Drawn close, by visual splendor,
until we cannot stand the heat.
Minds grope for answers to grand questions,
pondered by the generations,
refined like metal through endurance,
we long to find the truth we seek.

CONTENTS

ABSTRACTS (continued)

CHARACTERISTICS OF EASTERN U.S. OAK FIRE REGIMES

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FIRE HISTORY IN A SOUTHERN APPALACHIAN DECIDUOUS FOREST

Norman L. Christensen, Jr. and Kurt Fesenmeyer

Research Professor (NLC), Nicholas School of the Environment, Duke University, A124 LSRC, Box 90328, Durham, NC 27708; GIS Specialist (KF). NLC is corresponding author: to contact, call 919-613-8052 or email at normc@duke.edu.

Abstract.—Because there are few long-term dendrochronological and lake sediment data for the southern Appalachians, little is known regarding the history of fire in this region's forests through the Holocene. Radio-carbon ages for 82 soil charcoal samples collected from local depositional sites along a topographic gradient from mixed hardwood (*Liriodendron tulipifera* and *Quercus* spp.) to oak-pine (*Quercus prinus* and *Pinus rigida*) forest provide a coarse-grained picture of changes in fire frequency within a 10-ha area during the Holocene. Fires were frequent over the past 4,000 years, and their frequency appears to have increased significantly about 1,200 years before present (YBP), coinciding with the advent of the Mississippian Native American culture in this region. Our results are consistent with the widely held view that fires have become less frequent in this region over the past 250 years. The inbuilt error associated with estimating actual fire dates from charcoal fragments inherently limits our ability to infer the specifics of and changes in fire regime over time. Notwithstanding these uncertainties, it is clear that fire has been an important part of eastern deciduous forest ecosystems through much of the Holocene. Nevertheless, fire regimes and forest composition have likely changed during this time owing to changes in human activities and climate. These facts have important consequences for restoration and fire management.

INTRODUCTION

Up to the late 20th century, most ecologists tacitly assumed that fire was not an important factor in eastern deciduous forests. Nowhere is fire mentioned in E. Lucy Braun's monographic treatment of this biome (Braun 1950). Oosting (1942) likewise omitted any mention of fire in his survey of the successional and mature communities of the eastern Piedmont. Whittaker (1956) speculated that wildfire played an important role in the maintenance of ridge top shrub and pine dominated communities of the southern Appalachians but not in deciduous hardwoods. Nevertheless, early explorers and settlers often described the vegetation of these regions as woodlands with open grassy understory conditions indicative of high frequency, low intensity fire regimes (Bartram 1791, Lawson 1701). The prevalence of fire-adapted oaks (*Quercus*), hickory (*Carya*) and chestnut (*Castanea*) supports this assessment. Occasional stand replacement fires may have been important during times of severe drought, particularly if occurring after canopy disturbance (e.g., blowdown) when dead and down fuel levels were high (Abrams 1992, Brown 2002).

Nowadays, there is general agreement that fire played an important role in presettlement deciduous forests, and debate has shifted to the nature and frequency of fire in the array of mixed pine-hardwood and hardwood communities along topographic and moisture gradients (Brose and others 2001, Delcourt and Delcourt 1997, Runkle 1985, van Lear and Waldrop 1989). It is almost certainly the case that fire regimes changed through the Holocene with changes in climate and use of fire by Native Americans (Delcourt and Delcourt 1997, Delcourt and others 1986, Gragson and others 2008, Springer and others 2010), and the changes following European settlement were even more pronounced (MacCleery 1996, Nowacki and Abrams 2008, Yarnell 1998). During the 17th and 18th centuries, increased human access, land clearing, and altered fuel conditions likely increased the frequency of fires (Jurgelski 2008). Over most of the last century, however, active fire suppression and landscape fragmentation have reduced fire frequency in most areas (Jurgelski 2008). Nowacki and Abrams (2008) argue that successional change in the absence of fire has "mesophied" eastern deciduous forests, that is, it has altered forest composition and structure in ways that have greatly diminished flammability and, therefore, fire likelihood across this entire region.

The nature of presettlement fire regimes and the changes that have occurred in those regimes since European settlement are quite relevant to modern forest management. They are key to our interpretation of the likely composition and structure of presettlement forests and, thus, to the establishment of targets for forest restoration (Nowacki and Abrams 2008, Scharf 2010). They are important to our understanding of the likely responses of species to the increased use of prescribed fire for understory fuel management and forest restoration. Finally, land managers need such information in order to emulate historic disturbance regimes through silvicultural practices (Engstrom and others 1999, Kimball and others 1995, Vose and others 1999). While regional understanding of fire occurrence and frequency is useful, we are most in need of stand-level information on fire return intervals and fire behavior.

Dendrochronological studies in the Appalachians have provided useful information regarding fire frequency in oak-pine dominated forests during the past several centuries (Abrams and others 1995, Guyette and Cutter 1991, Nowacki and Abrams 1997, Ruffner and Abrams 2002, Schuler and McClain 2003, Shumway and others 2001). However, fire-scarred trees older than 400 years are exceedingly rare in this region.

Charcoal in wetland or lake sediments has been used to evaluate changes in fire occurrence at watershed or regional scales (Delcourt and Delcourt 1997, Delcourt and others 1986), but appropriate depositional environments are rare over much of this biome. Springer and others (2010) combined analyses of isotopic anomalies in stalagmite carbon with radiocarbon dates of sediment charcoal from a West Virginia cave to infer changes in fire regimes and human land use in the area surrounding the cave. These studies provide insights into long-term changes in the frequency of fire on landscapes surrounding the sample sites. They do not reveal much information about fire regimes at the scale of individual stands or variation in fire regimes among forest communities within a watershed. Welch (1999) used soil charcoal to verify the past occurrence of fire in southern Appalachian pine stands, but charcoal fragments were not carbon dated and it is not clear when these fires occurred.

Here we summarize our recent study (Fesenmeyer and Christensen 2010) in which we used radiocarbon dating of small pieces of charcoal in soil to reconstruct the presettlement, stand-level fire history along a topographic gradient in a southern Appalachian forest. We discuss both the limitations and the values of that study both to our understanding of the changing role of fire during the Holocene and the application of that understanding to restoration management.

METHODS

Our study was carried out in the Wine Spring Creek Ecosystem Management Area (WSCEMA) of the Nanatahala National Forest in Macon County, North Carolina (35°N latitude, 83°W longitude). The WSCEMA is located on the western slope of the Nantahala Mountains. Sampling was done over an area of approximately 10 ha at 1,280-1,430 m elevation, extending from a perennial stream across a south-southwest-west facing slope to a ridge top. The sample area was selected because it included the gradient of vegetation from hemlock-hardwood cove near streams through mixed-oak hardwood on hill slopes to chestnut oak-pine forests on the ridge top. Live pitch pine trees were widely scattered on the ridge top; dead boles and logs of this species were more abundant, suggesting that its abundance has decreased over the past several decades. The well-developed soil A horizon indicated that this site had never been subjected to agriculture. A 1997 prescribed fire at this site provided abundant charcoal of recent age for comparison. Other research from WSCEMA has focused on the effects of this prescribed fire and is described in Elliott and others (1999) and Vose and others (1999). There was no record nor was there any evidence at this site of any other fires in the past century.

Wood charcoal is an inert and recalcitrant form of carbon that can persist in soils for millennia following fires (De Lafontaine and others 2011). Although charred material may be transported significant distances by wind and water, the vast majority of charcoal that accumulates in depositional locations (such as small depressions) within a forest likely originated within a few tens of meters of those locations (Blackford 2000, Higuera and others 2007). Thus, the range of charcoal ages at sites likely reflects the history of fire at or very near that site.

Sediments containing significant amounts of charcoal often accumulated immediately upslope and downslope from rock outcrops (height = 0.5 to 3 m) scattered across the study area. Using a slide hammer soil sampler, we obtained 5 cm diameter intact soil cores from 33 rock-outcrop locations. After removing the O horizon (duff layer), we collected soil cores varying from 10 to 30 cm depending on the depth to bedrock at each sample location. For purposes of comparison, sample sites were divided into two groups—xeric (ridge top and upper slope oak-pine sites) and mesic (downslope hardwood sites)—based on overstory vegetation cover.

In the laboratory, soil cores were divided into 2 cm sections, taking care to preserve the stratigraphy of the sample. The details of laboratory treatment and selection of charcoal fragments (1-50 mg each) are described in Fesenmeyer and Christensen (2010). A total of 82 individual charcoal fragments from 18 sample locations were selected for carbon dating. This number was determined by funding available for carbon dating. Fragments were selected to represent entire core horizons among different vegetation types on the site. The calibrated ^{14}C radiocarbon date and radiometric error of each fragment was determined using accelerator mass spectrometry (AMS) (Reimer and others 2004). The radiometric (2 sigma) error associated with AMS dates varied among samples from as little as 20 years to as much as 400 years. These data were subjected to cumulative probabilities analysis (Meyer and others 1992) using the "sum probabilities" option in CALIB 5.0.1 (Lafortune and others 2006, Stuiver and others 2005). This analysis allowed us to represent for a given year the relative probability that any fire represented in our dataset occurred at the site.

The calibrated radiometric age and error of a charcoal fragment corresponds not to the age of a fire event, but to the time when the wood that comprises a charcoal fragment was actually produced. The so-called "inbuilt error" of a date is estimated by the probability distribution of the difference between the date of carbon assimilation into wood and the date of the fire event that consumed and converted the wood to charcoal (Carcaillet 1998, Gavin 2001). The inbuilt error is additive to the radiometric error, and it is typically assumed to depend on stand age structure and the rate of wood decay in the ecosystem (Gavin 2001, Gavin and others 2003). However, the age distribution of charcoal remaining after a fire is also influenced by the prevailing fire regime itself, because fire frequency and intensity affect both the amount of available fuel and how much is actually charred (Higuera and others 2005).

RESULTS

Calibrated carbon dates ranged in age from 0 (probably corresponding to the 1997 prescribed fire in this area) to 4,000 years before present (YBP), with the exception of one fragment dated at 10,560 ±120 years. This was so far outside the range for all other samples that it was not considered in subsequent analyses. However, this very old fragment is evidence that fires occurred at this site in the early Holocene at about the same time that humans first occupied this region (Gragson and others 2008). Charcoal sample ages (± 2 sigma) are arrayed according to age and site, and the summed probability distributions for the two site groups are plotted in Figure 1. This distribution of relative probabilities suggests that fires have regularly occurred at xeric oak-pine sites over the past 4,000 years. The record of fire events at mesic hardwood sites begins about 2,000 yr ago. There is a relatively abrupt increase in the relative probability of a year being represented in our charcoal sample at about 1,000 YBP in both xeric and mesic sites.

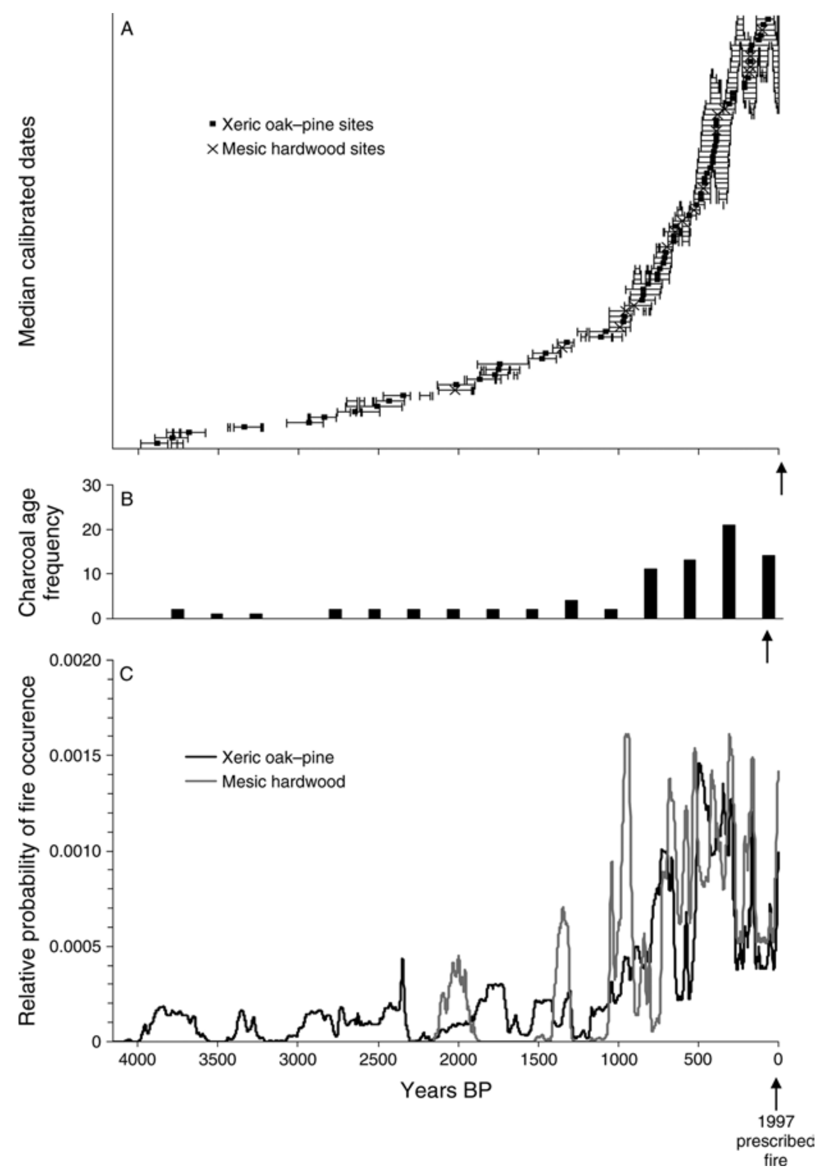

Figure 1.—A) Ages of 82 charcoal fragments arrayed by calibrated ARM date with 2-sigma error bars. Note that 2-sigma errors are discontinuous for some ages. B) The frequency of charcoal ages in 250 year time bins. C) The summed probability that any fire represented in our dataset occurred at the site. (Figure used with permission from Fesenmeyer and Christensen 2010).

Using Gavin's methodology and age-distributions (Gavin 2001, Gavin and others 2003) for existing forests and wood decay rates for this region suggested by Harmon and others (1986), the inbuilt error for the oak-pine forests is likely between +50 and +100 years, although without knowledge of the actual fire regime, this estimate is quite uncertain. Therefore, the most probable dates of fire events are, on average, 50-100 years younger than the most probable radiometric dates of charcoal fragments. Furthermore, the radiometric errors used in the summed probability analysis underestimate the total error surrounding estimated dates of fire events. Thus, it is not possible to determine whether charcoal fragments with ages that differ by less than 150-200 years are products of the same fire or of different fires.

Fires clearly have occurred regularly on this forested slope over the past 4,000 years. Furthermore, fires were not confined to ridge top sites that currently support oak-pine forests, but they extended into areas that are today dominated by mesic hardwood communities. That no charcoal fragments > 2020 YBP were found in mesic hardwood sites may have been due to unequal sampling effort rather than inherent differences in fire regime between site types. However, it is well established that their moist, flaccid litter renders mesic hardwood ecosystems more resistant to fire than other forest types (Nowacki and Abrams 2008).

The number of samples represented in particular age bins diminishes with increasing charcoal age. This pattern of diminishing data with time is inherent in virtually all historical data sets (Egan and Howell 2001). However, the abrupt appearance 4,000 YBP of regular fires and the marked increase in the summed relative probability of charcoal ages at 1,000 years are consistent with historical trends in human activity and fire frequency hypothesized for the southern Appalachians by Delcourt and Delcourt (1986, 1998).

DISCUSSION AND MANAGEMENT IMPLICATIONS

Abrams and Nowacki (2008) argued that Native Americans used fire widely to manipulate vegetation in the eastern United States. Delcourt and Delcourt (1986, 1998) noted that Native Americans of the Woodland Tradition appeared in the vicinity of our study area about 4,000 years ago, and they attribute fires during that period to these hunter-gatherers. Mississippian people appeared in this area about 1,000 YBP, coinciding with the appearance of *Zea maize* and a number of weedy herbs in the pollen record. The widespread use of fire by Mississippian Native Americans is well documented (Delcourt and others 1998, Hatley 1993).

In eastern Kentucky, Delcourt and others (1998) linked Native American use of fire to the dominance of oak-hickory forests starting 3,000 yrs ago (also see Ison 2000). Springer and others (2010) recent analysis of pollen and charcoal deposits in a West Virginia cave suggests an increase in fire in that location beginning 4,000 YBP and lasting until the arrival of Europeans. Here, too, Native Americans were implicated. In the only additional soil charcoal study in the southern portion of the Eastern Deciduous Biome, Hart and others (2008) described a comparable range of fire occurrence (events spanning 6785 to 174 YBP) in a hardwood deciduous forest on the Cumberland Plateau of middle Tennessee (located 185 km from our study area).

Because of the uncertainties associated with radiometric and inbuilt errors, it was not possible to estimate fire return intervals for our study area. Nevertheless, the summed probability plot (Fig. 1c) is consistent with frequent, low severity fires during the past millennium. This time period coincides with the expansion of Mississippian people into this area. If our charcoal samples are representative of the fire events during the period 4,000-1,000 YBP, then we might speculate that fires during that period were less frequent and, therefore, more severe during that time interval.

After accounting for the prescribed fire, a simple histogram of charcoal ages suggests that fires may have become less frequent during the past 250 years (Fig. 1b). This coincides with the decline in Native American populations in this region after the beginning of European exploration and settlement. Changes in fire frequency over the past millennium were undoubtedly influenced by changes in climate as well.

The actual composition of the forests that burned during the past four millennia remains unknown. However, this amount of fire suggests that pines were likely far more abundant in presettlement times than today. The nature of presettlement fires (e.g., their seasonality and severity) also remains unknown. But these data suggest that fire regimes changed considerably from Woodland to Mississippian to European Settlement times.

These results eliminate any doubt about whether fire played a significant role in presettlement deciduous forest ecosystems; it most certainly did. Given its historical importance, we can also presume that many deciduous forest species are in various ways adapted to fire regimes of one kind or another. Although the specifics are far from clear, changes in fire regimes (fire frequency and severity) undoubtedly caused changes in the relative abundance of different tree species from one time period to another. As fires became more common during the past 4,000 years, they likely contributed to increased prevalence of pines and fire tolerant hardwoods such as oaks and hickories. The significant diminution in fire occurrence over the past two centuries has contributed to the increased prevalence of fire intolerant species such as red maple (Abrams 1992) and to the "mesophication" of eastern forests (Nowacki and Abrams 2008).

These very general conclusions are directly relevant to restoration and fire management in eastern deciduous forests. First, the historic range of variation in fire regimes and ecosystem structures over the past several millennia appears to have been very large compared to the expected range of variation in these features over decadal time intervals. It is variation over these shorter time intervals that is most relevant to restoration and fire management today. Second, even though the specifics are vague, changes in fire regimes over time suggests that the choice of particular historical restoration targets, whether for forest composition or fire prescriptions, is arbitrary. Furthermore, even if we knew the specifics of the relationship between forest composition and fire regime change, restoration to a particular historical target may not be possible. Given ongoing climate change, land fragmentation, and abundant non-native species, restoration of a particular fire regime will not necessarily restore the community of species that were historically associated with it.

This does not mean that detailed information on past fire regimes and their relationship to forest composition is not important. The end goal of fire restoration management is not fire itself, but the diversity, structure and key processes in eastern deciduous forest ecosystems. A clear understanding of the details of the historical relationship between fire regimes and these ecosystem features is critical to achieving that goal, and we have much yet to learn.

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ANTHROPOLOGY OF FIRE IN THE OZARK HIGHLAND REGION

David H. Jurney

Heritage Program Manager, Ozark-St. Francis National Forests, U.S. Forest Service,
605 West Main, Russellville, AR 72801. To contact, call 479-964-7237 or email at djurney@fs.fed.us.

Abstract.--Native Americans are often considered to have exploited available natural resources rather than modifying their environments to maximize yields. As simpler societies evolved into more complex ones, there is a consensus that intensification of habitat modifications also increased. However, landscape scale archeological inventories now show relatively intensive modifications of specific landscapes through most of human history, including so-called simpler societies. Records of these modifications are difficult to obtain. Archeological site distributions can be used to understand human settlement and selection of particular ecosystems. Another method uses tree-ring dating (dendrochronology) on fire-scarred trees that recorded fire frequencies and fire return intervals at points on the landscape that frequently burned due to natural and cultural processes. General Land Office records (1815-1840) provide firm evidence for culturally-induced changes in species distribution and historical landscape vegetation mosaics for the Ozark Highlands and Arkansas River valley.

INTRODUCTION

The Ozark Highlands have for a long time been considered a “marginal” region marked by cultural conservatism and lack of developed prehistoric agricultural and urban settlement (Sabo 1986). This concept has also been applied to humans practicing a hunting and/or gathering subsistence strategy, and so-called “primitive” societies are considered to lack sophisticated environmental management skills. Through the study of human use of fire and inter-regional comparisons of patterns of cultural development, the “marginal-primitive” view of Ozark prehistory and history is now challenged. The theme of this paper is that fire use by native people at various stages of cultural complexity indicates landscape scale manipulation of natural resources, implying greater environmental sophistication of hunter-gatherers than previously supposed.

Throughout the history of anthropology, cultural anthropologists and archeologists were bounded by theoretical assumptions that reflected the accepted norms of European and Euro-American societies; primarily those of colonial powers. Julian Steward, for example, popularized the notion of “cultural ecology” but viewed Indians, particularly hunters and gatherers, as living fossils representing an earlier, more “primitive” stage of human evolution (Steward 1955). Cultural ecology became the dominant paradigm in North American prehistoric archaeology, where “all cultures are adaptations to their specific environments” (Cande 2008). A few voices in the wilderness, particularly Day (1953) and Stewart (1956, 2002), challenged both ecological and anthropological theorists with field data on Native American use of fire, cultural and subsistence practices, and the broader environmental consequences resulting from burning particular habitats at certain times of the year.

Miller (1972) employed the hypothesis that hunter-gatherers were totally dependent, and agriculturalists partially dependent, on the available natural vegetation and fauna. Citing Steward (1955), Miller posited that "fire was the prime tool available" to native North Americans, and his work using General Land Office (GLO) records and historical accounts of northwestern Arkansas and the Ozarks has become a seminal environmental reference for archeologists working in the region. Miller hypothesized that several plant communities (and associated fauna), especially prairies (upland and lowland), barrens (open areas with frequent fire regime), and open oak woodlands, were maintained for at least the last 1500 years by Native Americans (see Curtis 1959, Miller 1972, Sears 1925), and substantiated this inference with the prehistoric floral and faunal materials recovered from bluff shelters in the Ozarks (Cande 2008). Similar patterns of human use of fire are known from the Cumberland Plateau (Delcourt and others 1998). Ecological communities such as open oak woodlands, barrens, and prairies were maintained by Native American burning, and possibly natural fires, and were rapidly replaced by full forest vegetation (Kay 2003) after Euro-American settlement by the relative suppression of fire and landscape fragmentation. In another study, Jurney and Stahle (2004) found that some Ozark Highland prairies found in the GLO records are located on soils that formed under hardwood forest, suggesting that these prairies may have been culturally induced. During the 19th century, dendrochronological (tree-ring dating) data from fire-scarred trees indicate burning was conducted in some forested areas (e.g., Cross Timbers of western Ozarks) to improve range (Clark and others 2007, Griffin 2002), while other areas (e.g., central Ozarks) received less burning (Guyette and others 2002, Guyette and Spetich 2003). In the 20th century, fire was removed from many areas of the Ozark ecosystem, particularly where national forests were established and fire was considered counterproductive for timber management (Bass 1981).

The ecological role of fire and human influences over the structure and distribution of plants has and will continue to be controversial. Two extremes in this debate argue that: 1) fire is destructive and should not be (or never was) applied by humans to the landscape, and 2) fire is an essential management tool to restore forests and grasslands to more natural, ecologically healthy, or desired historical reference conditions. The use of long-term, longitudinal ecological studies has challenged the traditional "succession" paradigm of "let Mother Nature take her course", or "study only the wilderness", and the human role in vegetation changes through time is now a generally accepted fact (Anderson 2006, Denevan 1992, Mann 2005, Raup and Carlson 1941). The anthropological assumption that hunter-gatherers were "primitive," ecologically inept, and bound to the resources available to them (Anderson 2006) is central to the hypothesis that the origins of agriculture constituted a "revolutionary change" in how humans related to their particular environments. This theoretical bias persists today in archaeology and is challenged by the evidence presented in this paper.

STUDY AREA

The study area falls primarily in the Ozark Plateau of north-central Arkansas, with outliers in the Mississippi Embayment and Gulf Coastal Plain, the Arkansas River Valley, and the northern uplift of the Ouachita Mountains (Magazine Mountain). The Arkansas Valley consists of Pennsylvanian-age clastic sediments arranged into broad synclines (troughs) with relatively narrow intervening anticlines (arches). The synclines are most conspicuous and are formed when the capping sandstones are breached, resulting in a more rapid erosion of underlying shale. The Ozark Plateau is made of generally level-bedded Paleozoic age strata divided into the Salem Plateau, the Springfield Plateau, and the Boston Mountains (USDA FS 2005). All these plateaus are deeply dissected by numerous streams. The St. Francis National Forest is located on Tertiary marginal marine and coastal plain continental deposits with a veneer of Quaternary terrace and alluvial deposits. Crowley's Ridge, formed by aeolian loess, runs through the center of this forest adjacent to the floodplains of the St. Francis River and the Mississippi River.

Land managers, archaeologists, and geoscientists need to develop sound baseline historical data on plant distributions and ecological processes over entire landscapes (Bragg 2002, 2004; Warren 1984; Warren and O'Brien 1984) as an aid to current forest, shrubland, and grassland ecosystem management and paleoenvironmental reconstruction. The National Hierarchical Framework of Ecological Units (NHFEU) is a standardized classification and mapping system for dividing the earth's biota into progressively smaller areas of increasingly similar ecology, providing baseline management information for each unit. It is derived from patterns of climate, soils, air quality, hydrology, geology, landforms and topography, potential natural plant communities, and natural disturbances. The Ozark-St. Francis National Forests in Arkansas (OSFNF) have completed (USDA FS 2005) an approximation of land type associations (LTAs) tiered to the NHFEU. This Forest has management units falling in the northern Ouachita Mountains (Magazine Mountain), the Ozark highlands and the Arkansas River valley rolling hills (Wedington Unit, Bayou, Buffalo, Pleasant Hill, Boston Mountain), and the Mississippi (St. Francis NF) alluvial valley (Fenneman 1938, Thornbury 1965). The LTAs generally trend east-to-west following the sloping trend of the underlying geology, with some orientation along streams that drain the interior Ozarks.

The maximum southern advance of glaciers (northern Missouri) occurred around 18,000 years before present (YBP, 1950 as datum). Pleistocene records from various paleoenvironmental contexts (pollen in adjacent areas, solution cavern speleothems, dendroclimatology) indicate the Ozark forests were composed of boreal species, predominantly spruce (*Picea* spp.). Summers were substantially cooler than present. Faunal species included the Columbia mammoth (*Mammuthus columbi*), American mastodon (*Mammut americanum*), ground sloth (*Megalonyx jeffersonii*), and horse (*Equus* spp.). Around 14,000 YBP, more mixed boreal forest developed with spruce and jack pine (*Pinus banksiana*), and the addition of oak (*Quercus* spp.), hickory (*Carya* spp.), ash (*Fraxinus* spp.), and maple (*Acer* spp.) as minor components. This mosaic of boreal and deciduous ecosystems has no modern analogue in present North America (Denniston and others 2000), and large scale herbivory may also have contributed to the vegetation mosaics of this period (Collins 1992, Nelson 2005). A brief cooling retrenchment occurred between 12,900 and 11,500 YBP, referred to as the Younger Dryas (Dansgaard and others 1989), which abruptly terminated with warming up to 7 °C (12.6 °F) in just a few years. Many large herbivores became extinct just prior to or following this event. Dry westerly winds increased, with eventual intrusion of prairie grasses into the Ozarks (Denniston and others 2000). Around 9,500 YBP oak savannah was well established across the Ozarks. A rapid decrease in speleothem values ca. 9,500 YBP indicates an increase in deciduous vegetation (Denniston and others 2000), followed by a prairie dominated environment about 7,500 YBP. Nelson (2005) attributes the spread of prairie ecosystems from the Great Plains to Ohio during this period to the combined effects of warming conditions, increased aridity, and Native American fire sets. This climatic episode was interrupted by increased deciduous forest ca. 4,500 and 3,000 YBP. Prairie returned between 3,000 and 1,500 YBP, with the establishment of current deciduous forest (Denniston and others 2000). The climate record shows that radical environmental and climatic shifts (Stahle and others 2000) have occurred in the Ozarks, with commensurate changes in fire ecology. Periods with expansion of grasslands were probably most pyrogenic, followed (in descending order of fire frequency) by pine forests, open oak woodlands, and closed deciduous forest. Humans were present throughout the last 18,000 years, with increased population spikes ca. 4,500 and 1,500 years ago (Sabo and Early 1990), domestication of wild plants and use of tropical cultigens ca. 4,200 YBP, and the introduction of maize agriculture ca. 750 YBP. These population spikes occurred during shifts from grassland to deciduous forest, and from forest to grasslands.

The modern climate is characterized as temperate, with west-to-east moving atmospheric circulation of air masses originating from the eastern Pacific, western United States, the Gulf of Mexico, and Canada. During winter, extreme north-to-south temperature variations occur. During summer, daily variations are less significant from north-to-south. Rainfall averages 114.3-139.7 cm (45-55 inches) per year, with March, May, and November the wettest and July, August, and January the driest (USDA FS 2005). Thunderstorms are greatest during the spring, summer, and fall, averaging 10-25 events during each season. Tropical cyclone and frontal passages are most severe in winter and spring. Tornados have been reported every month but are most frequent in April and May. Nine major droughts (reconstructed below average June-July rainfall) have occurred during the last 308 years (1703-1704, 1727-1741, 1780-1789, 1831-1840, 1855-1866, 1886-1887, 1923-1925, 1950-1957, and 1963-1965), generally lasting 2 to 3 years in duration (Blasing and others 1988).

Increased wildfires of human origin and lightning ignitions are noted for these periods, with some corresponding to drought episodes (Clark and others 2007, Jurney and others 2004). Florida and Texas experience more lightning strikes than any other areas of the United States (Orville 1992, Silver and Orville 1994), yet Florida's fire records indicate an average of 15 lightning-caused fires per million acres per year (437.56 lightning fires per year; 8.2 percent of total). Texas fire records indicate an average of one lightning fire per million acres per year (23.5 fires per year; 1 percent of total); and Florida has almost three times as much rainfall (Jurney and others 2004). Arkansas fire records indicate an average of 4.3 lightning fires per million acres per year (93.5 fires per year; 2.6 percent of total). Only during exceptional climatic episodes does lightning match the frequency of human-caused fires (Jurney and others 2004). In Arkansas from 1916-1990, the mean lightning-caused fire frequency was 2.78 lightning fires per year (range 0-25.7 percent). In this 75-year period of record, 26 years had greater than average numbers of lightning-caused fires, 15.4 percent of which were drought years (1956, 5.7 percent of total; 1954, 4 percent; 1964, 3 percent; 1923, 2.8 percent) with less than average tree-ring growth. In the 49 years of less than average numbers of lightning-caused fires, 8.2 percent were drought years (1924, 2.7 percent of total; 1925, 2.7 percent; 1965, 2.6 percent; 1957, 1.8 percent; 1952, 1.7 percent; 1953, 1.7 percent; 1963, 1.6 percent; and 1951, 1.1 percent). These data suggest that during drought years there is a greater chance for lightning-caused fires; however, during some drought years, human-caused fires also proportionately increased.

Another aspect of climatic cycles is the El Nino-Southern Oscillation (ENSO) which are short duration 3-7 year events which intensify both droughts and wet periods. ENSO events are known in the Pleistocene (radiometrically dated to at least 10,770 YBP), and it is possible that they can trigger broader patterns of climate change (Dunbar 2006, Guyette and others 2006). In the Ozarks, ENSO events are marked by the presence of 70 frost-damaged growth rings (spring wood) in oaks over the last 330 (AD 1650-1980) years that formed from false spring events (late spring severe frost damage captured in tree growth rings), correlating with the La Nina phase of ENSO (Stahle 1990). In 2007, such a false spring hit Arkansas during a La Nina year and produced extensive damage to both white oak (subgenera *Leucobalanus* with an annual nut mast) and red oak (subgenera *Erythrobalanus* with a 2-year nut mast) acorn yields, with lag effects lasting into the spring of 2008. Thus, periods of drought and false spring events that show the influence of ENSO have the potential to increase the effects of wildland fire ignitions (Clark and others 2007, Guyette and others 2006) and allow humans to further influence the spread of landscape scale fires. Also, climate changes and fluctuations influence the annual yields of nut mast and other natural resources, as well as predisposing the landscape to large scale fire ignition and fire spread.

PREHISTORIC AGROFORESTRY

Native agroforestry practices are increasingly recognized through archaeological research, use of aerial and satellite imagery, and paleoenvironmental reconstruction around the planet. Many Native American elders report that plants benefit from human use, and many may actually depend on humans using them to fully function in the ecosystem (Anderson 2006). Denevan (1992) argues that the forest structure and landscapes of North and South America were greatly influenced by humans through the widespread use of fire to concentrate food resources. Forests of widely spaced trees with grass and herb understory were observed in New England, the Midwest, and southeastern North America upon European introduction but have vanished today (Black and others 2006, Denevan 1992). Data compiled from modern vegetation studies (Guldin and others 1999) compared to GLO spatial data (Foster and others 2004, Foti 2004) indicate that the Ozark Highlands forest of Arkansas is 2.3-2.8 times denser today than 180 years ago. Davies (1994) suggests that periodic low-intensity fires, particularly between November and April, increased vegetation yield from 20-100 percent and increased large birds and mammals by 100-400 percent. Humans used fire to concentrate foods on highly productive portions of the landscape (Williams 1993) and may have used similar strategies to achieve a "primary forest efficiency," a phrase coined by Caldwell (1958) and amplified in the works of Smith (1974, 1975, 1978) in the Mississippi Valley and in the Ozarks by Raab (1976) and Lockhart and others (1995). My discussion of Ozark Native American use of fire to modify their environment summarizes known information relating to archaeological finds of plants whose life histories suggest some form of human intervention in the ecosystem (Nowacki and Abrams 2008), dendrochronology, and proxy records of vegetated landscapes. Many species that are rare today appear to be artifacts of such fire regimes, including the federally endangered red-cockaded woodpecker (*Picoides borealis*) and Indiana bat (*Myotis sodalis*).

Beginning with the prehistoric archaeological record in the Arkansas Ozarks, dry rockshelter deposits preserve perishable material culture and food remains and provide empirical evidence for anthropogenic landscapes. Figure 1 shows giant cane (*Arundinaria* spp.) remains from the Penhook Rockshelter (3PP402) on the Middle Fork Illinois Bayou in the Lower Atoka Hills and Mountains LTA. This species is rare in the vicinity of this shelter today. Cane was a ubiquitous material found in dry shelter deposits that was used for food, basketry, tools, and other functions. It is dependent on fire to maintain habitat and stand viability. For the Late Prehistoric and Historic periods, the GLO surveys and some historical accounts provide data on landscape-scale vegetation patterns and disturbance regimes. General Land Office records indicate extensive stands of fire-dependent cane (*Arundinaria* spp.) were present in early 19th century Ozark valleys (Fig. 2) in the Lee Creek Rolling Hills, Lee Creek Atoka Hills, Atoka Mulberry Mountain Valleys, Arkansas Valley Hills, Lower Atoka Hills and Mountains, Bloyd Mountain Valleys, White River Rolling Hills, Wedington Boone Hills, and Mississippi Bottomlands LTAs. The floodplain landtypes within these LTAs require a 5-10 year fire return interval for health of cane stands (Brantley and Platt 2001, Frost and others 1986, Frost 1995, Gagnon 2006) and were probably influenced by, if not created by, late prehistoric burning practices.

Figure 1.—Giant cane (*Arundinaria* spp.) fragments from the Penhook Rockshelter (3PP402).

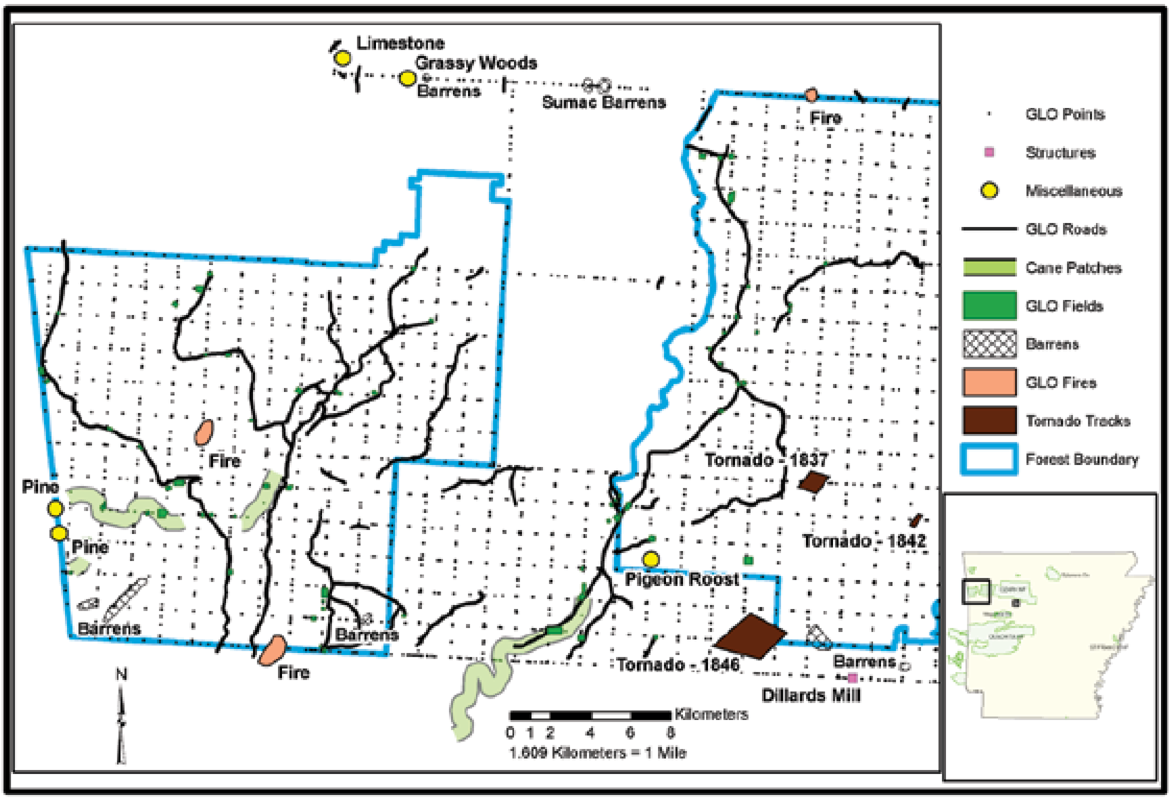

Figure 2.—General Land Office information related to vegetation and cultural and natural disturbances on the Lee Creek Unit of the Ozark-St. Francis National Forests.

Most evidence for prehistoric manipulation of their environments is derived from the excavation of perishable food and material culture remains from Ozark rockshelters, focused primarily in northwestern Arkansas. Most of the excavations were done in the 1930s and lack firm chronological and spatial controls. Since these sites have suffered over 100 years of looting, few remain that can be scientifically investigated today. Therefore, the studies of Gayle Fritz provide the best chronological control for Native American agroforestry, domestication of native wild plants, and adoption of tropical maize-bean-squash agriculture. Of particular note is the domestication of two native plants: *Iva annua* (sumpweeed) which favors floodplains, and *Phalaris caroliniana* (May grass or canary grass) which favors sandy soils and canopy disturbances from floodplains to uplands. Fritz (1986a, 1986b, 1990, 1994, 1997) argues they were domesticated because they have larger seeds than modern varieties and were recovered from archaeological contexts outside their modern ranges. Unfortunately, the paleobotanical literature lists species by presence/absence with little quantitative information on species abundance.

The life histories of other flora and fauna species strongly associated with fire-adapted ecosystems (e.g., shortleaf pine [*Pinus echinata*], various white and red oaks, bison [*Bison bison*], and elk [*Cervus elaphus*]) and their distributions confirm that native fire use played a role in ecosystem structure and function. Table 1 is a list of 64 plants found in Ozark rockshelters (Gilmore 1931) and provides a rough assessment of the ratio of fire-adapted to

Table 1.—List of plant species recovered from Ozark rockshelters and identified by Gilmore in 1931. Theo Witsell (Arkansas Department of Natural Resources) provided updates to the scientific nomenclature. Fire code denotes species response to fire (1 = responds favorably; 0 = fire intolerant) as determined from the Fire Effects Information System database (USDA FS 2012). Note that many species respond favorably to environments cleared by fire. The last column lists any quantified information provided by Gilmore (1931).

Scientific Name	Common Name	Fire Code	Quantitation Information
Pinus echinata	shortleaf pine	1	one specimen
Pinus taeda	loblolly pine	1	one specimen twisted cord
Juniperus virginiana	red cedar	0	common in graves
Typha latifolia	cattail	1	abundant
Tripsacum dactyloides	gama grass	1	no data
Zea mays	Indian Corn	1	abundant all sites
Andropogon furcatus	big bluestem	1	many specimens (*Andropogon gerardii*)
Andropogon scoparius	little bluestem	1	no data (*Schizachyrium scoparium*)
Sorghastrum avenaceum	Indian grass	1	1 specimen (*Sorghastrum nutans*)
Phalaris caroliniana	Carolina canary grass	1	many sheaves
Elymus arkansanus	slender wild rye	1	no data (*Elymus villosus*)
Arundinaria macrosperma	giant cane	1	no data (probably *Arundinaria gigantean*)
Arundinaria tecta	switch cane	1	no data
Scirpus validus	great bulrush	1	1 specimen (*Schoenoplectus tabernaemontani*)
Acornus calamus	calamus sweet flag	0	abundant all sites
Allium sp.	wild onion	1	no data
Yucca sp.	Spanish bayonet	1	no data
Populus deltoides	cottonwood	0	many specimens
Salix nigra	black willow	1	1 bundle
Juglans nigra	black walnut	1	no data
Carya illinoiensis	pecan	0	no data
Carya cordiformis			
Carya laciniosa	shagbark hickory	0	many specimens
Carya alba	mockernut hickory	1	no data
Corylus americana	hazelnut	0	many specimens
Castanea pumila	chinquapin	1	no data
Quercus falcata/rubra	red oak	1	no data
Quercus alba	white oak	1	no data
Quercus marilandica	blackjack oak	1	no data
Quercus imbricaria	shingle oak	0	no data
Quercus macrocarpa	bur oak	1	no data
Ulmus fulva	slippery elm	0	no data
Morus rubra	red mulberry	0	no data
Maclura pomifera	bois d’arc	0	1 specimen bark cord
Amaranthus spp.	pigweed	1	many sheaves
Chenopodium sp.	lambsquarters, goosefoot	1	many sheaves
Asimina triloba	pawpaw	0	1 seed
Platanus occidentalis	sycamore	0	1 specimen
Prunus chicasa	chickasaw plum	1	1 specimen (probably *P. angustifolia*)
Gleditsia triacanthos	honey locust	0	1 specimen
Lespedeza capitata	rabbitfoot clover	1	no data
Phaseolus vulgaris	garden bean	1	varieties all sites
Rhus glabra	smooth sumac	1	many empty panicles

(Table 1 continued on next page)

Table 1 (continued).—List of plant species recovered from Ozark rockshelters and identified by Gilmore in 1931. Theo Witsell (Arkansas Department of Natural Resources) provided updates to the scientific nomenclature. Fire code denotes species response to fire (1 = responds favorably; 0 = fire intolerant) as determined from the Fire Effects Information System database (USDA FS 2012). Note that many species respond favorably to environments cleared by fire. The last column lists any quantified information provided by Gilmore (1931).

Scientific Name	Common Name	Fire Code	Quantitation Information
Sapindus drummondii	soapberry	0	1 specimen
Vitis spp.	wild grape	0	abundant all sites
Tilia americana	basswood, linden	0	no data
Dirca palustris	leatherwood	0	no data
Cornus florida	dogwood	1	no data
Diospyros virginiana	persimmon	1	no data
Fraxinus spp.	ash	0	no data
Apocynum cannabinum	Indian hemp	1	many specimens
Onosmodium subsetosum	Ozark false gromwell	1	no data (*O. molle* var. *subsetosum*)
Dasistoma sp.	false foxglove	1	1 specimen (may be *Aureolaria* sp.), some
Dasistoma spp.	(moved to *Aureolaria*)		
Catalpa speciosa	catalpa	0	specimens
Viburnum sp.	viburnum	1	1 seed
Cucurbita maxima	winter squash	1	1 shell, 1 peduncle, several seeds
Cucurbita ovifera	egg gourd	1	1 complete shell
Cucurbita pepo	pumpkin/summer squash	1	1 complete shell
Iva xanthifolia	marsh elder	1	quantities
Ambrosia trifida	giant ragweed	1	considerable quantities, large seeds
Liatris squarrosa	blazing star	1	no data
Liatris pycnostachya	Kansas gayflower	1	no data
Helianthus annuus	sunflower (two cultivated varieties	1	quantities
Cirsium altissium	thistle	1	1 leaf
Rudbeckia sp.	cone flower	1	1 seed head disc

non-fire-adapted plants exploited by Ozark Native Americans. Since full quantitative information was not available from the literature, each species was coded to indicate species with a favorable response to fire (coded as 1) and species that grow in non-pyrogenic ecosystems or that respond unfavorably to fire (coded as 0) based on the Fire Effects Information System (USDA FS 2012). Species that are favored by fire dominate (70 percent) over species that are from non-pyrogenic ecosystems or that do not respond favorably to fire (30 percent). Table 2 lists the plant remains found in the intestines of an elderly woman whose remains were exposed in the drip line of a rockshelter. All identified plant remains of this person’s last meal were fire-adapted (100 percent) species. Ants, fruit beetles, and mites were found in the unidentified fruity substances. The ingestion of charcoal and sumac (*Rhus glabra*) suggests that the woman may have been ill, since sumac was used for food as well as medicine to stop vomiting (Hamel and Chiltosky 1975). Food residues from a ceramic vessel at the open air Copperhead Site (AD 819-1299), 3CW951 in the Atoka Hills and Valleys LTA (Cummings and Yost 2009), indicate cooking of squash/pumpkin (*Curcubita pepo*), sunflower (*Helianthus* spp.), oak acorns, beans (*Phaseolus* sp.), and sumac berries (*Rhus* spp.). All of these species (100 percent) were fire-adapted. Table 3 lists the plant remains from three rockshelters on Lee Creek in western Arkansas identified by Fritz (1986a).

Table 2.—Listing of materials recovered from the feces of an elderly prehistoric woman indicating consumption of fruits with pests, oak acorns, and sumac (from Wakefield and Dellinger 1936). At least half of the remains provide evidence of fire use or fire-adapted habitats. Quantitative data were not provided. Species that respond favorably to fire are denoted by 1, and species that are fire intolerant are denoted by 0, however note that many species respond favorably to environments cleared by fire.

Scientific Name	Type of Material	Fire Code	Quantity
Rhus glabra	sumac	1	no data
Quercus velutina	black oak	1	no data
Nitidulidae	fruit beetles		2 specimens
Tyroglyphidae	mite in decaying matter		a number
Formicidae	ant		1 specimen
- - - - -	fruit substance		
- - - - -	charcoal	1	mixed throughout
	fungus		growth on materials

Table 3.—Plant list from three rockshelters in Lee Creek (Fritz 1986b). Quantitative data were not provided, but are inferred by site catalog numbers provided in Fritz 1986a. Species that respond favorably to fire are denoted by 1, and species that are fire intolerant are denoted by 0; however, note that many species respond favorably to environments cleared by fire.

Scientific Name	Common Name	Fire Code	Quantitation Information
Zea mays	Indian corn	1	37 specimens
Cucurbita maxima	winter squash	1	38 specimens
Cucurbita ovifera	egg gourd	1	12 specimens
Cucurbita pepo	pumpkin/summer squash	1	lumped with winter squash
Laegenaria	bottle gourd	1	12 specimens
Phaselous vulgaris	garden bean	1	11 specimens
Phaselous polystachios	wild bean	1	1 specimen
Strophostyles helvula	wild bean	1	1 specimen
Helianthus sp.	sunflower	1	16 specimens
Helianthus tuberosus	“Jerusalem artichoke”	1	lumped with sunflower
Iva annua	sumpweed\polygonum	1	4 achenes
Phalaris caroliniana	maygrass	1	1 specimen
Hordeum pusillum	little barley	1	
Amaranthus sp.	amaranth	1	1 specimen
Ambrosia trifida	ragweed	1	1 specimen
Nuphar luteum	spatterdock/water lily	0	1 specimen
Optunia sp.	cactus	0	3 specimens
Citrullus lanatus	watermelon	0	3 specimens
Juglans nigra	walnut	0	11 specimens
Carya spp.	hickory	1	8 specimens
Quercus spp.	oak	1	19 specimens
Fagus grandifolia	beech	0	1 specimen
Prunus spp.	plum	1	10 specimens
Prunus virginiana	chokecherry	1	lumped with plum
Diospyros virginiana	persimmon	1	4 specimens
Platanus occidentalis	sycamore	0	2 specimens
Ostrya virginiana	ironwood (hophornbeam)	0	3 specimens
Acer sp.	maple	0	1 specimen

(Table 3 continued on next page)

Table 3 (continued).—Plant list from three rockshelters in Lee Creek (Fritz 1986b). Quantitative data were not provided, but are inferred by site catalog numbers provided in Fritz 1986a. Species that respond favorably to fire are denoted by 1, and species that are fire intolerant are denoted by 0; however, note that many species respond favorably to environments cleared by fire.

Scientific Name	Common Name	Fire Code	Quantitation Information
Nyassa sylvatica	sourgum	0	2 specimens
Compositae spp.	(*Asteraceae* sp.)	1	1 specimen
Arundinaria sp.	Cane	1	15 specimens
Juniperus sp.	Juniper	0	9 specimens
Pinus spp.	Pine	1	5 specimens
Vitis spp.	Grape	1	4 specimens
Asimina triloba	pawpaw	0	1 specimen
Vaccinium spp.	Blueberry	1	1 specimen
Rubus spp.	blackberry/bramble	1	2 specimens
Celtis occidentalis	hackberry	1	8 specimens
Viburnum sp.	Haw	1	4 specimens
Viburnum prunifolium	black haw	1	lumped with viburnum
Crataegus spp.	Hawthorn	1	1 specimen
Rhus spp.	Sumac	1	1 specimen

Here, species that are favored by fire comprise 76.2 percent of the plants recovered while fire sensitive plants comprise 23.8 percent of the assemblage. Primary occupations date to the Late Archaic-Woodland periods (3000-1000 YBP) for these species. Table 4 lists nine plants identified by Fritz (1997) that were recovered from a cache (radiocarbon dated 1301-898 BC, Fritz 1986b) in the Marble Falls Rockshelter on Mill Creek in north-central Arkansas. Fritz interprets that four species of wild plants (*Curcurbita pepo* spp. *ovifera*, *Chenopodium berlanderi* spp. *jonesianum*, *Iva annua* var. *macrocarpa*, and *Helianthus annus* var. *macrocarpus*) were domesticated and, along with *Ambrosia trifida*, were harvested from a garden and stored to be planted or eaten the following spring. All of these species (100 percent) are fire-adapted.

Table 4.—Listing of domesticated wild plants from seed cache at Marble Bluff (Fritz 1997). Species that respond favorably to fire are denoted by 1, and species that are fire intolerant are denoted by 0; however, note that many species respond favorably to environments cleared by fire.

Scientific Name	Common Name	Fire Code	Quantity
Chenopodium berlandieri	Lambsquarters, goosefoot	1	
Ambrosia trifida	giant ragweed (stored in bags)	1	
Lamiaceae	mint	1	capsules
Helianthus	sunflower	1	58 seeds
Amaranthus	amaranth	1	18 achenes
Polygonum	knotweed	1	
Curcubita pepo	gourd/squash	1	8 seeds
Iva annua	sumpweed	1	
Phalaris caroliniana	canary grass	1	least common

Other indicators suggesting Native American ecosystem modifications include the presence of large herbivores such as bison and elk in Arkansas (Collins 1992, Nuttall 1821, Schroeder 1981). Bison and elk remains are known from rockshelter and open site deposits in the Ozark Highlands (Cleland 1965, Sabo and others 1990, Scott 1993, Trubowitz 1980). Six bison images painted by Native Americans are present in the dark zone of Gustafson Cave (3ST70) on a tributary of the White River on the eastern front of the Ozarks in the Sylamore Hills LTA. Three images have fletched arrows extending through the torso of the animal, matching ethnohistorical evidence (Swanton 1942) of the use of bois d'arc (*Maclura pomifera*) for bows by the Caddo Indians that shot arrows through the animal. Schambach (2000, 2001b) argues that the Spiroans of the Arkansas Valley near Fort Smith were key participants in interregional exchange during the last 1500 years. Bow wood from the bois d'arc collected in northern Texas (Jurney 1994) was exchanged for seed corn (stored in specialized ceramic vessels) and turquoise (Jurney and Young 1995) from the Pueblos of the Southwest and bison meat and hides from the Plains nomads (Schambach 2001b). These goods were then routed into the Mississippi Valley, and thence to the southeastern Gulf region. Bois d'arc has been found under Caddoan mounds in Louisiana, at the Lyons Bluff site in Mississippi, and at Fort Toulouse in Alabama (Jurney 1994), all areas where it was not a major constituent of the original plant landscape after the terminal Pleistocene. The Gustafson Cave rock art also is the first written record (ca. AD 200-900) of bison in Arkansas and suggests some form of human manipulation of this species. During the middle 1700s, Le Page Du Pratz (Du Pratz 1774) observed that the bottomland forest and canebrakes at the mouth of the St. Francis River (the St. Francis National Forest) were "always covered with herds of buffalo, notwithstanding they are hunted every winter in those parts" (see Key 2001). Among the Quapaw Indians and throughout Arkansas, these prairies were maintained by fire, "attracting grazers like bison" (Key 2001). In northwest Arkansas, historical accounts of bison in 1829 estimated one head for every 3 acre density, with the last bison observed in 1832 (Miller 1972). Patch burning of grasslands has shown that, given the choice, bison will only graze in burned areas with incipient revegetation (Kerby and others 2007), a behavioral response which was undoubtedly noted by Native Americans.

There was a heavy dependence on oak acorns found at all rockshelter sites (Harrington 1960) in the Ozarks and adjacent areas. The relative abundance of white oak and red oak (Jurney and Stahle 2004) mirrors that of the 1815-1840 reference conditions found in GLO surveys. Fritz (1986b) found that oak (*Quercus* spp.) acorns were the most abundant plant remains from the Lee Creek Valley, a tributary of the Arkansas River in the Lee Creek Atoka Hills LTA. These remains were apparently associated with Caddoan-affiliated farmer-hunters and their Fourche Maline (Woodland, 350 BC-AD 950) predecessors (Gilliland and others 2009; Schambach 2001a; Trubowitz 1980, 1983). In the Lee Creek area, Hilliard (1986) found dominance (67 percent) of red/black oak species (*Quercus shumardii*, *Q. veluntina*, and *Q.rubra*) in archeological sites, with high frequencies of white oak species (*Q. alba*, *Q. stellata*) growing in the same area. General Land Office records dating to the early 19th century indicate an opposite trend, with 62 percent white oak group species and 38 percent red/black oak species (Foti and others 2007). Hilliard hypothesized the dominance of red/black oak species was due to human selection. Since red/black oaks germinate in spring and white oaks in the fall, white oaks are preferred by wildlife competitors, and are more susceptible to insect infestation. All oak acorns found in archaeological storage contexts had been scorched or parched to prevent insect infestation and germinated before the acorns were consumed. This same phenomenon could occur if Native Americans burned the woods during the fall, halting insect entry and allowing humans to more easily find acorns in forest litter; thereby out-competing wildlife such as turkey, deer, blue jay, squirrel, and passenger pigeon (Ellsworth and McComb 2003, James and Neal 1986) that feed on acorns. If humans did not collect acorns soon after they fell, competing wildlife, insects, molds, and unfettered germination would greatly reduce the amounts available for human consumption. Such adaptive use of fire is known to be used by the Cherokee Indians who collected large quantities of chestnuts (*Castanea* spp.) in this manner (Fowler and Konopik 2007). Life history studies of oaks point to fire adaptations with oak stand development dependent on advanced regeneration coupled with sprout-generated cohorts (Guyette and Dey 2004, Guyette and others 2004). It is possible that Native American agroforestry practices were based on similar observations.

Sabo and others (2004) estimate that an average community of 20 during the Archaic (7550-600 BC) would require 5 acres of openings in forest canopy for camps per 4.7 square mile catchment across the Ozarks (see also Lockhart and others 1995). By using these data, approximately 7,960 people occupied over 2,000 acres of camps on the Ozark National Forest (1.2 million acres). By the Woodland transition (600 BC-AD 600), with the use of adopted horticulture of native plants, Sabo and others (2004) estimate that an average community of 250 would require 62.5 acres of garden space per 4.7 square mile catchment. Using these data, an Ozark NF-wide population of 99,734 people farmed over 25,000 acres of gardens. With the development of sedentary agriculture ca. AD 600-1500, Sabo and others (2004) estimate an average community of 400 would require 125 acres of garden space per 4.7 square mile catchment. Using these data, a population of 158,200 people farmed over 376,000 acres of gardens (Fritz 1994) on the Ozark National Forest. However, these estimates do not include nut, animal, and plant food collecting across the landscape that may have included the broadcast use of fire. An estimated population of 2.26 people/square mile was present on the Ozark National Forest during the late prehistoric/protohistoric periods ca. AD 1500-1700 (Sabo and others 2004). During the historic periods, land was purchased in up to 160 acre grants by 40 ac parcels at $1.25/acre (Rafferty 2001), and the nature of the rectangular land surveys (AD 1815-1840) greatly affected the size and shape of farmstead allotments. Rafferty (2001) presents population data that indicates by 1930, the interior highlands of the Ozark-St. Francis National Forests (OSNF) were settled by 6-8 people/square mile, the lower Atoka Hills and Arkansas Valley by 18-45 people/square mi, and the St. Francis/Mississippi valley by 45-90 people/square mi. Using these data, in the uplands of the OSNF approximately 15,000 people were present, the Atoka Hills and Arkansas Valley had approximately 84,375 people, and the smaller St. Francis National Forest had 2812 people. Thus, during the early 20th century, from 8-20 times more people were present on the Forests than during the late prehistoric/protohistoric periods at or near the time of European contact.

Human use of fire is now receiving attention through the reconstruction of fire regimes by using dendrochronology for the study of fire-scarred trees. Clark and others (2007) found that the mean fire return interval for all fires from AD 1772-2002 was 2.0 years and 7.0 for moderate-scale (more than 25 percent of sampled trees have that fire-scar year) fires in the Cross Timbers of Oklahoma. Low intensity fires probably were not recorded as fire scars on oak trees unless there was a pre-existing wound on the tree (Clark and others 2007). This fire return interval decreased following Euro-American settlement of the area, probably due to heavy grazing practices (Lane 1939). Oak recruitment was also found to increase with shorter fire return intervals (Clark and others 2007). In the lower Boston Mountains of the Ozarks, Guyette and Spetich (2003) found that between the years 1670-2000, the mean fire return interval ranged from 4.6 years (1680-1820), to 3.1 years (1821-1880), to over 80 years (1921-2000). This trend was opposite that of Clark and others (2007) until 1921 and may be due to sampling vagaries, such as temporal factors and number of long-lived trees, and landscape differences, such as topography, slope, aspect, and local climate variability. Fire scar records indicate that between 1748-1881 (excluding those low intensity fires that may not have left tree-ring evidence), an area about the size of the Ozarks burned every 20 years under moderate to severe drought conditions (Guyette and others 2006, Spetich and He 2008), producing a mosaic of burned and non-burned areas due to topographic roughness, fuels variability, and localized conditions (Collins 1992, Frost 1995, Frost and others 1986).

PREHISTORIC SITE DISTRIBUTIONS

Table 5 lists the Landtype Associations (Fig. 3) on the Ozark-St. Francis National Forests with the number of prehistoric and historic components in each. Examination of the topographic distribution (Fig. 4) of historic and prehistoric sites across all Landtype Associations on the Ozark-St. Francis National Forests reveals that relatively more prehistoric sites are found in the Floodplain, South Aspect Lower Slope, and South Aspect Middle Slope landtypes than historic sites. The south aspect is the most pyrogenic landtype in the Ozark Highlands. Relative frequencies are nearly identical in the North Aspect Upper Slope and North Aspect Lower Slope landtypes. Historic sites exceed prehistoric sites in the Ridge and Flat Upper Slope landtypes.

In addition to the frequency distribution discussed above, a component density value is calculated for each by dividing the various acreages surveyed within each LTA by the number of sites discovered. Five LTAs have the densest number of both prehistoric and historic sites; Boone Mountain Valleys, Lee Creek Limestone Valleys, White River Rolling Hills, Lee Creek Atoka Hills, and Mesic Limestone Mountain Valleys.

Table 5.—Landtype Associations (LTAs) and Prehistoric and Historic Archeological Site Component Density Values (CDV) on the Ozark-St. Francis National Forests, Arkansas.

LTA	LTA Name	Acreage	Pre CDV	Hist CDV
0	Private	8695.57	414.1	44.4
1	Boone Mtn. Valleys	739.15	369.6	24.6
2	Mesic Limest. Mtn. Valleys	5934.4	197.8	131.9
3	Mesic Morrow Mtn. Uplands	50935.72	134.4	125.5
4	Mesic Atoka Mtn. Uplands	37909.24	972	170.8
5	Hale Mtn. Valleys	32208.34	1073.6	187.3
6	Bloyd Mtn. Valleys	84279.92	726.6	233.5
7	Atoka Mulberry Mtn. Valleys	55574.1	805.4	243.7
8	Lower Atoka Hills & Mtns.	85770	779.7	196.3
9	Arkansas Valley Hills	12628.76	451	157.9
10	Lee Creek Rolling Hills	4036.12	807.2	36.4
11	Lee Creek Atoka Hills	18555.78	343.6	51.1
12	Lee Creek Limest. Valleys	2641.99	188.7	41.9
13	Wedington Calc. Uplands	1590.72	113.6	56.8
14	Wedington Boone Hills	12663.6	107.3	58.9
15	Sylamore Boone Hills	21256.27	250.1	110.7
16	Sylamore Hills	10904.09	340.7	139.8
17	White R. Rolling Hills	19649.74	194.6	99.2
18	Magazine Rolling Valley Hills	42387.45	399.9	129.2
19	Magazine Valley Ridge	13612.08	1944.6	349
20	Magazine Valley Hills	41403.76	828.1	221.4
21	Magazine Mtn. Uplands	5969.06	663.2	170.5
22	Mississippi Bottomlands	10539.3	619.9	107.5
23	Crowley’s Ridge	10350.8	2587.7	145.8
24	Bear/Storm Ck. Lakes	761.69	0	152.3

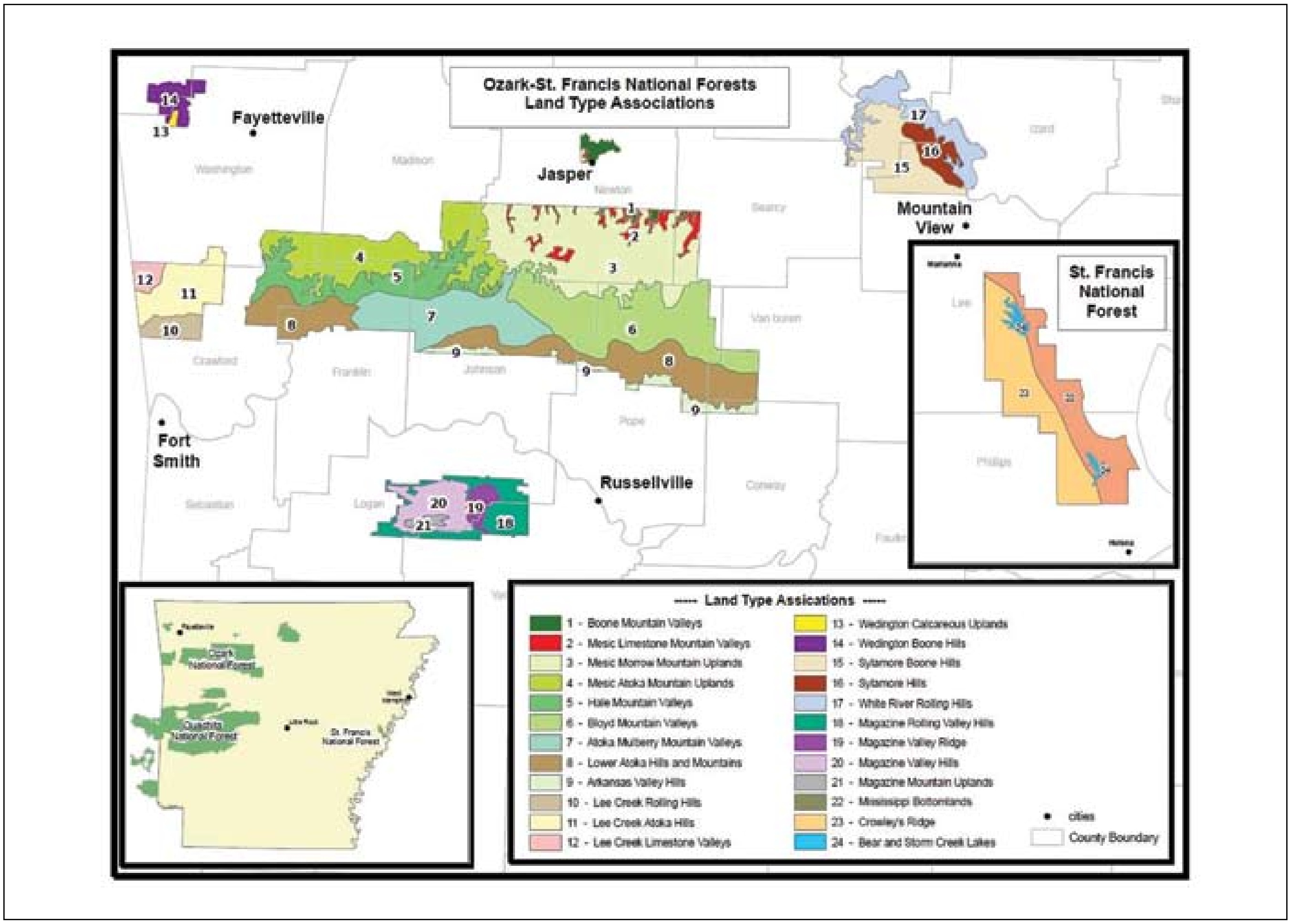

Figure 3.—Ozark-St. Francis National Forests Land Type Associations.

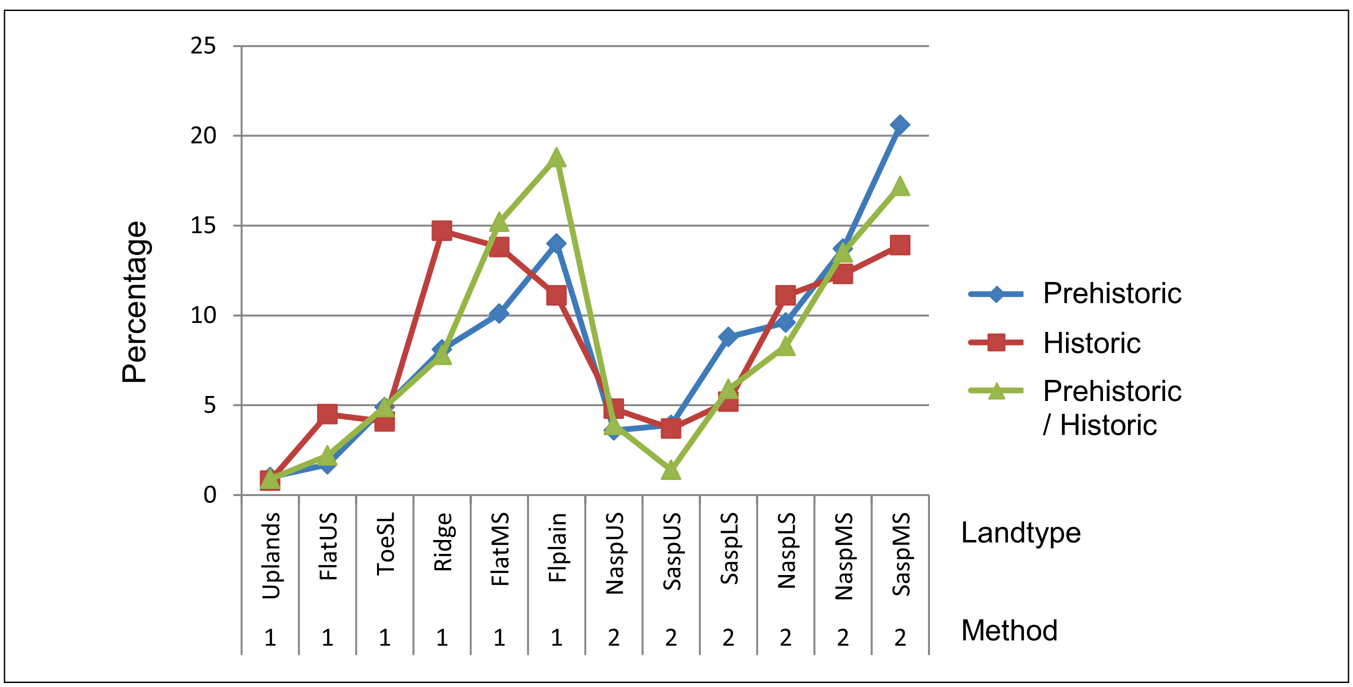

Figure 4.—Percentages of sites among landtypes on the Ozark-St. Francis National Forests. Landtype codes are: US = upper slope, ToeSL = toe slope, LS = lower slope, MS = mid slope, Flplain = Floodplain, Nasp = north aspect, Sasp = south aspect. Method: 1 = shovel probing, 2 = walkover survey.

The five LTAs with the densest prehistoric sites are Wedington Boone Hills, Wedington Calcareous Uplands, Mesic Morrow Mountain Uplands, Lee Creek Limestone Valleys, and White River Rolling Hills. This greater density correlates with the local prevalence of naturally occurring chert (Burlington/Keokuk/Cotter, Reeds Spring, and Pitkin) for manufacture of stone tools within or immediately adjacent to all of these LTAs. As noted earlier, the Wedington unit has extensive prairies dervied from the field notes and maps of the GLO surveys, with some prairies located on soils that developed under hardwood forest. Thus the LTAs with the highest density of sites are also the ones that have the most evidence of fire. The five least dense LTAs with prehistoric sites include Crowley's Ridge, Magazine Valley Ridge, Hale Mountain Valleys, Mesic Atoka Mountain Uplands, and Magazine Valley Hills. These areas lack local stone suitable for tool manufacture and lack readily available or permanent water sources. These areas also have the least evidence of fires, derived from the GLO field notes and maps.

The five LTAs with densest historic sites are Boone Mountain Valleys, Lee Creek Rolling Hills, Lee Creek Limestone Valleys, Lee Creek Atoka Hills, and Wedington Calcareous Uplands. The greater density correlates with more open woodlands and prairies, as well as dependable water flow in local streams and spring recharge rates. The five LTAs with the least dense historic sites include Magazine Valley Ridge, Atoka Mulberry Mountain Valleys, Boyd Mountain Valleys, Magazine Valley Hills, and Lower Atoka Hills and Mountains. These are the least arable LTAs with steep slopes, cobble-laden stream heads, and remoteness from established transportation routes.

CONCLUSIONS

Fire is one of the oldest tools known to man and his relatives, dated in archeological contexts for *Homo erectus* to at least 780,000 years ago at Zhoukoudian Cave, ca. 48 km southwest of Beijing in China (Jia and Huang 1990). The contextual associations of man and fire at Zhoukoudian Cave are in dispute, with some insisting that lightning could have caused archaeological evidence for fire, therefore absolute proof of early man's deliberate manipulation of fire is still in question (James 1989).

The widespread use of nut (acorns, hickory, chinquapin/chestnut, walnut) resources in the eastern Woodlands in general, and the Ozark Highlands specifically, is evident in the archaeological records, written accounts, and land surveys. Mature oaks and hickories yield more mast than younger trees, and the seasonal use of fire to collect and preserve mast in quantities of chestnut suitable to feed large populations is known among the Cherokees of the southern Appalachians (Fowler and Konopik 2007). Based on the archeological evidence in rockshelters (Hilliard 1986), it is also highly likely seasonal fire was used by the Ozark Native Americans to collect and preserve oak as well. Fire was used to open grasslands and alter cover to increase nutrient cycling and also created the edge environment where prey could be concentrated. Canebrakes require periodic burning to maintain equilibrium and prevent floodplain canopy closure (Brantley and Platt 2001) and are an excellent example of high-yield, fire-adapted grasslands that are suitable for tools, fiber, food, and ceremonial use. Due to lack of fire in modern floodplain forests, cane is rapidly facing extirpation.

Although fire is a natural component of a disturbance regime in ecosystems of the Arkansas Ozarks, it appears that humans greatly extended the intensity and coverage of burns to collect and concentrate resources, beginning with hunter-gatherers and culminating with agriculturalists. This inference is based on the ethnohistorical evidence as well as direct archaeological evidence from dry rockshelter deposits, General Land Office field notes and maps, and dendrochronological evidence from fire-scarred trees for the Ozarks. It is now recognized that disequilibrium in ecological communities is more common than equilibrium (Pielou 1991). Healthy forests and ecosystems contain mosaics of stands at different ages and compositions, reflecting the vagaries of disturbances (Nelson 2005, Oliver and Larson 1996). Therefore, humans, and possibly ancestral humans, introduced disequilibrium into ecosystems with experiential knowledge designed to provide edge environments and maximize yields from hunting and gathering as well as incipient agroforestry.

The Native Americans of the Ozarks were historically seen as marginal participants in broader cultural developments. However, archaeological investigations have now covered a greater part of the region, and the marginality theme is now in question. During the earliest human occupations from ca. 18,000-6,000 YBP, population was low, but two important circumstances occurred that affected ecosystems: large herbivores became extinct, and prairies expanded from the Great Plains to the Ohio River. Several species of wild plants became domesticated ca. 4,200-3,000 YBP due to human manipulation, and intensive collection of other plants, including mast crops, indicate extensive agroforestry during this period.

The presence of Mesoamerican tropical plants such as maize (Sabo and Early 1990) and cotton (Horton 2008) in the Ozarks, and the distribution of large population centers indicate that ca. AD 1200-1500, an agricultural revolution did occur in the study area. The people of the Ozarks participated actively in regional exchanges and were not the conservative isolationists described in popular myth. Wild plant data (pollen, archaeological excavations, fire-scarred trees, and General Land Office records) from the Ozarks and Cumberland Plateau indicate genetic manipulation of a suite of species prior to the adoption of full-scale agriculture (Gremillion 1997). Native Americans in southeastern North America as early as 4000 BC had developed sophisticated knowledge of plants' life histories, and a form of agroforestry was developed to maximize the yield of oak (both regions) and chestnut (east of Mississippi River) for developing populations. In this paper, several types of proxy information are presented. However, three time capsules are noted where human consumption of fire-adapted species is clearly demonstrated. In one, the food items were taken from the intestines of a prehistoric woman. In another, food residues were identified from a single ceramic vessel. Finally, in the Marble Bluff seed cache, domesticated wild plants were clearly collected and stored for spring planting or consumption. In all three instances, all species are fire-adapted.

As a result of more intensive and extensive archaeological investigations, and the recordation of traditional Native American ecological knowledge, it is now recognized that most biotic communities are in a constant state of change due to natural and human-caused disturbances. In successional communities, it is necessary to manage the habitat to achieve stability or a desired stage of development (Anderson 2006). Fire use was the principal tool of Native Americans to achieve landscape scale modification of plant and animal communities.

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HISTORY OF FIRE IN EASTERN OAK FORESTS AND IMPLICATIONS FOR RESTORATION

Justin L. Hart and Megan L. Buchanan

Assistant Professor (JLH), Department of Geography, University of Alabama, Box 870322, Tuscaloosa, AL 35487. Graduate Student (MLB). JLH is corresponding author: to contact, call 205-348-4190 or email hart013@bama.ua.edu.

Abstract.—Our understanding of long-term fire history in the eastern United States is derived from the interpretation of a variety of archives. While cultural records are available for some sites, biological archives are most frequently used to reconstruct long-term historical fire regimes. The three most commonly used biological archives in eastern oak forests include: the relative abundance of micro- and macroscopic charcoal found in lake and wetland sediment cores, charcoal macrofossils recovered from mineral soil, and dated fire scars on tree cross-sections. Quantitative data from these biological records are essential to fully elucidate the relationships between fire and oak forest dynamics. In addition to providing a basis for the development and refinement of ecological theory, these data have practical utility as they can be used in restoration planning to set desired future conditions and establish silvicultural treatments that maintain oak dominance or mimic historical disturbance regimes. Here we review the three biological archives most commonly used to reconstruct historical fire regimes in the Central Hardwood Forest Region, synthesize results of investigations that have relied upon these techniques, and discuss the implications of these findings for restoration efforts. At present, ca. 100 fire reconstructions have been developed from fire scarred trees and soil and sedimentary charcoal in the region. Results from the reviewed published studies reveal that fire histories are site specific. Therefore, managers focused on ecological restoration are best advised to construct a place-based history rather than rely solely on results from other studies to set restoration targets and monitor treatment success.

INTRODUCTION

Over the past several decades the role of fire in oak (*Quercus*) forests throughout the eastern United States has received increased attention driven largely by successional changes hypothesized to be caused by 20th century human alteration of fire regimes (Abrams and Downs 1990, Cho and Boerner 1991, Goebel and Hix 1997, Lorimer 1993). In addition, concern that fuel loadings have exceeded the historic range of variability has generated interest in the history of fire in oak forests (Brose and others 2001, Graham and McCarthy 2006, Loucks and others 2008).

Vast areas of the eastern United States landscape are characterized by oak forest cover (Braun 1950, Dyer 2006). Paleoecological investigations indicate that oak has dominated forest communities of the region throughout much of the Holocene. However, a widespread pattern of forest composition change is evident throughout eastern oak forests. Oak regeneration failure has been reported from oak-dominated stands over a variety of site types. Coupled with this regeneration failure is an increase in the density and dominance of mesic species, particularly sugar maple (*Acer saccharum* Marsh.) and red maple (*Acer rubrum* L.) (Abrams 1998, Fei and Steiner 2009, Lorimer 1984).

The profusion of quantitative data reported from sites throughout the eastern United States has led many researchers to project a pervasive and inevitable transition from oak systems to those dominated by maple and other mesic taxa (Abrams and Nowacki 2008, Nowacki and Abrams 2008). This change in species dominance will undoubtedly have major ramifications for biodiversity, wildlife population densities, timber production, and a host of ecosystem processes such as forest hydrology, nutrient cycling, and fuel loading (Alexander and Arthur 2010, McShea and others 2007, Nowacki and Abrams 2008). Active fire suppression that began in the early 20th century is the most often cited explanation for the oak replacement pattern. While variability exists at the species level, oaks are considered tolerant of fire and only moderately tolerant of shade. Adaptations to fire include thick bark, the ability to stump sprout, and resistance to rot after scarring (Abrams 1992, Smith and Sutherland 1999). In contrast, maple and other mesic taxa have morphological characteristics such as thin bark and shallow rooting that make them fire-sensitive. Therefore, it is hypothesized that historic surface fires maintained oak dominance by removing more mesophytic, shade-tolerant, and fire-sensitive competition from the understory (Abrams 1992, Lorimer 2001, Nowacki and Abrams 2008). While alternative hypotheses have been proposed to explain this successional shift (e.g., climate change, alterations in land use, facilitative processes, extirpation of American chestnut (*Castanea dentata* [Marsh.] Borkh.), and changes in wildlife population densities (Hart and others 2008b, Lorimer 1993, McEwan and others 2011), the oak-fire hypothesis is undoubtedly the dominant paradigm. As such, quantification of historical fire regimes of oak ecosystems is essential.

Our understanding of the history of fire in eastern oak systems is derived from the interpretation of a variety of archives. To reconstruct fire history, researchers have relied upon witness tree analyses, field notes from land surveyors, early explorer and European settler accounts, and other land-use records (Ruffner 2006, Russell 1983, Whitney 1994). While documentary or cultural archives such as these are available for some sites, reconstructions of long-term historical fire regimes are typically developed using biological archives. Biological records used to document past fire events include dendrochronology or fire scar analysis; charcoal analysis of lake sediments, wetlands, or peat bogs; black carbon analysis of marine sediments; pedanthracology or macroscopic charcoal analysis in mineral soil; molecular markers of combustion; fuel and soil magnetism; and sedimentology (Conedera and others 2009). The three most commonly used biological archives of fire history in eastern oak forests include: the relative abundance or influx of micro- and macroscopic charcoal found in lake and wetland sediment cores, charcoal macrofossils recovered from mineral soil, and dated fire scars on tree cross-sections. Quantitative data from these biological records are essential to fully elucidate the relationships between fire and oak forest dynamics. In addition to providing a basis for the development and refinement of ecological theory, these data have practical utility as they can be used in restoration planning to set desired future conditions and establish silvicultural treatments that maintain oak dominance or mimic historical disturbance regimes.

The goal of this paper is to review the three biological archives most commonly used to reconstruct historical fire regimes, synthesize results of investigations that have relied upon these techniques, and discuss the implications of these findings for restoration efforts. We limited our review to the Central Hardwood Forest Region (CHFR) because a formal eastern oak forest unit is not recognized, the CHFR is a well established spatial unit, and oak is the dominant genus of the region. Several CHFR boundaries are accepted and we chose those defined by Fralish and Franklin (2002) and Fralish (2003).

BACKGROUND ON BIOLOGICAL FIRE RECONSTRUCTION TECHNIQUES

Sedimentary Charcoal Analysis

Charcoal is produced during fire events as organic material is partially combusted. This combustion results in black pyrogenic carbon ranging from soot and graphite particles to coarse charcoal fragments and charred wood (Conedera and others 2009, Ohlson and others 2009). Microscopic (ca. 10-200 µm length) and macroscopic (ca. >100-200 µm length) charcoal in sediment cores retrieved from lakes, wetlands, and peat bogs may be used to reconstruct historical fire characteristics (e.g., Delcourt and Delcourt 1997, 1998; Delcourt and others 1998). In these paleoecological analyses, the relative abundance or influx of charcoal is used to assess fire frequency and/or magnitude. The primary advantage of this biological archive is the depth of record, as it is possible to document variability in vegetation composition and fire spanning the Holocene (Clark and others 1996, Delcourt and others 1998). These records reveal periods, rather than dates, when fire was more or less common based on the relative position and abundance of the quantified charcoal in the varved (i.e., layered) sediment core. Consequently, temporal resolution is coarse relative to macroscopic soil charcoal or fire scar analyses. In fact, the temporal resolution of this biological archive exceeds the hypothesized return interval of fire in many oak forests. In addition, sedimentary charcoal records are not spatially explicit as lakes receive charcoal inputs from broad source areas (Clark 1988, Clark and Royall 1996). Calibration studies have demonstrated that macroscopic charcoal particles typically originate within a few hundred meters of deposition sites while microcharcoal may originate up to 100 km from the deposition site (Clark 1989, 1990; Clark and others 1998; Patterson and others 1987). Thus, documented microcharcoal could in fact have originated in a non-oak dominated stand kilometers from the study site and even macrocharcoal could originate from fire in xeric, pine (*Pinus*)-dominated stands that may not be representative of local forest composition. Furthermore, fires in oak forests are typically low intensity burns, and the quantity of charcoal produced from such events may be negligible and undetectable in the sedimentary record (Abrams and Seischab 1997). Therefore, sediment cores from lakes and wetlands may not provide accurate records of fire frequency and be biased towards the documentation of intense and/or high magnitude events. Nonetheless, these records are useful to understand long-term patterns of oak forest composition and the relative importance of fire in these systems.

Soil Charcoal Analysis

Macroscopic (generally considered >2 mm length) charcoal fragments recovered from soil cores provide historical fire data at fine spatial resolutions. Macroscopic charcoal particles are sufficiently large to resist entrainment by wind during or after fires and by overland flow on hillslopes. These macrofossils are considered primary charcoal and provide evidence of historical fire at the stand-scale (Gavin and others 2003, Hart and others 2008a, Talon and others 2005) though even larger pieces of charcoal are likely formed in situ and indicate fire at the exact location of the soil core sample (Gavin and others 2003, Ohlson and Tryterud 2000). In addition to fine spatial resolution, macroscopic soil charcoal provides long-term fire records. Mean residence time of macroscopic charcoal varies by geographic location, but charcoal may be preserved in mineral soils of the eastern United States for up to ca. 10,000 years (Fesenmyer and Christensen 2010, Hart and others 2008a). Therefore, charcoal macrofossils may provide fire history records at the stand-scale throughout the Holocene. While the utility has not been fully explored in eastern oak systems, charcoal macrofossils can be identified to species or genera providing information on taxa that inhabited stands that were disturbed by fire. The major limitations to this biological archive include the inbuilt age error (Gavin 2001) associated with accelerator mass spectrometry (AMS) ^{14}C dating of the charcoal macrofossils. This technique actually provides the date that carbon was assimilated by the plant rather than the time of the fire event. This dating analysis is also cost prohibitive, and many samples are required to develop robust fire histories. Macroscopic soil charcoal analysis may indeed be the best method to reconstruct long-term fire histories in mesic oak stands. However, more methodological studies in the eastern United States are warranted (e.g., inbuilt age error estimates by species and site type, charcoal loss during subsequent fire, and charcoal transport by size and site condition).

Fire Scar Analysis

Forest fire histories are often reconstructed by assigning the calendar year and often the season of formation to fire scars found on tree cross-sections (Fritts and Swetnam 1989, Kipfmueller and Swetnam 2001). Fire scar analysis provides the finest spatial and temporal resolution of the three biological archives most commonly used. As trees are sessile, the spatial resolution is known to the exact location of the sampled individual (Kipfmueller and Swetnam 2001, Swetnam and others 1999). This allows for reconstruction of the spatial extent of past fires. Along with forest composition and age structure data, the magnitude of the historical disturbances can also be documented which provides information on the role of fire in forest community organization. Annual resolution allows researchers to analyze historic fires with regard to short-term forcing factors that may influence fire characteristics (e.g., contemporaneous and previous climate characteristics, land-use change). Furthermore, intra-annual resolution allows for the documentation of fire seasonality, shifts in which may provide information on ignition sources (Lafon 2010). Though fire scar analysis provides annual or intra-annual resolution of fire data at fine spatial resolution, this line of evidence is inherently limited by the occurrence of old trees. Thus, the temporal depth of record is constrained by the age of the trees or remnant wood on the site. In the eastern United States, this archive provides records typically extending to a maximum of 400 years and often much less. Fire scar analyses are further constrained by tree selection and sample extraction. Long fire-free periods allow for wounds from prior fires to heal, and damaged trees may, therefore, not reveal external diagnostic characteristics of the records contained within. In addition, tree morphology (e.g., thick bark) may prevent scarring from low intensity surface fires (Guyette and others 2006a, McEwan and others 2007a, Smith and Sutherland 1999), resulting in an underrepresentation of these fires in the reconstructed dataset. Similarly, stand-replacing fires are also likely to be underrepresented in historical fire datasets as these events remove wood thereby destroying direct dendroecological evidence of the disturbance (Guyette and others 2006a). Fire scar analysis typically requires that complete cross-sections or partial wedges are removed from trees (Arno and Sneck 1977, Baisan and Swetnam 1990). While logs and standing dead trees can be sampled, attaining an appropriate sample depth may require the partial destruction of living trees. Even if only partial wedges are collected, this sampling scheme is restricted on many sites because it affects the structural integrity of stems and causes the trees to be more susceptible to pathogens.

GENERAL TRENDS

Sedimentary Charcoal Analysis

In the CHFR, fewer than 10 published studies have used charcoal analyzed from sediment cores to discern information on the long-term patterns of vegetation and fire (Table 1); however, several important sedimentary charcoal studies (e.g., Clark and others 1996, Clark and Royall 1996) have been conducted just outside the bounds of the CHFR. Collectively, studies from within the CHFR have

Table 1.—Descriptive data for all sedimentary charcoal sites from published studies in the Central Hardwood Forest Region

Reference	State	Length of record
Delcourt and others 1998	KY	9,500 YBP
Delcourt and Delcourt 1997	NC	3,900 YBP
Cridlebaugh 1984	TN	900 YBP
Cridlebaugh 1984	TN	2,800 YBP
Haas 2008	TN	425 YBP
Haas 2008	TN	2,800 YBP
Kneller and Peteet 1999	VA	17,345 YBP
Kneller and Peteet 1993	VA	17,130 YBP
White 2007	WV	8,180 YBP

reported nine sedimentary charcoal records from six different sites (multiple studies were conducted at the same site). The sedimentary charcoal studies for the CHFR are clustered in the central and southern Appalachian Highlands (Fig. 1). The longest record from this archive extends over 17,000 years before present (YBP) and shows an increase in charcoal abundance coincident with the rise of the oak-hickory (*Carya*) forest type in the central Appalachians at the beginning of the Holocene (Kneller and Peteet 1999). This marks the beginning of the Holocene and rapid climate change and does not provide that increased fire frequency was a causal factor.

No clear ubiquitous patterns were evident as the results from the studies revealed unique fire histories based on the abundance of charcoal in the sedimentary record. The results emphasize that fire history is region, landscape, and even site specific. Nonetheless, useful information on the relationships between humans, fire, and vegetation was gleaned from these studies. In all these paleoecological investigations, charcoal recovered from sediment cores exceeded the amounts the authors hypothesized would be produced by natural ignitions, leading them to speculate that most of the fires were from human ignitions. Working from this assumption, human use of fire generally

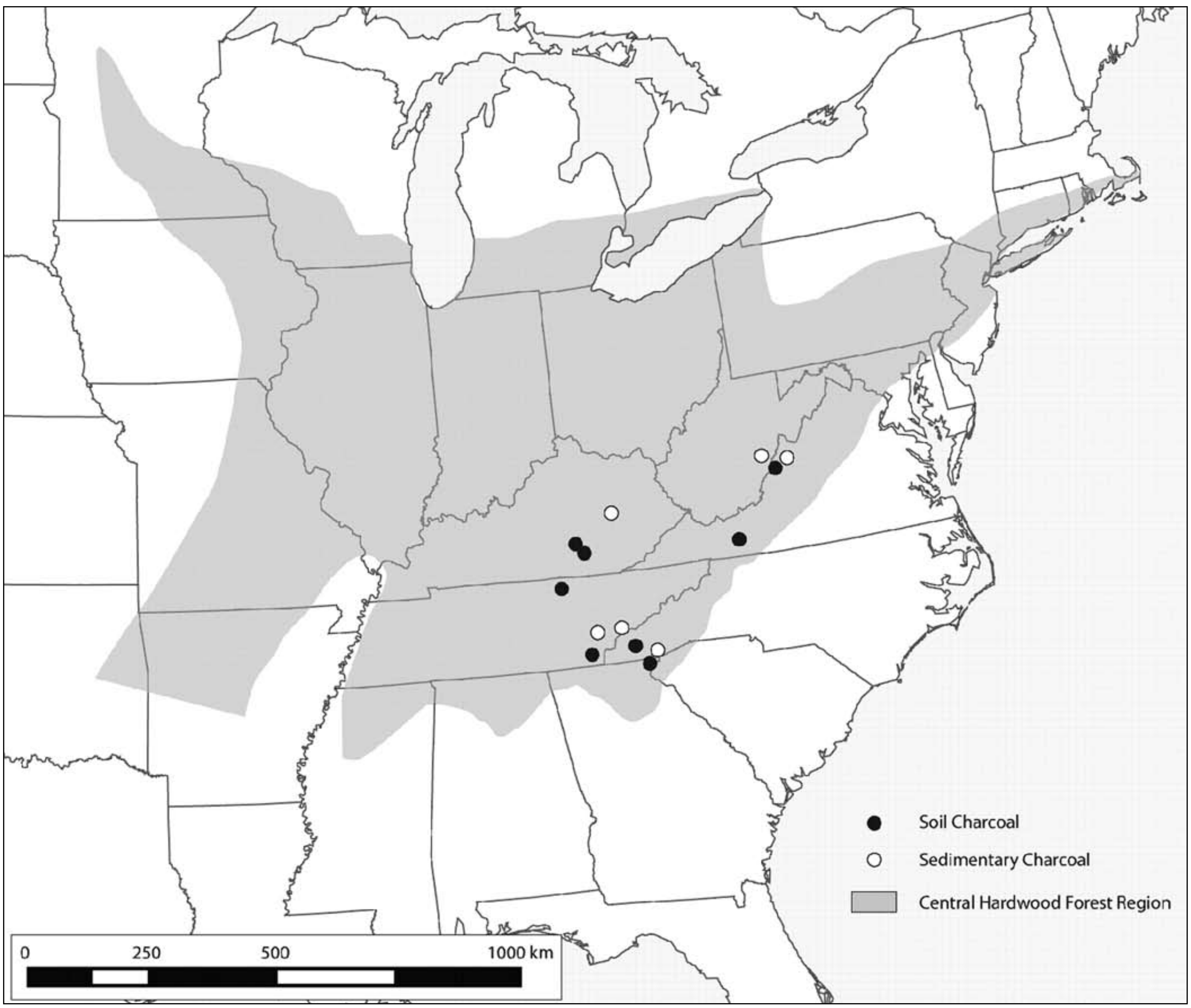

Figure 1.—Map of the Central Hardwood Forest Region (Fralish 2003, Fralish and Franklin 2002) and the location of sites with fire reconstructions based on analysis of soil and sedimentary charcoal.

increased during the Woodland cultural period (ca. 3,000 YBP) (Delcourt and Delcourt 1998). The abundance of charcoal from a site in the Ridge and Valley of east Tennessee peaked at the transition from Woodland to Mississippian periods (ca. 1,000 YBP) and from Mississippian to historic periods (ca. 300 YBP) (Delcourt and Delcourt 1998). Delcourt and Delcourt (1997) hypothesized that fires, at least of the southern Appalachian Highlands, were largely restricted to ridgetops and upper slopes and that lower slope positions supporting more mesic species were relatively protected from Native-set fires prior to European settlement. Additionally, they speculated that human-set fires in the southern Appalachians represented intermediate-scale disturbances that emphasized ecotones, increased gamma diversity, and increased the abundance of oak species on upper slope positions. Aboriginal fires would have likely facilitated and maintained oak dominance (and that of other disturbance dependent taxa as well) on ridgetops and upper slopes. Even at a local scale, these anthropogenic disturbances may have resulted in a patchwork of forest types that included fire-adapted and fire-tolerant species at various stages of succession (Delcourt and Delcourt 1997, Delcourt and others 1998).

Soil Charcoal Analysis

To date, macroscopic charcoal recovered from soil samples has been used in three published studies to reconstruct fire history in the CHFR (Table 2). These three studies presented data from nine different sites located in the southern Appalachian Highlands (Fig. 1). However, only Hart and others (2008a) work exclusively in oak dominated stands. Dated fire events (from AMS ^{14}C analyses) were only reported in two of these studies (Fesenmyer and Christensen 2010, Hart and others 2008a). Collectively, these two studies reported 87 AMS dates from charcoal macrofossils recovered from soil samples; however, only Fesenmyer and Christensen (2010) had sufficient dating to provide meaningful information on changes in fire frequency for their study site. Nonetheless, each of these studies made contributions to our understanding of historic fire regimes and the use of this method in the CHFR.

Welch (1999) established that macroscopic charcoal could be recovered from mineral soil in the temperate region of North America, that charcoal was abundant in pine and mixed pine-oak dominated stands of the southern Appalachians, and that charcoal accumulation did not vary by slope position indicating its presence is evidence of local fire. Hart and others (2008a) were the first to use macroscopic soil charcoal in mineral soils to elucidate fire history information in oak stands. Their study established that charcoal could be used to reconstruct stand-scale fire history in mesic oak systems, that charcoal macrofossils can be preserved in mineral soils of the region for millennia, and that macroscopic charcoal fragments could be identified to document taxa of the region that previously

Table 2.—Descriptive data for all soil macrocharcoal sites from published studies in the Central Hardwood Forest Region

Reference	State	Length of record	Number of radiocarbon dates
Welch 1999	GA	na	0
Welch 1999	KY	na	0
Welch 1999	KY	na	0
Welch 1999	NC	na	0
Fesenmyer and Christensen 2010	NC	10,570 YBP[a]	82
Welch 1999	TN	na	0
Hart and others 2008a	TN	6,735 YBP	5
Welch 1999	VA	na	0
Welch 1999	VA	na	0

[a]Only a single date was older than 4,000 YBP.

inhabited sites that burned during previous fires. To date, Fesenmyer and Christensen (2010) published the most extensive macroscopic soil charcoal study in the CHFR. They provided that even protected microsites devoid of trees (e.g., near rock outcrops) still had sufficient mixing of soil to require AMS dating of charcoal (i.e., depth does not correspond to charcoal age), that differences in fire frequency could be detected with this archive between mesic and xeric sites, and that fire frequency increased ca. 4,000 YBP (during the late Archaic period) and increased drastically ca. 1,000 YBP for their study site in the Blue Ridge physiographic province. In fact, only one charcoal date was older than 4,000 YBP. This drastic increase in fire frequency at 1,000 YBP was attributed to the rise of the Mississippian cultural tradition.

Fire Scar Analysis

We reviewed fire scar-based data from more than 70 sites in the CHFR representing 35 different published studies (Fig 2, Table 3). Of all biologically derived fire histories in the region, over 80 percent have been developed using fire scarred trees. The longest tree ring based fire records begin in 1581 (located in

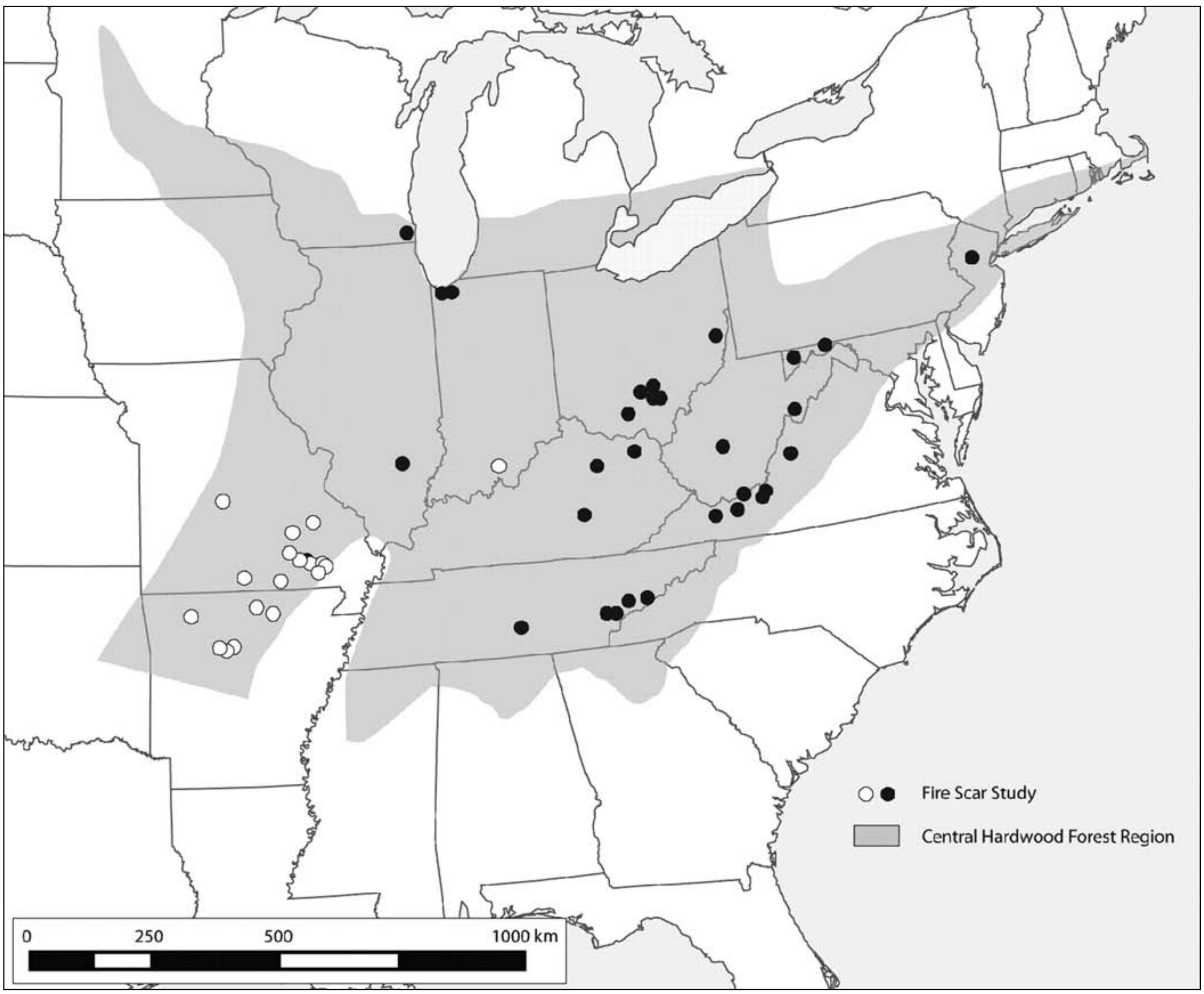

Figure 2.—Map of the Central Hardwood Forest Region (Fralish 2003, Fralish and Franklin 2002) and the location of sites with fire reconstructions based on dated fire scars in tree cross-sections. White circles represent temporally delineated fire history records that pre-date European settlement. Black circles represent all other fire scar records.

Missouri), but the mean initiation date for the total fire scar network is in the mid-1700s. Regional fire scar chronologies extend to the 17th century for 21 sites and to the 18th century for 50 sites, cumulatively. Approximately 35 percent of the fire scar records in the CHFR begin post-800. Fire frequency statistics of some form (e.g., mean fire return intervals) are available for the overwhelming majority of sites.

Table 3.—Descriptive data for fire scar based reconstruction sites from published studies in the Central Hardwood Forest Region. Only time periods with provided mean fire interval (MFI) values are reported. All MFI values are rounded to the nearest integer.

Reference	State	Period of record	MFI
Jenkins and others 1997	AR	1770-1993	6
Guyette and Spetich 2003	AR	1680-1820	5
		1821-1880	3
		1881-1920	5
Guyette and Spetich 2003	AR	1680-1820	16
		1821-1880	3
		1881-1920	2
Guyette and Spetich 2003	AR	1680-1820	13
		1821-1880	2
		1881-1920	1
Guyette and others 2006b	AR	1810-1830	2†
		1821-1880	2†
		1881-1920	2†
Guyette and others 2006b	AR	1810-1830	1†
		1821-1880	2†
		1881-1920	2†
Guyette and others 2006b	AR	1810-1830	2†
		1821-1880	4†
		1881-1920	5†
Stambaugh and Guyette 2006	AR	1670-1820	8
		1821-1880	2
		1881-1920	3
Engbring and others 2008	AR	1820-1900	2
		1901-1930	2
		1931-2003	3
McClain and others 2010	IL	1776-1850	2
		1851-1884	No fires
		1885-1996	1
Guyette and others 2003a	IN	1693-1801	No fires
		1888-1929	2
Henderson and Long 1984	IN	1929-1981	5
Henderson and Long 1984	IN	1933-1981	2
Cole and Taylor 1995	IN	1900-1990	5
McEwan and others 2007b	KY	1885-1954	9
McEwan and others 2007b	KY	1893-1954	12
McEwan and others 2007b	KY	1879-1900	2
Shumway and others 2001	MD	1616-1992	8†
Guyette and Cutter 1991	MO	1710-1810	4
		1810-1989	6
Guyette and Cutter 1997	MO	1821-1940	5
Guyette and Cutter 1997	MO	1581-1700	10
		1701-1820	3
		1821-1940	4
Guyette and Cutter 1997	MO	1581-1700	8
		1701-1820	4
		1821-1940	4
Guyette and Cutter 1997	MO	1701-1820	9
		1821-1940	3
Guyette and Cutter 1997	MO	1581-1700	6
		1701-1820	5
		1821-1940	5
Guyette and Cutter 1997	MO	1581-1700	30
		1701-1820	7
		1821-1940	2
Guyette and Cutter 1997	MO	1701-1820	9
		1821-1940	2
Guyette and Cutter 1997	MO	1581-1700	30
		1701-1820	13
		1821-1940	4
Guyette and Cutter 1997	MO	1581-1700	21
		1701-1820	13
Guyette and Cutter 1997	MO	1701-1820	11
		1821-1940	2
Guyette and Cutter 1997	MO	1581-1700	>37
		1701-1820	4
		1821-1940	2
Guyette and Cutter 1997	MO	1581-1700	12
		1701-1820	3
		1821-1940	2
Guyette and Cutter 1997	MO	1701-1820	12
		1821-1940	4
Guyette and Cutter 1997	MO	1701-1820	6
		1821-1940	2
Guyette and Cutter 1997	MO	1581-1700	19
		1701-1820	6
		1821-1940	3
Guyette and Cutter 1997	MO	1701-1820	13
		1821-1940	4
Guyette and Cutter 1997	MO	1701-1820	20
		1821-1940	5
Guyette and Cutter 1997	MO	1581-1700	7
		1701-1820	6
		1821-1940	2
Guyette and Cutter 1997	MO	1701-1820	16
		1821-1940	7
Guyette and Cutter 1997	MO	1701-1820	>45
		1821-1940	4
Guyette and Cutter 1997	MO	1701-1820	>50
		1821-1940	8
Guyette and Cutter 1997	MO	1701-1820	9
		1821-1940	6

†Weibull median or Weibull modal fire interval

(Table 3 continued on next page)

Table 3 (continued).—Descriptive data for fire scar based reconstruction sites from published studies in the Central Hardwood Forest Region. Only time periods with provided mean fire interval (MFI) values are reported. All MFI values are rounded to the nearest integer.

Reference	State	Period of record	MFI
Guyette and Cutter 1997	MO	1701-1820	10
		1821-1940	4
Guyette and others 2003b	MO	1700-1780	10
		1781-1820	3
		1821-1850	2
		1851-1890	2
		1891-1940	3
Guyette and others 2003b	MO	1700-1780	16
		1781-1820	4
		1821-1850	1
		1851-1890	1
		1891-1940	2
Guyette and others 2003b	MO	1700-1780	13
		1781-1820	3
		1821-1850	3
		1851-1890	2
		1891-1940	6
Guyette and Dey 1997a	MO	1700-1820	7
		1821-1930	2
Guyette and Dey 1997b	MO	1701-1820	6
		1821-1900	3
Stambaugh and others 2005	MO	1634-1780	22
		1780-1850	2
		1851-1930	2
Cutter and Guyette 1994	MO	1740-1850	3
		1850-1991	24
Guyette and Stambaugh 2004	MO	1604-1700	7
		1701-1820	4
		1821-1940	4
Dey and others 2004	MO	1705-1830	4
		1831-1960	8
Buell and others 1954	NJ	1627-1950	>10
McCarthy and others 2001	OH	1624-1997	2
McEwan and others 2007b	OH	1917-1936	2
McEwan and others 2007b	OH	1875-1934	8
McEwan and others 2007b	OH	1878-1931	7
McEwan and others 2007b	OH	1900-1936	9
McEwan and others 2007b	OH	1889-1931	6
McEwan and others 2007b	OH	1889-1931	5
Hutchinson and others 2008	OH	1855-1935	9
Hutchinson and others 2008	OH	1858-1935	9
Hutchinson and others 2008	OH	1844-1935	15
Sutherland 1997	OH	1856-1995	5
Guyette and Stambaugh 2005	TN	1740-2002	5
Armbrister 2002	TN	1837-1934	7
Feathers 2010	TN	1685-2008	6
Feathers 2010	TN	1678-2008	3
Harmon 1982	TN	1856-1940	13
Laforest and others 2007	TN	1836-1929	7
Aldrich and others 2010	VA	1704-2003	5-17
DeWeese 2007	VA	1779-1934	3
DeWeese 2007	VA	1758-1934	4
DeWeese 2007	VA	1810-1934	2
DeWeese 2007	VA	1789-1934	3
Hoss and others 2008	VA	1794-2005	3
Sutherland 1993	VA	1765-1993	9-11
Wolf 2004	WI	1829-1839	4
		1840-1871	20
		1871-2004	5
Maxwell and Hicks 2010	WV	1898-2005	5
Schuler and McClain 2003	WV	1846-2002	18

Similar to the soil and sedimentary charcoal based fire networks, the fire scar based network is spatially clustered within certain subregions of the CHFR (Fig. 2). While both charcoal-derived records are concentrated in the Appalachian Highlands, over half of the published fire scar based histories have been focused in the Interior Highlands. Of all states that occupy a portion of the CHFR, Missouri has the highest number of site-specific fire scar reconstructions followed by Ohio (Table 3). A total of 12 states which comprise a portion of the CHFR have at least one fire scar based reconstruction. The Appalachian Plateaus (Mixed Mesophytic Forest Subdivision) and the Ridge and Valley (Appalachian Oak Forest Subdivison) provinces had the second and third highest occurrences of fire scar reconstructions. The longest fire scar records were reported from the Interior Highlands, with a mean record initiation date in the late-1600s (the oldest being 1581). The Appalachian Plateaus region supported the shortest fire records with a mean fire chronology start date in the mid-1800s (although the longest record for the province extended to 1616).

Of all site-specific fire scar datasets, slightly more than half report a temporally delineated fire frequency record based on human settlement and land-use patterns. Temporal delineation allows for comparisons between periods with different human population densities and land-uses, factors which are major influences on historic fire regimes (Guyette and others 2002, Guyette and Spetich 2003). Throughout the CHFR, indeed throughout the eastern United States, population density has widely fluctuated over the last four centuries. Within the depth of record afforded by fire scar analysis, changes in human population density and fire are commonly represented by four periods: 1) Native American depopulation (ca. mid-1500s to 1800); 2) Native American repopulation (not a full recovery but population increase above the minimum) and early European settlement (ca. 1800 to 1850); 3) widespread European settlement (ca. 1850 to 1930); and 4) fire suppression (ca. 1930 to present). The timing of these events differed between regions (Denevan 1992, Millner and others 2001, Ramenofsky 1987).

Decimation of Native American populations by the spread of alien contagious infectious diseases in the eastern United States began as early as the 16th century in the Mississippi River Valley but was not widespread throughout the region until the mid-17th century (Denevan 1992, Millner and others 2001, Ramenofsky 1987). The timing of widespread European settlement and intensive land use also differed significantly between regions. For example, areas of the mid-Atlantic, the Northeast, and portions of Kentucky and Ohio were settled by 1800 whereas broad-scale settlement of the eastern United States was not complete until 1850 (Gerlach 1970). In fact, widespread European settlement on the Cumberland Plateau, located only ca. 550 km from major eastern port cities, did not occur until well into the 1800s. In summary, the commonly used temporal designations are site-specific based on local population densities and culture. Of the common temporal delineations, only the onset of the fire suppression period has a largely static date (between 1920 and 1940).

Within the CHFR fire scar record, 25 sites provide explicit fire data for portions of the Native American depopulation period. The overwhelming majority of records that extend to this period are located in Missouri and Arkansas. This period is characterized by relatively long fire-free intervals. For example, during the Native American depopulation period, no fires were recorded for over a century in an Indiana barren (Guyette and others 2003a), and the mean fire return interval (MFI) was more than 37 years in a Missouri hardwood savanna (Guyette and Cutter 1997). The vast majority of the temporally delineated records that extend to this period display longer fire return intervals as compared to the subsequent period of repopulation and early European settlement. This pattern has been explained by low human population density and thus fewer ignitions. However, this pattern was not evident in two Appalachian Highland fire reconstructions. Shumway and others (2001) sampled on side slopes of Savage Mountain in western Maryland and obtained a fire scar record extending to 1616. The authors did not find differences in fire importance between the pre-European and post-European settlement periods. They mention direct evidence of Native American activity downslope from the sample sites. During the period of aboriginal depopulation, Native American settlements were sparsely scattered throughout the eastern United States (Millner and others 2001). It is therefore possible that the sample site was located near a Native American settlement location that was not completely eradicated by the infectious diseases that decimated a majority of Native American populations in the region. Aldrich and others (2010) reconstructed fire history at Mill Mountain, a xeric ridgetop site in the Ridge and Valley of Virginia. Similar to Shumway and others (2001), the fire regime did not differ significantly between the pre-European and post-European settlement periods. However, throughout the extent of the record (beginning ca. 1700), European influence was documented in the area (e.g., hunting, trading, and raiding parties), and the site was then settled in the mid-1700s. Therefore, these pre-European settlement fires may have been either anthropogenic ignitions or natural ignitions from terrain-induced thunderstorms during dry conditions (Aldrich and others 2010), a distinction that is indiscernible in the fire scar record alone. Though this study is beneficial for understanding ridgetop pine-oak communities in the Central Appalachians, the findings may not be representative of the broader eastern oak forest region.

The Native American repopulation and early European settlement period is short (e.g., some studies classify this period as lasting 20 years), variable, and not used in all fire reconstructions. Aboriginal populations rebounded from the decimation at different speeds (Denevan 1992, Ramenofsky 1987). Similarly, early European settlement was variable in extent and intensity. Generally, fire during this period was more frequent than the preceding depopulation period but less frequent than the subsequent period of widespread European settlement. The fire return intervals during this transitional period were variable and typically ranged from a low of 1 to a high of 12 years between fire events. Two studies in the Central Lowlands documented anomalous fire histories during the early European settlement period. McClain and others (2010) and Wolf (2004) documented fire-free periods at the onset of early European settlement. This pattern was attributed to a purposeful avoidance of fire by European settlers as they feared destruction of crops, fences, buildings, and other property (McClain and others 2010).

During widespread European settlement, the spatial extent and intensity of human impacts on forest communities increased throughout the eastern United States (Cronon 1983, Motzkin and others 1999, Whitney 1994). This period was typified by frequent fires, more frequent than those during the preceding periods. Indeed, many studies documented fire return intervals as short as 1 to 3 years. In contrast to many presettlement fires, fires during this period were typically smaller in extent because of fuel fragmentation and fire breaks (Guyette and others 2002). However, Shumway and others (2001) found that presettlement fires were smaller in relative extent but greater in relative intensity than postsettlement fires. This period of frequent fire ended in the early 20th century with the onset of active fire suppression.

Regional patterns of fire occurrence are difficult to discern because many records are not temporally delineated, include depopulation and fire suppression periods, and fire histories are site specific. However, some general spatial trends are evident. As compared to eastern portions of the CHFR, fire return intervals were generally shorter in the western portions of the region. Many studies conducted in the western portions documented fire return intervals of less than 3 years with some annual fires. In contrast, the mean fire return interval in eastern portions was ca. 7 years with some longer fire free intervals. Again, we stress that fire histories are site specific and that, with the distribution of fire scar studies, this comparison is largely between the Interior Highlands and the Appalachian Highlands.

SYNTHESIS OF FIRE HISTORY

Elucidation of broadscale historical fire regime characteristics in the CHFR was difficult because each of the reviewed biological archives provided information with different spatial and temporal resolutions. Nonetheless, some general patterns could be gleaned from these records. Based on charcoal data, fire frequency in the CHFR increased during cultural transition periods (i.e., from Archaic to Woodland and from Woodland to Mississippian). These results could be interpreted that fire return intervals were shorter during these transition periods, that fire was more widespread, or both. Regardless, the increase in fire (as evidenced by an increased in charcoal) during these periods may have resulted in long-lasting legacies in some oak ecosystems. The fire scar record provided more detailed fire history information, albeit over a shorter period of history. In general, fire-free intervals were longest during the Native American depopulation period. Fire was more common during the Native American repopulation and early European settlement period, but did not drastically increase until the period of widespread European settlement. Many studies during the widespread European settlement period revealed MFIs of 1 to 3 years. These fires were typically smaller in extent than those of prior periods because of fire breaks caused by forest fragmentation. This trend illustrated the influence of human population density and land use, which differed by culture, on fire regimes. Certainly there were exceptions to these general trends, and notably some of the outliers were clustered spatially in distinct physiographic provinces. We suggest fire historians continue to analyze temporally delineated fire history characteristics. The dates of cultural milestones (e.g., European settlement) differ across the CHFR, and site-specific temporal delineations allow for comparison of historical periods across a multitude of sites and thus, allow for a clearer interpretation of broadscale trends. Without question, additional fire histories developed from each of these archives are needed to understand past fire regimes and the influence of those events on oak ecosystems, especially in regions where fire reconstructions are sparse or on site types where fire history is poorly understood (e.g., mesic and submesic sites).

RESTORATION IMPLICATIONS

Ecological restoration is the act of returning an ecosystem to a prior state (Egan and Howell 2001). The practice of restoration implies that the site has been degraded and that the previous condition is more desirable based on management goals (Swetnam and others 1999). Fire has been incorporated into a variety of restoration plans (Pyke and others 2010) as prescribed fires can be used in structure- or process-based restoration activities (Parsons and others 1986, Swetnam and others 1999, Vale 1987). In the former, fire is used as a silvicultural tool much like thinning or herbicide application to shape species composition and stand structure to that of the target reference conditions (Brose and others 2001). In the latter, prescribed fire is used to mimic historical disturbance from an identified time period (e.g., aboriginal burning) (Brose and others 2001). Certainly, prescribed fire is being used by forest managers for purposes other than ecological restoration. However, if fire is to be used in ecological restoration to achieve management goals, certain procedures should be regarded. First, a reference model should be developed by identifying a discrete time period that represents the desired conditions for the site (Landres and others 1999). Second, managers should implement a multi-proxy reconstruction of site history using cultural and biological archives. During this process, the historic range of variability (HRV), which is the range of conditions within which ecosystems are in dynamic equilibirum, should be quantified as it provides the restoration targets (Egan and Howell 2001, Swetnam and others 1999). The HRV refers to both pattern and process and incorporates many variables including composition and structure measures and information regarding disturbance type, frequency, extent, and magnitude. Third, a silvicultural prescription should be developed to restore the site to the target conditions (i.e., within the HRV). Fire can be implemented as either a structure- or process-based treatment. Finally, criteria should be developed to monitor restoration success (Egan and Howell 2001). While reference models can be developed using ecological theory or environmental reconstructions, results from the published studies reviewed in this manuscript reveal that fire histories are site specific. Therefore, managers focused on ecological restoration are best advised to construct a place-based history and not rely on results from other studies alone to set restoration targets and monitor treatment success.

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FIRE AND VEGETATION DYNAMICS IN THE CROSS TIMBERS FORESTS OF SOUTH-CENTRAL NORTH AMERICA

Steve W. Hallgren, Ryan D. DeSantis, and Jesse A. Burton

Associate Professor (SWH), Department of Natural Resource Ecology and Management, Oklahoma State University, 008C Ag Hall, Stillwater, OK 74078; Postdoctoral Research Associate (RDD); Fire Ecologist (JAB). SWH is corresponding author: to contact, call 405-744-6805 or email at steve.hallgren@okstate.edu.

Abstract.—The vegetation of the Cross Timbers forests was maintained for thousands of years by fire; the historic fire return interval was likely 1 to 10 years. Fire suppression over the past century led to a doubling of stand basal area, increased diversity of tree species, reduced dominance of oak, and increased mesic species intolerant of fire. Dendrochronological studies suggested tree mortality from severe droughts opened stands and accelerated recruitment of eastern redcedar (*Juniperus virginiana* L.) where fire was suppressed. Without reintroduction of fire, Cross Timbers forests will become denser and more diverse with a less diverse understory of herbaceous species, and in many locations, eastern redcedar may become dominant. Research showed prescribed burning every 4 years was sufficient to control understory hardwoods and benefit herbaceous plant cover and diversity which increased with fire up to a burn interval of 2 years. Because low intensity surface fires characteristic of the region usually do not remove established trees, mechanical or chemical methods may be necessary to restore open forests and savannas. Further research is warranted to determine how restoring Cross Timbers forests and savannas to more open conditions will affect ecosystem services including carbon storage, water supply, biological diversity, and wildlife resources.

INTRODUCTION

The ecotone between the eastern deciduous forest and southern Great Plains is a patchwork of forest, savanna, and prairie. This vegetation type, known as the Cross Timbers (Fig. 1), is believed to have once covered nearly 8 million ha from southern Kansas to north-central Texas (Küchler 1964, Therrell and Stahle 1998). The forests of the Cross Timbers may contain some of the least disturbed primary forest in the eastern United States (Therrell and Stahle 1998). Large tracts of what appear to be old-growth forests are still found throughout the region where they have remained relatively undisturbed due to their location on poor shallow soils with rocky outcrops and steep slopes. These areas were rarely logged due to the timber having low commercial value, and they were rarely cultivated for crops due to the poor soil. Many of these forests contain 200 to 400 year-old post oaks (*Quercus stellata* Wangehn.) and 500 year-old eastern redcedar (*Juniperus virginiana* L.).

The Cross Timbers provides ecosystem services to a regional population of over 7 million persons, and this number is expected to grow to over 17 million by 2040 (Texas Forest Service 2008). The capacity of the ecosystem to provide services is threatened by many factors including changes in land use, urbanization, pollution, overgrazing, global climate change, invasive species, and changes in the fire regime. A study of 143 Cross Timbers stands found that over the past 50 years, approximately 50 percent of the forest area was lost, mostly to agriculture (Stallings 2008).

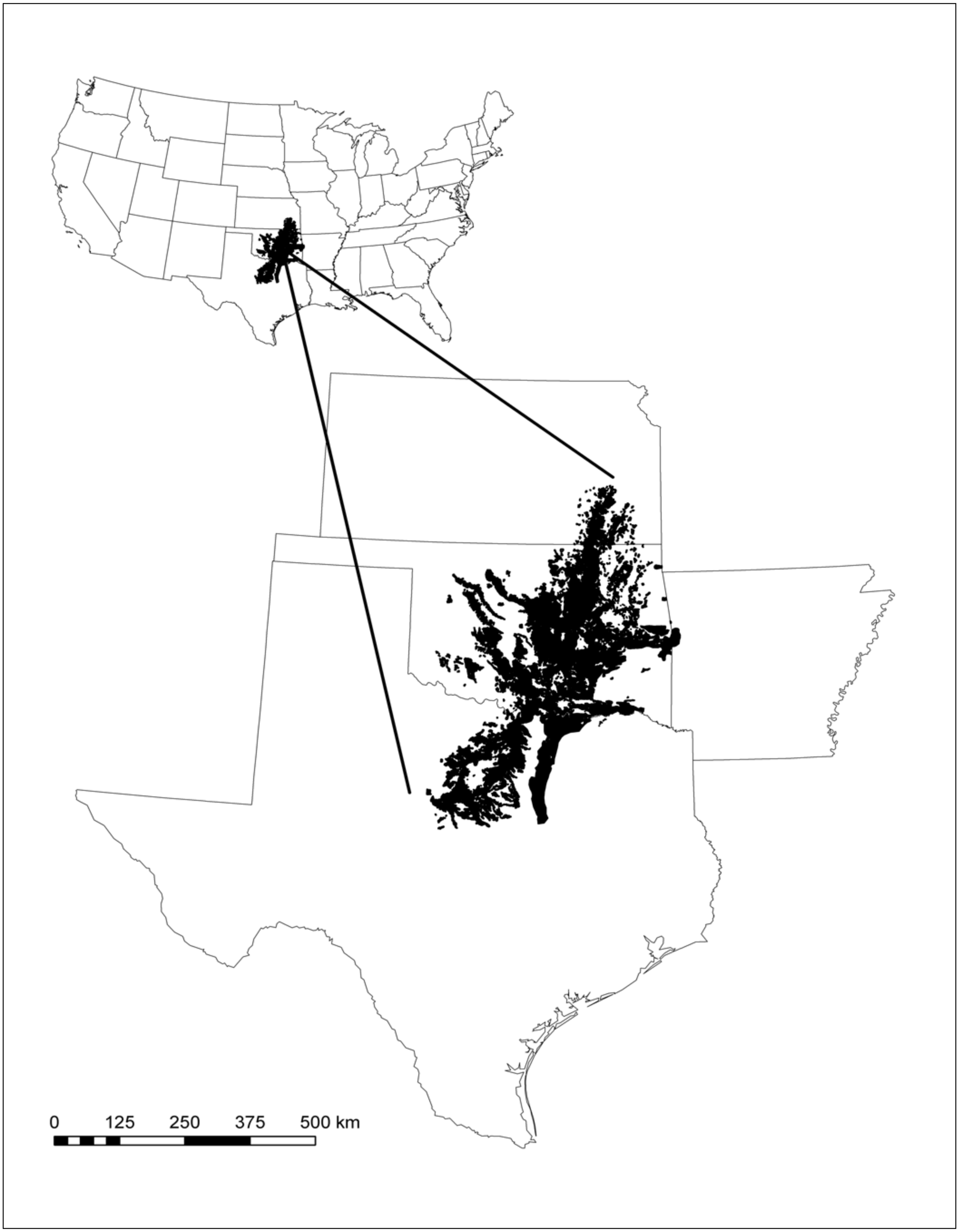

Figure 1.—Geographic extent of the Cross Timbers.

Encroachment by eastern redcedar due mainly to fire suppression has become one of the major threats to sustainable management in the region (Drake and Todd 2002). When eastern redcedar invades grasslands, it reduces plant species richness (Briggs and others 2002, Gehring and Bragg 1992, Linneman and Palmer 2006) and forage production for livestock and wildlife (Briggs and others 2002) and creates more dangerous fuel conditions for wildfire (Drake and Todd 2002). It seems likely eastern redcedar encroachment in the forest would have similar results, but research to confirm this assumption has not been done. There is strong interest in maintaining and restoring Cross Timbers forests to pre-Euro-American settlement (EAS) conditions, and prescribed fire is considered to be an important tool to achieve this objective. Unfortunately, there is relatively little research to provide guidance for prescribed burning of forests.

This paper presents the current knowledge of the role of fire in shaping the forest vegetation of the Cross Timbers. It describes an anthropogenic landscape where thousands of years of human fire created a vegetation type that is changing rapidly to more dense mesophytic forests due to fire suppression. This new trajectory cannot be altered without the reintroduction of fire and substantial mechanical reduction of woody vegetation. The success of Cross Timbers vegetation restoration will be measured by a return to and maintenance of the savanna, woodland, and open upland oak forests last observed at the beginning of Euro-American settlement.

PHYSICAL SETTING

Low elevation and low relief characterize the Cross Timbers Region. Elevation ranges from a low of approximately 150 m along the Red River to a high of 430 m in the Arbuckle Mountains and 755 m in the Wichita Mountains (Johnson 2006b). The rock outcrops are mainly horizontal or slightly dipping sandstones and shales of the Pennsylvanian, Permian, Cretaceous, and Tertiary age. Ridges are formed by the more resistant sandstones. Because most of the surface rock is not very resistant to erosion, much of the area is low rolling hills and broad flat valleys, and the maximum relief rarely exceeds 180 m (Johnson 2006a).

The mean annual temperature decreases from 18 °C in the south to 13 °C in the north, and the mean annual rainfall increases from 600 mm in the west to 1010 mm in the east. Precipitation is well distributed throughout the year with a maximum in May. Wide swings in precipitation can result in severe droughts. The most remarkable recent droughts for the region occurred in the 1930s and 1950s. The summers are hot and the temperature can exceed 40 °C, and winters are generally mild, but when cold arctic air intrudes, the temperatures drop to -25 °C (Arndt 2011).

VEGETATION

The vegetation has undergone many changes during the Holocene. Pollen studies at a site near the east-central part of the Cross Timbers in Oklahoma suggested that as the climate changed following the last retreat of the glaciers, the initial vegetation was grassland that began to give way to oaks about 9000 years before present (YBP) (Bryant and Holloway 1985). For thousands of years the vegetation appeared to be oak savanna, and during mid postglacial times about 5000 YBP, the increase in oak dominance led to the development of oak woodlands. Pines arrived about 2100 YBP and an oak-hickory-pine forest finally developed approximately 1200 YBP, but pine never developed dominance and is restricted to isolated pockets and the eastern edge of the Cross Timbers forests. Charcoal was present indicating fire was an important disturbance force since at least 5000 YBP (Albert 1981).

Forest canopy vegetation is dominated by post oak and blackjack oak (*Quercus marilandica* Muenchh.) and subdominant species include black hickory (*Carya texana* Buckl.) and black oak (*Q. velutina* Lam.) (Ewing and others 1984, Rice and Penfound 1959). The overstory trees are small in stature; the top of the canopy is usually between 12 and 14 m (Fig. 2). Closed stands can attain a basal area of 23 to 27 m^2/ha with 1400 to 1800 trees/ha and an average stand diameter at breast height (d.b.h.) of 14 to 17 cm (Karki 2007). The understory shrubs include poison ivy (*Toxicodendron radicans* [L.] Kuntze), rough-leaf dogwood (*Cornus drummondii* Meyer), redbud (*Cercis canadensis* L.), and coralberry (*Symphoricarpus orbiculatus* Moench) (Ewing and others 1984). Common herbaceous plants in the forest understory and prairies include bluestems (*Schizachyrium scoparium* [Michx.] Nash and *Andropogon* spp. L.), Indiangrass (*Sorghastrum nutans* [L.] Nash), and rosette grasses (*Dichanthelium* spp. [Hitchc. & Chase] Gould) (Ewing and others 1984, Tyrl and others 2002). The Cross Timbers ecotonal position contributes to its high taxonomic diversity. A study of the The Nature Conservancy's Tallgrass Prairie Preserve found 763 species in 411 genera and 109 families (Palmer 2007). Over 1100 plant species were found at the Lyndon B. Johnson (LBJ) National Grasslands in north-central Texas (O'Kennon, pers. comm., Botanical Research Institute of Texas, 2009).

The dominant forces shaping the vegetation of the ecotone include soils, climate, and fire. The savannah and forest vegetation occur mostly on the coarse textured soils derived from sandstones or granites (Dwyer and Santelmann 1964, Rice and Penfound 1959). Grasslands occur predominantly on fine textured soil derived from shale and limestone. That means the climate is not a true grassland or

Figure 2.—Typical Oklahoma Cross Timbers forest in dormant season.

forest climate, because either vegetation type can be supported depending on the soil texture and its effects on water relations (Rice and Penfound 1959). The region-wide gradient of decreasing moisture from east to west is strongly manifested in the vegetation that decreases in stature and richness from east to west as the forests give way to savannas, and eventually to grasslands. Fire is a major force in shaping the vegetation mosaic.

FIRE

The potential vegetation of the region as mapped and defined by Küchler (1964) was assumed to support understory fires with a return interval of 0 to 10 years (Brown 2000). It is commonly held that fire suppression has effectively ended wildland fire as a force in shaping the ecosystems. However, recent evidence suggested fire suppression may not have been effective everywhere. Fire scar analysis found the mean fire interval (MFI) in some locations decreased after Euro-American Settlement (Allen and Palmer 2011, Clark and others 2007, DeSantis and others 2010b) or changed very little (Stambaugh and others 2009). This may indicate fire suppression has been spatially highly variable. It was shown for similar forest vegetation in Missouri that MFI declined soon after EAS because settlers learned to use fire for many of the same purposes as the Native Americans. MFI did not increase until there was a substantial increase in human population in the 20th century that brought an end to fire use and resulted in strong support for fire suppression to protect property (Guyette and others 2002). Perhaps variability in human population density and land use across the Cross Timbers landscape has resulted in a patchwork of vegetation with very different MFIs.

Based on fire scar analysis of the dominant oaks that extends back to the early 18th century (Table 1), it seems likely most of the fires were human caused and occurred during the dormant season. From 80 to 97 percent of the fires were classified as occurring during the dormant season because the scars were located at the boundary between annual rings. A recent study determined the dormant season for cambial activity in post oak and blackjack oak was from mid-August to late March (Thapa, pers. comm., Oklahoma State University, 2010). This means scars during that period were caused by dormant season fires. Historical records of 418 dated fires in the Great Plains over the period 1535 to 1890 mentioned only one lightning fire. Lightning is abundant in the region, but most of it occurs with heavy rainfall which is not advantageous for igniting wildland fires. The most common reasons for Native Americans to set fires were for communication and warfare. In contrast, most fires started by settlers were accidental (Moore 1972). Approximately 60 percent of the fires occurred during the dormant season, mid-August to late March, and 40 percent were recorded during the growing season for the entire Great Plains and for the Cross Timbers Region. Another study of records of 590 fires in the Great Plains found 50 percent of the fires occurred in September and October, the dormant season (Gaskill 1906). These reports support the conclusion that although a substantial number of fires occurred during the growing season, slightly more than half of

Table 1.—Mean fire interval (MFI) and prevalence of dormant season fires for selected sites pre- and post-EAS (Euro-American settlement)

Location[1]	MFI (years) Pre-EAS	MFI (years) Post-EAS[2]	Dormant season (percent)	Reference
TGPP	3.4	1.3*	77	Allen and Palmer 2011
OWMA	4.0	2.0*	95	DeSantis and others 2010b
KAFP	4.9	2.1*	--	Clark and others 2005
WMWR	4.4	5.2	97	Stambaugh and others 2009

[1] TGPP = Tallgrass Prairie Preserve; OWMA = Okmulgee Wildlife Management Area; KAFP = Keystone Ancient Forest Preserve; and WMWR = Wichita Mountains Wildlife Refuge.
[2] * = Pre- and post-EAS MFI different at P=0.05.

fires occurred in the dormant season. There is some evidence that fires burn more frequently (Stambaugh and others 2009) or to a greater extent (Clark and others 2007) during periods of severe drought. But the link between fire frequency and extent and weather has not been established. Most often correlation analyses with drought indices show no relation to fire occurrence (Allen and Palmer 2011, Clark and others 2005, DeSantis and others 2010b, Stambaugh and others 2009).

STAND DEVELOPMENT

In addition to fire there are numerous natural disturbance forces acting on these forests including tornados, straight-line wind, drought, hail, and ice that kill or damage large branches, single trees, and small groups of trees (Clark 2003). Large-scale catastrophic disturbances are not common. Tornados, one of the few forces to create large disturbances, are rare (return interval ~2000 years [NOAA 2011]) relative to the lifespan of the major tree species (300 to 500 years [Therrell and Stahle 1998]). The fires typical of the region are low intensity, rarely kill individual trees >5 cm d.b.h. (Burton and others 2010), and are not stand replacing events. Although the canopy trees are considered to be relatively intolerant of shade, there is evidence of continuous replacement of canopy trees by growth of understory saplings into the canopy without gaps (Clark and others 2005, Karki 2007). There are also small gaps that facilitate growth of saplings into the canopy (Karki 2007). Research suggested gap-phase regeneration was the major type of canopy replacement in these xeric oak forests, as most trees had rapid growth from the pith indicating they started in a gap (Clark 2003).

Recent research found no evidence for change in overstory composition in Cross Timbers forests where gap-phase regeneration was occurring with frequent burning (Karki 2007). Although blackjack oak contributed a higher percentage of the gap makers than its proportion of the canopy, it and all the other species had replacement probabilities sufficient to maintain their relative abundance in the canopy. In contrast, there was strong evidence for major changes in forest composition and density where fire had been suppressed. Remeasurement in the 2000s of plots first measured in the 1950s (Fig. 3) found basal area had doubled, oak dominance had declined, mesophytic and fire-intolerant species including eastern redcedar had increased, and tree species richness had increased (DeSantis and others 2010a). Dendrochronological studies showed a large increase in eastern redcedar recruitment beginning in the 1960s and declining oak recruitment in the 1970s (Fig. 4). There was strong evidence the combination of oak mortality from severe

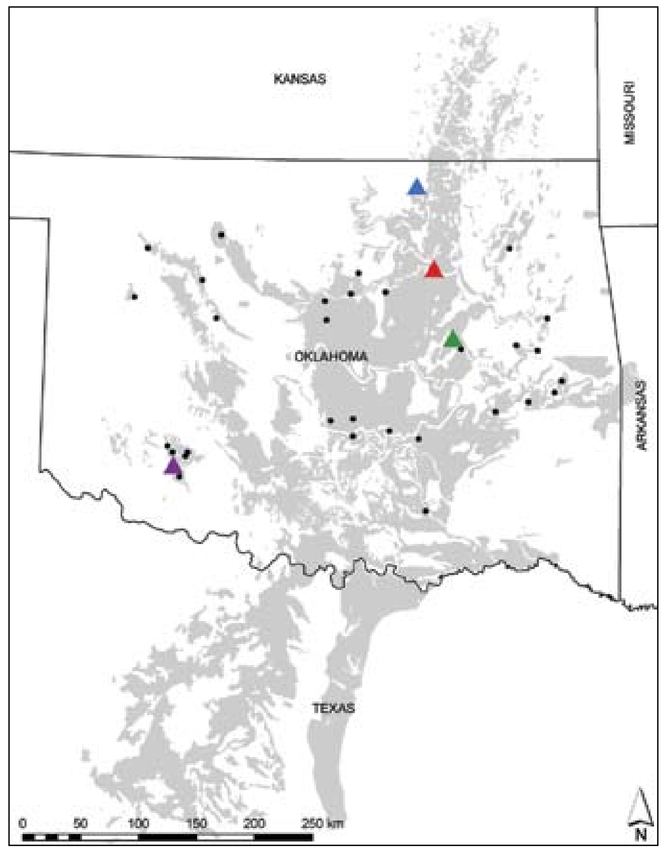

Figure 3.—Map of Cross Timbers with research sites. Black dots show locations of 30 forest stands measured by Rice and Penfound (1959) in the 1950s and again by DeSantis and others (2010a) in 2008 and 2009. Blue triangle = Tallgrass Prairie Preserve (Allen and Palmer 2011, Karki 2007); green triangle = Okmulgee Wildlife Management Area (Burton and others 2010, Burton and others 2011, DeSantis and Hallgren 2011, DeSantis and others 2010b, Karki 2007); red triangle = Keystone Ancient Forest Preserve (Clark and others 2005, Karki 2007); and purple triangle = Wichita Mountains National Wildlife Refuge (Stambaugh and others 2009). Adapted from Therrell and Stahle (1998).

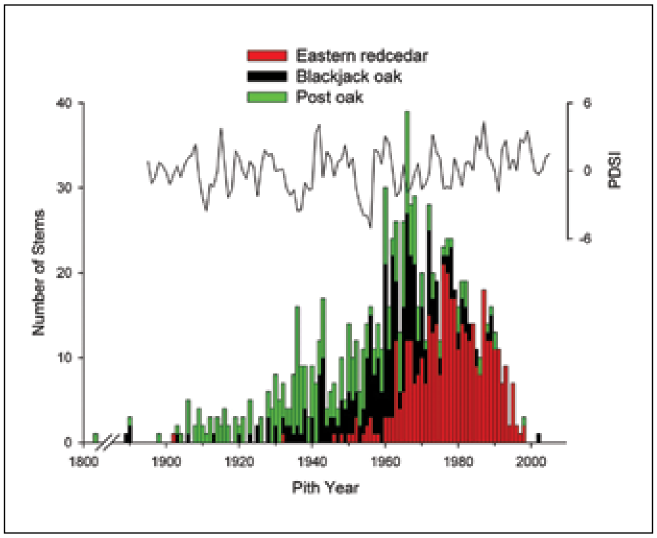

Figure 4.—Recruitment year for *Juniperus virginiana*, *Quercus marilandica*, and *Quercus stellata*, and instrumental Palmer Drought Severity Index (PDSI) in Cross Timbers forests of Oklahoma. Data represent sums of number of stems for three Oklahoma regions and average of regional PDSI (from DeSantis and others 2011).

droughts, mostly in the 1950s, and fire suppression were responsible for the changes (DeSantis and others 2011). The mortality of mainly blackjack oak from the 1950s drought (Rice and Penfound 1959) opened the stands for recruitment of new cohorts of forest tree regeneration. Fire suppression removed the control on eastern redcedar encroachment and ended the stimulation of oak sprouting. Eastern redcedar had always been a minor component in riparian zones and on ridges where fire burned infrequently. It is disseminated widely and effectively by birds (Holthuijzen and Sharik 1984, Holthuijzen and others 1986) and cannot tolerate fire when small (Engle and Stritzke 1995). It appeared suppression of fire may have contributed to eastern redcedar becoming widespread and may lead to its dominance in some forests.

Regeneration of the dominant oaks, post oak and blackjack oak, is highly adapted to fire. Although successful reproduction by seedlings is very rare (Backoulou 1998, Clark and Hallgren 2003, DeSantis and Hallgren 2011), both species sprout prolifically. A burn frequency of 1 in 10 years was found to produce the highest number of sprouts, up to 22,000 stems/ha of both species combined. Mortality thinned the sprouts over time since the last burn. Apparently these oak species are capable of accumulating reproduction over time even without fire, as stands not burned for over 20 years were found to have sprout densities as high as 11,000 stems/ha. Although the average age of sprouts in nonburned stands was only 5.5 years and the height was only 40 cm, some of the sprouts were 20 years old (DeSantis and Hallgren 2011).

Regeneration in these post oak-blackjack oak forests can be characterized as auto-accumulating (Clark and Hallgren 2003, Johnson 1993). There can be four or more cohorts of reproduction that accumulate over at least 20 years. Most sprouts are from stumps and seedlings (d.b.h. <5 cm), and both species produce a substantial number of root sprouts (attachment to root >25 cm from proximal end). Post oak can produce 30 percent root sprouts, and blackjack oak around 20 percent (Clark and Hallgren 2003, DeSantis and Hallgren 2011). A recent study found high density sprouting 6 years after a fire where none of the reproduction dated before the fire, and many of the root systems of young plants were much older, up to 67 years. The height growth of both species is relatively slow due to low moisture and light in these closed xeric stands, attaining 6 to 10 cm per year over the first 6 years (Clark and Hallgren 2003, DeSantis and Hallgren 2011). This slow growth means the sprouts may not become large enough to resist even the low intensity burns until after many years.

Post oak and blackjack oak dominance of these forests is well documented. A study of Oklahoma's forests in the 1950s found them to be the most important species on a statewide basis by every ecological measure including presence, constance, frequency, density, and basal area (Rice and Penfound 1959). Species of secondary importance were black oak and black hickory in the eastern Cross Timbers. A remeasurement in the 2000s of 30 of the 208 original stands measured in the 1950s found post oak was still the most important canopy tree across the state, but blackjack oak had decreased in importance, and the second most important species had changed to eastern redcedar in the west and central regions and black hickory in the east. The species shift was even greater in the saplings which reflect composition of the future canopy. The relative density of post oak and blackjack oak saplings dropped by nearly two-thirds across the Cross Timbers. The replacement species included eastern redcedar in the west and black hickory and winged elm (*Ulmus alata* Michx.) in the east (DeSantis and others 2010a).

PRESCRIBED FIRE

Taken together the vegetation type and climate suggest the Cross Timbers forests supported frequent low intensity fires pre-EAS. The fine fuels in the grasslands were easier to ignite than the oak leaf litter, so fires likely started there and burned into forests. When Native Americans were managing the landscape, fires occurred year-round. At present, there is very little information about fire effects on vegetation to help make decisions about prescribed fire. The most common type of prescribed fire is a low intensity dormant season burn (Fig. 5) (Weir 2011).

Prescribed fire both kills oak sprouts and stimulates sprouting. Although Cross Timbers forests produce abundant oak sprouts even without fire, infrequent prescribed burning (one fire per decade [FPD]) was found to increase the density of sprouts. Sprouts are susceptible to high mortality from burning, and because they are slow growing, it takes many years to become resistant to fires (DeSantis and Hallgren 2011). When oak sprouts grow above 1.4 m after 10 to 15 years, they begin to show resistance to burning, and trees >5 cm d.b.h. easily survive the low intensity dormant season fires commonly prescribed for Cross Timbers forests. In contrast, understory non-oak saplings and shrubs (>1.4 m tall and <5 cm d.b.h.) were strongly reduced by prescribed burning two or more times per decade, dropping from nearly 1,300 to 200 stems/ha (Fig. 6). Woody plant species richness was greatest when there were no burns in 20 years and dropped by 50 percent when fire frequency was increased to 5 FPD (12 vs. 6 species per treatment). Eventually, the small diameter non-oak saplings in the unburned stands may recruit into the larger size classes and change the overstory composition (Fig. 7). It appears that prescribed burning at two or more FPD may be effective in maintaining oak dominance in these stands (Burton and others 2010).

Figure 5.—Low-intensity dormant season prescribed fire typical for Cross Timbers forests. Prescribed fire was set when relative humidity ranged from 30 to 50 percent, temperature was <27 °C, and winds were <25 kph. The forest was burned every 4 years.

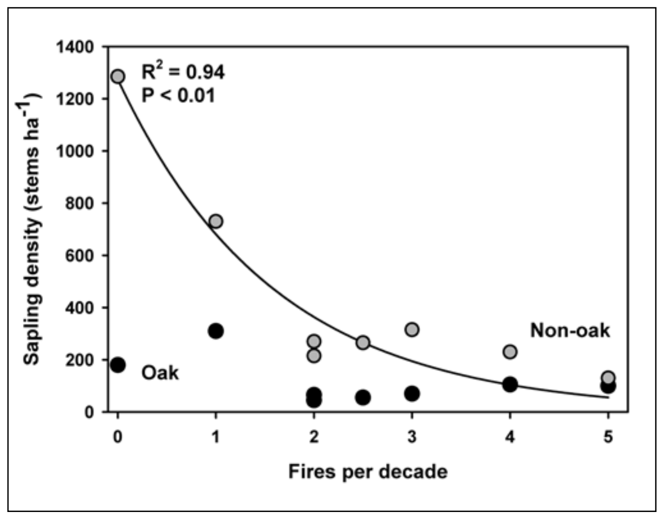

Figure 6.—Effect of fire frequency on sapling density of oak (*Quercus* spp.) and non-oak tree and shrub species (Burton and others 2010).

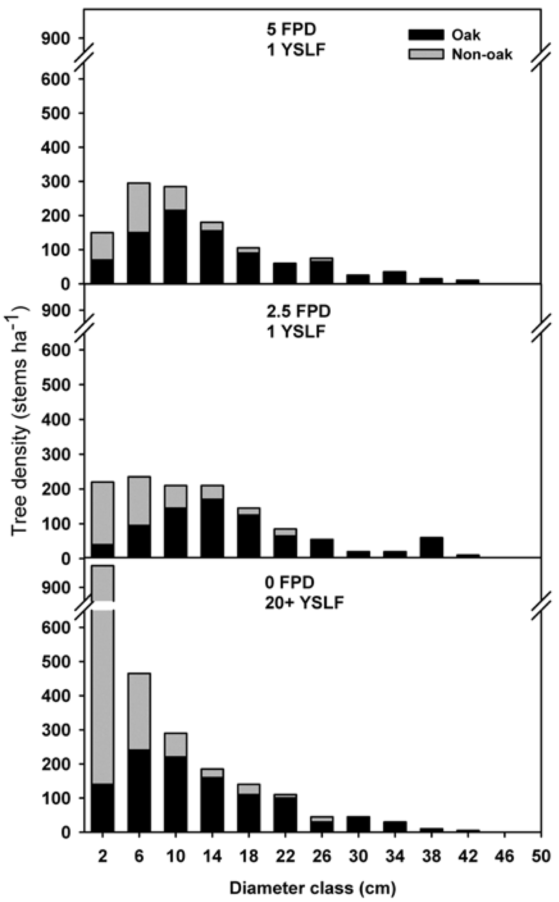

Figure 7.—Effect of fire frequency (fires per decade, FPD) and years since last fire (YSLF) on tree diameter distributions in terms of stems per ha by 4-cm diameter classes. Prescribed burning treatments were applied for 20 years (Burton and others 2010).

Despite the low light and strong competition for moisture and nutrients under the closed canopy of Cross Timbers forests, the herbaceous layer can be very responsive to prescribed burning (Fig. 8). This contrasts with the common belief that the understory herbaceous layer in oak forests will not respond to fire under a closed canopy (Franklin and others 2003, Harrington and Kathol 2009, Hutchinson and others 2005). In Cross Timbers forests with canopy closure of 88 to 96 percent, the understory <1.4 m tall was dominated by woody reproduction followed by forbs, C_3 graminoids, C_4 grasses, and legumes (Burton and others 2011). The cover of woody reproduction did not change with fire frequency, most likely due to its propensity to replace reproduction killed by fire with more sprouts. Forb cover increased threefold and C_3 graminoid cover increased twofold over the range of 0 to 5 FPD. Both plant functional groups increased in richness over the same range of fire frequencies. Fire frequency did not affect the cover or richness of C_4 grasses likely due to the low light intensity under the full canopy. Biomass of combined C_4 and C_3 graminoids increased nearly fourfold from 0 to 5 FPD. The direct reduction in litter by burning may contribute to the increased herbaceous cover at high fire frequencies (Hiers and others 2007). In addition, the frequent fires reduced the density of non-oak shrubs and saplings thereby indirectly reducing litter production and reducing competition (Burton and others 2010).

RESTORATION OF CROSS TIMBERS VEGETATION

Research conducted nearly 40 years ago presented a likely scenario of changes in forest and savanna vegetation resulting from fire suppression in the Cross Timbers (Johnson and Risser 1975). The forest appeared to have been a savanna that had filled in with trees over 80 to 90 years to become a forest. Comparison of the size class distribution of trees in the forest with a contemporary savanna showed nearly identical numbers of trees in the large size classes and a much larger number of small trees in the forest. It appeared the small post oaks had recruited in the period 1910 to 1940 and the blackjack oaks in the period 1890 to 1920. As a consequence of recruitment of these two oak species, the basal area of the forest was double that of the savanna. These results were interpreted to show that when fire use ended and fire suppression began after EAS, oak encroachment changed the savanna to a forest.

The savanna can be kept open by fire because the high production of herbaceous fine fuels creates conditions for fires hot enough to reduce and kill woody plants (Johnson and Risser 1975). Once the savanna becomes a forest, the fuel type changes to tree leaf litter that produces a less intense fire, and the closed canopy

Figure 8.—Typical forests on the Okmulgee Wildlife Management Area, Oklahoma where prescribed burns have been conducted for over 20 years at the rate of 0, 1, 2.5, and 5 fires per decade (FPD).

prevents production of large quantities of fine fuels. As a consequence, fire intensity in the forest is too low to kill established trees, and fire alone cannot restore closed forest to open savanna. A likely force to thin forests to savanna density is severe drought, and the last century had two very severe droughts in the 1930s and 1950s; however, neither of these disrupted the conversion of savanna to forest. The severe drought of the 1950s, which was acknowledged to have killed many oaks (Rice and Penfound 1959), appeared to open the forest to encroachment by non-oak mesophytic and fire sensitive species because there was no fire to control their establishment and stimulate the oak sprouts (DeSantis and others 2011). The mortality suffered in the 1950s drought was more than replaced over the next 50 years, and today these stands have twice the basal area they had then. The general belief is that without fire use in the Cross Timbers, woody encroachment will convert the grasslands to savannas, savannas to forests, and forests to denser and more diverse mesophytic forests.

Restoration of Cross Timbers vegetation to pre-EAS conditions will require treatments to reverse the encroachment of woody plants and increase grasslands and savannas and more open forests. Research has shown as few as two FPD can control the composition and structure of woody vegetation. When fire frequency is increased to five FPD, herbaceous plant diversity and grass biomass will strongly increase even under a closed canopy (Burton and others 2010, Burton and others 2011). Although fire can maintain the open grassland, savanna, and woodland, it rarely kills established trees (Briggs and others 2005, Burton and others 2010, Engle and others 2006). Opening the forests and savannas will require treatments to first reduce tree density, and second to maintain the open condition long-term. Both mechanical removal plus frequent fire (Masters and others 1993) and chemical thinning plus frequent fire (Scifers and others 1981, Stritzke and others 1991) have been successful at reducing overstory tree density and stimulating herbaceous biomass production. Although the dormant season is the most common time for prescribed burning in the region (Weir 2011), research has shown growing season burning may be more effective in controlling woody plants which may be more susceptible to damage when actively growing (Engle and others 1996, Waldrop and others 1992).

The pre-EAS Cross Timbers region was an anthropogenic landscape with more grasslands, savannas, and open forests than found today. Restoring the current vegetation to that condition will require treatments to remove woody vegetation and prescribed fire to prevent its return. Successful restoration will result in an oak dominated forest with only a minor component of eastern redcedar and mesophytic tree species. The woody vegetation will be less diverse, and herbaceous vegetation will be more diverse and abundant. Further research is warranted to increase the understanding of the most effective tools for Cross Timbers forest restoration and to gain the confidence of land managers in their use. The knowledge of mechanical and chemical control of woody plants needs to be increased. The current understanding of effects of prescribed burning is based almost entirely on research with low intensity dormant season fire in rangelands. The effects of growing season prescribed burning are poorly understood, and managers are reticent to employ it. Research on growing season prescribed burning has the potential to greatly increase the capability to restore Cross Timbers vegetation because its use will expand the window for prescribed burning and contribute to better control of woody plants (Weir 2011).

The woody encroachment of Cross Timbers plant communities over the past 100 years has changed more than the vegetation. Carbon storage, hydrologic cycles, biogeochemical cycles, and wildlife populations have also changed. Restoring the current landscape to pre-EAS vegetation may reverse these changes, and there is very little understanding of the consequences. Vegetation type and land use can have large effects on carbon storage. Forest lands can have much larger stocks of biomass carbon than grasslands, whereas grasslands can have larger stocks of soil organic carbon (SOC) (Jackson and others 2002, Post and Kwon 2000). Not only do grasslands allocate more net primary production (NPP) below ground, the distribution of carbon to the soil recalcitrant pools can be much higher in grasslands. The hydrologic and biogeochemical cycles are strongly affected by the type and density of vegetation (Huxman and others 2005, Wilcox and others 2006, Zhang and others 2001). Some wildlife benefit from the denser woody vegetation of a landscape without fire while others favor the vegetation produced by frequent fire, and landscape level species richness may be greatest where there is a patchwork of fire regimes (Engle and others 2008, Fuhlendorf and others 2006, Fuhlendorf and others 2009, Leslie and others 1996). Restoring the Cross Timbers to pre-EAS vegetation will require substantial efforts and major changes in land use. Intelligent and sustainable management of the Cross Timbers cannot be attained without more complete understanding of factors and forces controlling all its component resources and ecosystem services.

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FIRE ECOLOGY OF OAK ECOSYSTEMS: THE BASIS FOR MANAGEMENT

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OAK SAVANNA RESTORATION: OAK RESPONSE TO FIRE AND THINNING THROUGH 28 YEARS

Ronald E. Masters and Jack R. Waymire

Associate Professor (REM), University of Wisconsin - Stevens Point,
Stevens Point, WI 54481; Senior Biologist (JRW), Oklahoma Department of Wildlife Conservation.
REM is corresponding author: to contact, email at ron.masters@uwsp.edu.

Abstract.—We used a small plot study on Pushmataha Wildlife Management Area in southeast Oklahoma to determine the efficacy of fire frequency and thinning as management tools for restoration of oak savanna, oak woodlands, pine-bluestem woodlands, and pine savanna for application on a landscape scale. On selected experimental units, we initially reduced stand density to favor either oak canopy dominance or pine dominance to near presettlement stand density. Thinned stands were then subjected to 0-, 4-, 3-, 2-, and 1-year late dormant season (late February - early April) fire frequency regimes for 26 years. For comparison, we withheld control units from treatment and also included unthinned but with 4-year burn regime treatment units. We included two additional thinning treatments, the oak-savanna and pine-bluestem treatments, both with annual burn regimes; the oak-savanna had all pine removed (approximately 50 percent of the basal area) and the pine-bluestem had half of the hardwood thinned (approximately 25 percent of the pre-treatment basal area). We compared mortality rate, acorn production, and growth response of selected post oaks and blackjack oaks. We also assessed nutrient content of post oak acorns to determine prescribed fires potential influence on nutrient status. We found a differential response by species to presence or absence of fire; but all selected trees responded favorably in diameter growth to thinning. Blackjack oak mortality was highest on unthinned and unburned sites versus any of the fire treatments because of hypoxylon cancer, an indirect result of high stand density (competition) and drought stress. Mortality of post-oaks was related to initial burns, and to some extent, cumulative effects of fire frequency interacting with fuel loads. Although thinning efforts on a landscape level were applied on the Wildlife Management Area beginning in 1978, fire frequency was >4-year intervals, inadequate for maintenance of savanna and woodland structure. Based on small plot study results we begin landscape application of frequent fire on a 1-3 year cycle in 1997 and increased thinning in 1999-2001. Restoration thinning and a more focused burn regime was applied in a site specific manner on the landscape in 2008 through present. Woodland-grassland and forest-shrub obligate songbirds, white-tailed deer, and Rocky Mountain elk have responded favorably.

INTRODUCTION

Fire suppression along with attendant forest densification has had a pervasive, deleterious influence on numerous ecosystems across eastern North America (Nowacki and Abrams 2008). In particular, the interior Highlands of Oklahoma, Arkansas, and parts of Missouri have over the last century and more succeeded from a landscape matrix composed of prairie elements, oak and pine savanna (see Appendix for list of common and scientific names), and oak or pine woodland-grassland complexes to a dense homogeneous hardwood or oak-pine dominated land cover (Beilmann and Brenner 1951; Foti and Glenn 1991; Masters and others 1995, 2007). Historically associated wildlife species included bison, elk, white-tailed-deer, many other mammals, and birds in abundance (Featherstonhaugh 1844, Smith and Neal 1991).

This forest densification and homogenization has led to declines in species richness and abundance of flora (Gilliam and others 2006, Sparks and others 1998) and fauna. Declines in wildlife species included extirpation of bison, elk, some meso-mammals (Smith and Neal 1991), and near-loss of some small mammals and woodland-grassland obligate songbird species, particularly those associated with pine-grasslands (Masters and others 1998, 2002; Wilson and others 1995).

As forest density and canopy cover increases, understory herbage production declines and understory structure changes (Fenwood and others 1984; Masters and others 1993a, 1996). This has a profound influence on many species of wildlife in several respects. In closed canopy forests of the Ozark and Ouachita Mountain regions similar to other Interior Highlands in the eastern United States, the understory forage base for white-tailed deer and elk is often inadequate to meet nutritional requirements to sustain healthy populations year round. This problem is compounded in winter because of little or no suitable evergreen winter browse (Masters and others 1993a, 1997; Segelquist and Pennington 1968; Segelquist and others 1969, 1972). In addition to herbage declines, structural complexity changes with densification and canopy closure, thus precipitating major impacts on songbird and small mammal communities (Engstrom and others 1984; Masters and others 1998, 2002; Wilson and others 1995).

Oak mast provides a valuable food resource for numerous wildlife species (McShea and Healy 2002). Oak mast failures in this region have been linked to winter mortality of deer, decreased productivity, and summer fawn mortality (Logan 1972; Segelquist and others 1969, 1972). During late fall and winter, deer preferentially select hard mast and only turn to browse and herbage in appreciable amounts when acorns are unavailable (Harlow and others 1975, Segelquist and Green 1968).

Establishment of food plots has been the traditional approach to provide additional forage as a contingency for mast failure (Masters and others 1993a, 1997). Some value has been shown for providing supplemental forage in the limited productivity forests of the Interior Highlands (Segelquist 1974, Thompson and others 1991). Deer mortality is less in years of mast failure in areas of contiguous closed canopy forest where comparatively large forage clearings have been developed (Segelquist and Rogers 1974). However, adequate nutrition for deer in years of mast shortfall can only be provided in unique situations because oak mast is substantially higher in nutritional quality than typically available winter forage (Harlow and others 1975).

Pine-grassland restoration efforts via thinning and use of frequent fire have resulted in favorable population responses of pine-grassland obligate songbirds, small mammals, and in successful restoration of understory herbaceous communities and improved food resource availability for deer (Masters and others 1996, 1998; Sparks and others 1998; Wilson and others 1995). However, little research attention has been given to comparative efforts using thinning and frequent fire for restoration of oak savannas or mixed oak-pine woodlands on the effects on wildlife occurrence or on the long-term effects of varying fire frequency on oak survival and development.

In the late 1960s and early 1970s, the Pushmataha Wildlife Management Area (PWMA) in southeast Oklahoma experienced a precipitous decline of the deer population and of a recently established elk population. Because of the importance of oak mast for deer and the need for improving the forage base for these two focal species, the Oklahoma Department of Wildlife Conservation began long-term research on thinning and frequent fire strategies that retained selected oaks at lower stand densities on a series of small plots designated as the Pushmataha Forest Habitat Research Area (FHRA) (Masters 1991a, 1991b; Masters and others 1993a). The goal was to critically examine then current strategies to develop early succession openings (ESOs) for forage and to improve mast production by releasing selected oaks from competition.

The forest management strategy at that time employed commercial thinning of mixed oak-pine stands to create ESOs. These were created by harvesting merchantable pine and thinning remaining hardwoods to a target basal area (BA) of about 9 m^2/ha, approximating presettlement stand density in parts of the Ouachita Mountains (Foti and Glenn 1991, Masters and others 2007). The fire frequency needed to perpetuate early succession openings was unknown as was the effects of frequent fire on oak survival and general health. Conventional wisdom at that time suggested frequent fire was deleterious to oaks. Following mid-term results, management strategies were to be developed and applied on the landscape of PWMA.

To that end we examined post oak and blackjack oak response in untreated control, thinned only, and thinned and burned stands at 4, 3, 2, and 1 year burning cycles on the FHRA beginning in 1983. Our goal was to evaluate the validity of a current forest management strategy for determination of leave trees in timber harvest units. As an objective, we wanted to determine the effects of fire frequency on survival, diameter growth, crown development, and acorn production. Further, we wanted to determine if fire had an influence on nutrient content of acorns. Following thinning we predicted that selected trees would experience increased radial and crown growth and increased acorn production as a result of decreased canopy, water, and nutrient competition. We expected a cumulative increase in oak mortality within a 10-year study horizon, commensurate with crossing some fire-interval threshold encompassed within our increasing fire frequency treatments. Further, we expected that longer interval repeated burning would have positive cumulative effects on growth and survival over time as selected trees were freed from canopy and moisture competition.

Results from the small plot study would then be used to develop a restoration strategy for treatment application to the PWMA landscape. A critical part of that strategy included monitoring and evaluating deer, elk, and songbird response to landscape application of restoration treatments.

STUDY AREA

The study area was 29.1 ha in size and located within the FHRA (34°32’N, 95°21’W) on the 7,395 ha PWMA near Clayton, Oklahoma. The PWMA is located in the Kiamichi Mountains along the western edge of the Ouachita Highland Province. The study area is on thin, drought prone Hector-Pottsville soils developed from cherty shales and resistant sandstones. They belong to the Carnasaw-Pirum-Clebit association with a high proportion of surface rock and areas of rock outcrop (Bain and Watterson 1979). Soil nutrient levels and their response to our periodic prescribed fire and thinning treatments have been described by Masters and others (1993b). Available P, K, Ca, and Mg increased and nutrient cycling was enhanced when thinning and burning were applied together.

The climate is semi-humid to humid with hot summers and mild winters (Bain and Watterson 1979). Rainfall is somewhat unequally distributed throughout the year with August being drought prone and the driest month ($\bar{x}$ = 6.0 cm) and May the wettest ($\bar{x}$ = 14.7 cm) (Masters 1991b). From 1978-2010 average annual precipitation on or in the study area vicinity was 124.4 cm (unpublished data from Department of Natural Resource Ecology and Management, Oklahoma State University and Oklahoma Department of Agriculture, Forestry Services). From 1984 to 1996 annual precipitation was 146.5 cm (SD = 26.1), well above the 34 year average, and from 1997-2010 annual precipitation was 113.5 cm (SD = 27.6), which was significantly lower ($P > T$ =0.004) than the previous period. We obtained Palmer Drought Severity Index (PDSI) data during our study period for Oklahoma and Arkansas from the National Oceanic and Atmospheric Administration (2011). We used an October-September water-year for rainfall and drought indices data summarization (Fig. 1). In spring 1987, a late freeze affected acorn production and ice storms followed by high winds and crown damage to trees on FHRA occurred in December 1996, December 1997, December 2000, and January 2007.

Prior to acquisition from 1946-54, PWMA was grazed, selectively harvested, and frequently burned. The Forest Habitat Research Area was protected from further logging, grazing, and fire until 1984. The FHRA has been excluded from fire at least since 1969.

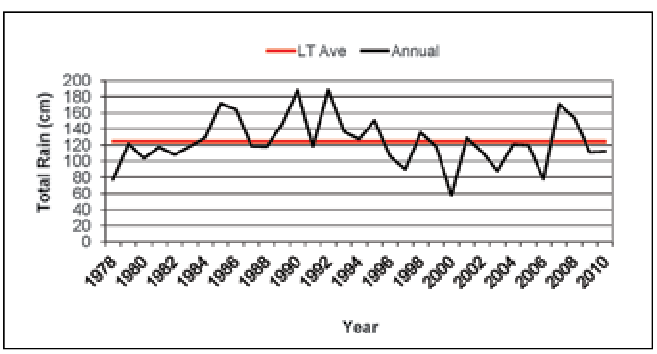

Figure 1.—Annual and long-term average (LT AVE) annual rainfall (cm) based on an October-September water-year for Pushmataha Forest Habitat Research Area 1978-2010, Pushmataha Wildlife Management Area, OK. Rainfall for the period 1983-1996 was higher (P=0.004) than for the period 1996-2010.

Prior to treatment and on unharvested sites (control and rough reduction burn), post oak, shortleaf pine, mockernut hickory and blackjack oak were respective dominates in the overstory. The BA of unharvested treatments ranged from 24-30 m^2/ha. Common understory species included: tree sparkleberry, poison ivy, Virginia creeper, greenbriar, and muscadine grape. Although sparsely occurring, the most prevalent understory herbaceous plants were little bluestem, panicums, and sedges (Masters 1991a, b).

Post-treatment harvested and thinned openings were dominated by tall grasses and woody sprouts of varying density, depending on fire frequency and time since burned for a given site. Grasses included big bluestem, little bluestem, and to a lesser extent, Indiangrass, and in drainages switchgrass. Post oak, blackjack oak, winged sumac, and blackberry were dominant understory woody species. The thinned overstory was comprised of sparse (~9 m^2/ha BA stems >5 cm at 1.4-m height) post oak and blackjack oak (Masters 1991a,b; Masters and others 1993a). The clearcut treatment was planted with loblolly pine in 1986. Masters (1991a,b), Masters and Engle (1994), Masters and others (1993a,b, 1997), and Crandall (2003) have described the study area, management, and vegetation of these stands. The deer population on the entire PWMA was estimated at 7.3 deer/km^2 ± 3.8 ($\bar{x}$ ± SE) in 1984 based on spotlight counts. At that time the total elk population was six animals (Masters 1991b).

METHODS

Experimental Design

Beginning in summer 1984 through present, 10 treatments with 1-4 replications were applied to 36, units (0.2-1.6 ha in size) in a completely randomized experimental design (Masters and others 1997). For the purposes of this paper, only the eight treatments that retained and involved monitoring of selected oaks were utilized (Table 1). The HT treatment included one replication until 1989 when a spot-over burned part of the unit. Because the monitored trees were not in the

Table 1.—Study treatment acronyms, descriptions, and number of replications (N) for the Pushmataha Forest Habitat Research Area, Pushmataha Wildlife Management Area, Oklahoma, for 1983-present

Treatment	Description	N
Control	Control, no thinning, no burning	3
RRB	Rough reduction, late-winter prescribed burn, 4-year interval	3
HNTI	Harvest pine timber only, late-winter prescribed burn, 1-year interval; Oak savanna, except 1995	3
HT	Harvest pine timber, thin hardwoods, no burn (natural regeneration to a mixed stand) (initially 1 replication was monitored, then an additional 3 were monitored after 1989)	4
HT4	Harvest pine timber, thin hardwoods, late-winter prescribed burn, 4-year interval	3
HT3	Harvest pine timber, thin hardwoods, late-winter prescribed burn, 3-year interval	2
HT2	Harvest pine timber, thin hardwoods, late-winter prescribed burn, 2-year interval, except 1995	3
HT1	Harvest pine timber, thin hardwoods, late-winter prescribed burn, 1-year interval, except 1995	3

burned area, the trees on this unit were retained for the purposes of this study. Three additional replications of the treatment were included at that time that had not previously been monitored.

During summer 1984, merchantable pine trees were harvested in assigned treatments, and hardwoods were selectively thinned by single stem injection using 2,4-D to approximately 9 m^2/ha basal area. Prescribed strip-head fires were applied on appropriate treatment units in mid-March and early April 1985 and in succeeding years at 1-, 2-, 3-, and 4-year intervals. Fireline intensity of March 1988 burns ranged from 628-903 kW/m (Masters and Engle 1994).

On each treatment unit, 10 permanent plots were established at 20 m intervals on two randomly located lines perpendicular to the contour. No sampling was conducted within 20 m of any edge to prevent bias from adjacent treatment units (Mueller-Dombois and Ellenberg 1974, Oosting 1956). Basal area of overstory vegetation was quantified each year using the variable radius plot method (Avery 1967). Basal areas of stems ≥5 cm diameter at breast height (d.b.h.) were measured with a 10-factor wedge prism at each permanent plot location. Baseline sampling before cultural treatment application was conducted in 1983. Overstory canopy cover was determined from mid-September to mid-October prior to leaf abscission. Estimates were made with a 9-point grid in a sighting tube with vertical and horizontal levels at plot center and cardinal points at 2 m and 4 m from each permanent plot location in most years from 1985 to 2010 (Mueller-Dombois and Ellenberg 1974).

Mast Tree Sampling

We selected three post oaks and three blackjack oaks, where available, as study trees from each treatment unit, following commercial pine harvest in 1984. Initially we sought to include black oaks as well, but given the paucity of this species on the study area and unequal distribution, only a few were available and chosen for monitoring. Black oaks did not occur in all units or treatments.

On each unit, all oak trees were systematically examined before selection. Those selected must not have been damaged by commercial timber harvest, in terms of canopy breakage or bark damage from felled trees or skidders. Further, trees were evaluated based on canopy development, height, and d.b.h. Oaks in the dominant or codominant position in the canopy were selected over intermediate or suppressed trees. However, some of the selected blackjack oaks were in intermediate canopy position as these were all that were available. The best formed and most vigorous appearing trees were selected from all candidate trees because acorn production is strongly related to tree diameter, crown area, and condition (Goodrum and others 1971). Trees on harvested units had complete crown release from competition and were generally farther than 20 m from the nearest residual tree. All competing residual trees greater than 5 cm in diameter that had touching or overlapping crowns were single stem injected as described earlier. We chose to select trees based on specified criteria rather than at random because we were evaluating validity of the current forest management strategy for determination of leave trees in timber harvest units.

Variables included for measurement included age, species, d.b.h., total height, crown area, and since 2000 crown density. Measurements were taken every year until 1989 and at least every 2 years thereafter. Mortality was tracked each year. In the first years of the study we did not measure crown area and height of trees on the annually burned treatments.

We measured d.b.h. annually to the nearest 0.1 cm using a standard diameter-tape. We used a clinometer to estimate total tree height to the nearest 0.5 m at 20 m distance until 2000. In following years a calibrated hypsometer was used to estimate height to the nearest 0.1 meter height.

To estimate crown area, we measured live crown diameter at two perpendicular axis and then calculated area based on the formula for the area of an ellipse whose semiaxes are a and b, $A = \Pi ab$. To assess crown density and health, from 1994 to 2010 we estimated canopy density beneath each sample tree from mid-September to early October prior to leaf abscission at cardinal points half the distance from the tree bole to canopy edge.

Postburn measurements after the initial thinning and first prescribed burn included bark-char height and height of limb scorching. Residual bark char prevented remeasurement in succeeding years. During early burns, fire behavior was estimated and fireline intensity was calculated after Masters and Engle (1994) and later modeled with BEHAVE or based on flame length estimates.

Acorn Sampling

We evaluated acorn production and viability from 1984-1989 and 1998 in Control, HT, HT4, HT3 and HT2 treatments. We used 18.93 liter buckets with an opening diameter of 28.96 cm for an effective trap area of 658.52 cm^2. Mast traps were fitted with wire screen covers to prevent wildlife depredation and placed at four quadrants under each tree; trap center was half the distance from the bole of the tree to the edge of the crown. We checked acorn traps at 1 week intervals beginning in early September until acorn drop was completed, generally late November during each sampling year. Acorns were counted, bagged, and the green weight recorded to the nearest 0.1 g. We also recorded the percent of acorns that were sound based on a viability rating of the percentage of sound acorns in the trap (Christisen and Kearby 1984, Goodrum and others 1971). We extrapolated total acorn production of viable acorns for each tree based on the crown area and percent of crown sampled.

In 1988, acorns from all post oaks on selected experimental units were collected for nutrient analysis by tree. Nutrient analysis of samples was conducted by Servi-Tech, Inc. (Dodge City, KS) and included moisture content, dry matter content, ash, crude protein, acid detergent fiber (ADF), total digestible nutrients (TDN), calcium, phosphorous, magnesium, and potassium.

Landscape Treatment and Wildlife Response

We selected stands to apply landscape thinning and burning treatments based on topographic position, site characteristics, and species composition. Historical references and settlement period tree density, basal area, and fire regime were also used to determine thinning levels and fire regime on targeted sites (Foti and Glenn 1991; Johnson 1986; Kreiter 1994; Masters and others 1995, 2007). Restoration thinning and burning began in 1999. Target stand types and target basal areas were oak-savanna and oak-pine savanna (7 m^2/ha), and oak-pine woodland and pine-bluestem woodland (14 m^2/ha). Riparian corridors and upland drainages were not thinned but were included in the adjacent administrative unit fire regime. More detail on proportion and juxtaposition of restoration treatments may be found in (Masters and Waymire 2012).

During August and September each year, spotlight counts were conducted for a minimum of 10 nights on a 16.1 km route using standardized Oklahoma Department of Wildlife Conservation protocols to index deer population status. In February of each year, total elk counts are made by driving the same route in late afternoon with the high count of the survey used as the population index. We used Breeding Bird Survey (BBS) data for Pushmataha County to derive trend data of target species of interest for PWMA (Sauer and others 2011). We chose 20 species to examine based on published accounts of habitat association and response to habitat change and whether or not they were species of special management concern (Cox and Widener 2008, Masters and others 2002, Wilson and others 1995). A total of 37 of the 50 stops in Pushmataha County were located on the PWMA. Individual stop data for each year from 1994-2009 were obtained for these 37 stops.

Data Analysis

Means were tested for homogeneity of variance among treatments using Levene's test (Snedecor and Cochran 1980). When variances were unequal we used the Kruskal-Wallis nonparametric test to determine treatment differences in acorn production, average annual diameter growth, tree height, and crown diameter (Steel and others 1997). Unit × year × treatment type III mean square was the error term (SAS Institute 1985). We used multiple comparisons between mean ranks with the Least Significant Difference (LSD) test with $P = 0.050$ (Steel and others 1997). We used rank transformations to analyze percentage data. Correlation analysis was used to examine relationships between weather, fire characteristics, and tree mortality and regression to examine linear trends in BBS data.

RESULTS

Stand Response

Pretreatment stands were characterized by closed canopy overstory and dense pine-hardwood midstory with extensive leaf-litter cover in the understory. Pine, hardwood, and total basal area ($\bar{x}$ ± SE; 25.3 m^2/ha ± 0.5) were similar (P>0.05) across all experimental units in 1983 but different (P<0.05) in 1984 after thinning and in following years (Table 2). Stand

Table 2.—Pine, hardwood, and total basal area (BA) (m^2/ha) change by treatment on Pushmataha Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK. Data in 1984 were post-thinning and in 2010 following fire frequency directed tree succession. Means with same letter were not different (P > F <0.004). See Table 1 for treatment descriptions. Most burns were in late February through mid-March.

Pine BA (m^2/ha)

Treatment	1984 Mean		SE	2010 Mean		SE
CONT	12.6	a	2.5	14.3	ab	2.0
RRB	13.9	a	1.7	15.5	ab	1.3
HNT1	0.3	b	0.2	0.9	d	0.3
HT	0.9	ab	-	21.1	a	6.2
HT4	1.1	ab	0.7	9.9	abc	3.7
HT3	0.6	b	0.3	2.0	cd	0.4
HT2	2.2	ab	0.7	2.6	bcd	0.5
HT1	0.5	b	0.2	1.1	d	0.6

Hardwood BA (m^2/ha)

Treatment	1984 Mean		SE	2010 Mean		SE
CONT	13.9	a	0.8	12.9	a	1.8
RRB	11.7	a	1.2	9.9	a	1.2
HNT1	9.6	ab	1.7	6.2	bc	1.1
HT	8.3	ab	-	10.0	ab	2.2
HT4	7.8	ab	0.9	3.9	cd	0.4
HT3	4.8	b	0.5	3.1	cd	0.1
HT2	6.3	ab	1.7	3.0	cd	0.3
HT1	6.8	ab	0.2	2.1	d	0.3

(Table 2 continued on next page)

Table 2 (continued).—Pine, hardwood, and total basal area (BA) (m^2/ha) change by treatment on Pushmataha Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK. Data in 1984 were post-thinning and in 2010 following fire frequency directed tree succession. Means with same letter were not different (P > F <0.004). See Table 1 for treatment descriptions. Most burns were in late February through mid-March.

Total BA (m^2/ha)

Treatment	1984 Mean		SE	2010 Mean		SE
CONT	26.5	a	1.7	27.4	a	0.7
RRB	25.6	a	1.0	25.2	a	0.5
HNT1	9.9	ab	1.8	7.2	bc	1.3
HT	9.2	ab	-	31.7	a	4.3
HT4	9.0	ab	1.5	13.8	ab	4.0
HT3	5.4	b	0.1	5.1	bc	0.5
HT2	8.5	ab	2.3	5.6	bc	0.7
HT1	7.3	ab	0.2	3.2	c	0.7

understory immediately following thinning was characterized by a clumped and patchy ground cover of slash and sparse cover of hardwood sprout shrubs, vines, grasses, and forbs (Fig. 2). After the first burn and following subsequent burns, grass and forb cover predominated on thinned and burned treatment stands and had 21 times greater standing crop than control. Herbaceous standing crop on unthinned but burned RRB was three times that of control stands (Masters and others 1993a) (Figs. 3 and 4). Over time tree structure developed along a fire frequency gradient with unburned stands developing into dense closed canopy stands. For frequently burned stands, as fire frequency increased tree density and basal area decreased (Table 2 and Figs. 3 and 4).

Mortality

By 2010 total mortality was 38 of the 131 selected trees for 29.1 percent (Fig. 5). Percent mortality of just blackjack oak was over double that of post oak (44.4 percent versus 18.1 percent). Fire related mortality for all trees was 16.8 percent, and hypoxylon mortality was 10.7 percent. A total of 12 blackjack oak trees and 1 black oak succumbed to hypoxylon canker. A higher proportion of total blackjack oak mortality was from disease rather than fire. Except for one tree, all blackjack oak mortality from hypoxylon canker was on unthinned treatments. Lightning mortality of two post oaks was 1.5 percent (Fig. 5).

Figure 2.—Example of stand conditions post pine harvest and hardwood thinning (HT) before the first burn. All HT units were similar in residual hardwood basal area before burning on the Pushmataha Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK.

Figure 3.—Photos of different treatments after 21 years of treatment application in early October 2006. All treatments in this and in Fig. 4 were similar to the control (top left) in 1983. Pine harvest and thinning was accomplished in early summer 1984 (see Fig. 2) and burning initiated in late winter 1985. Top left to right: **Control**, no treatment; **RRB**, no harvest or thinning, late winter burn on a 4-year interval. Bottom left to right: **HNT1**, harvest pine, no hardwood thinning, annual late winter burn; **HT**, harvest pine, no thinning and no burning on Pushmataha Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK.

Figure 4.—Photos of different treatments after 21 years of treatment application in early October 2006. All treatments in this figure were similar to the control (top left, Fig. 3) in 1983. Pine harvest and thinning was accomplished in early summer 1984 (see Fig. 2) and burning initiated in late winter 1985. Top left to right: **HT4**, harvest pine, thin hardwoods to approximately ½ of original basal area, late-winter burn at 4-year interval; **HT3**, harvest pine, thin hardwoods to approximately ½ of original basal area, late-winter burn at 3-year interval. Bottom left to right: **HT2**, harvest pine, thin hardwoods to approximately ½ of original basal area, late-winter burn at 2-year interval; **HT1**, harvest pine, thin hardwoods to approximately ½ of original basal area, late-winter burn at 1-year interval on Pushmataha Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK.

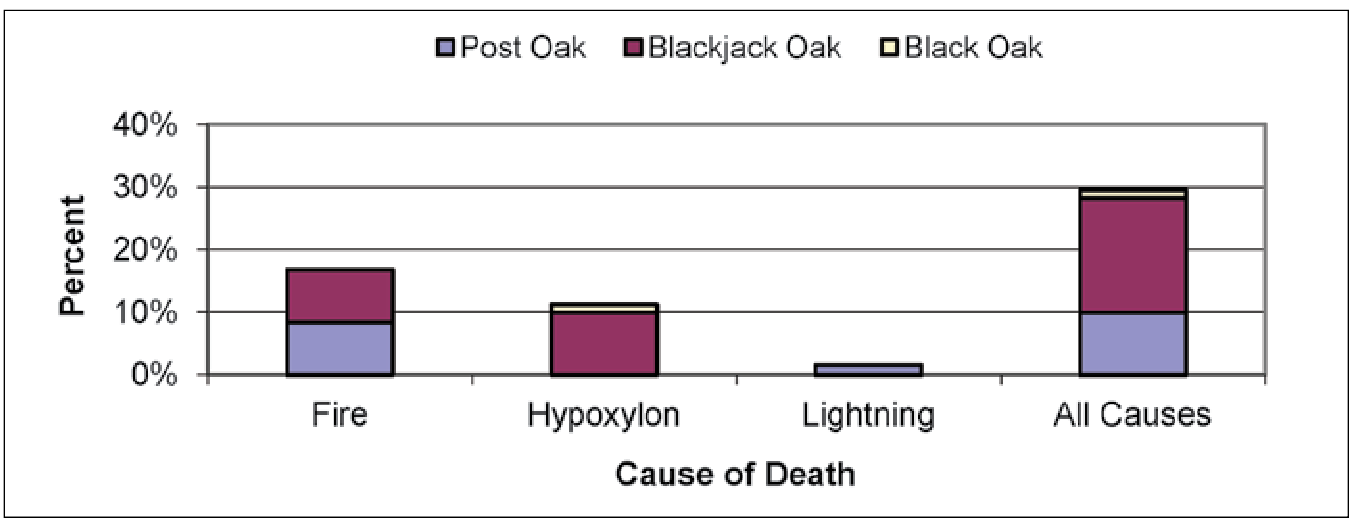

Figure 5.—Cause-specific mortality of selected post, blackjack, and black oaks on the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK from 1983-2010.

Blackjack and post oak mortality was 9.2 percent in the first growing season postburn and totaled 10.7 percent by the second year following introduction of fire on ESO treatments. This pulse of fire-caused mortality affected blackjack oak more severely than post oaks (Fig. 6). Smaller trees (lower diameter, height, and crown area) tended to succumb to fire in this initial pulse. Blackjack oak mortality leveled out following the first 2 years postfire reintroduction. The first canker-related mortality to blackjack oak occurred in 1994, and since 2002 subsequent mortality has been entirely from hypoxylon canker on control units and RRB treatments (Figs. 5 and 6). Of those trees dying from hypoxylon, 92.3 percent were during the period of below average rainfall and mortality was associated with the 2 previous years PDSI (*r*=0.327, *P*<0.05). The last fire-caused mortality of blackjack oak was in 2002. Control units experienced 87.5 percent mortality to blackjack oaks from hypoxylon canker (Fig. 7). Blackjack oak mortality on HT4, HT2, and HT1 treatments resulted from initial prescribed fires applied in the same week, and thus was not a fire frequency effect (Figs. 5 and 7).

Post oak attrition has been slight annually but steady until 2002 with 58 of the original 72 trees (81.9 percent) surviving thereafter through 2010 (Fig. 6). Post oak mortality was related largely to a fire frequency gradient on ESO treatments but also influenced by lightning mortality (Fig. 6 and 8). The HNT1 or post oak savanna treatment had the least post oak mortality of any of the thinned and burned treatments (*P*=0.004). Fuel loading tended to be lower on this treatment, particularly around the bole of selected trees, than on the other annually burned treatment, thus resulting in lower fire line intensity.

Crown Area, Height, and Diameter Response

Following commercial pine harvest and thinning of hardwoods in 1984, selected mast trees were similar in conformation (*P*<0.05) across all treatments. Average crown area, height, and d.b.h. of blackjack oak was

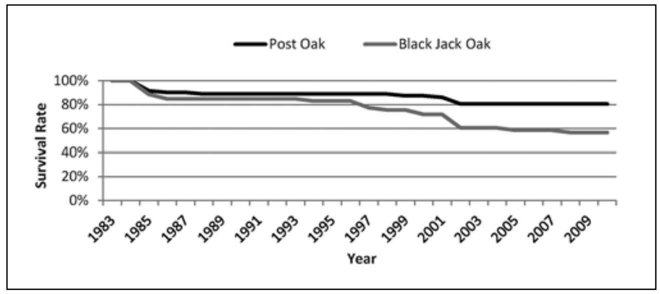

Figure 6.—Annual survival (%) of selected post and blackjack oaks from the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK from 1983-2010.

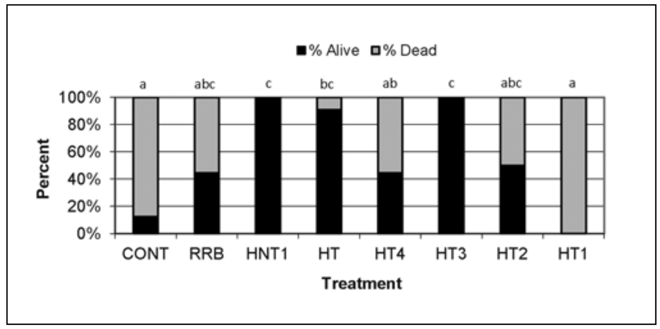

Figure 7.—Percent mortality of selected blackjack oaks by treatment on the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK from 1983-2010. See Table 1 for treatment descriptions. Mortality on the CONT, RRB and HT treatments was from hypoxylon canker. Mortality on the thinned and burned treatments was fire related. Percent mortality by treatment with the same letter was not different (*P*=0.014).

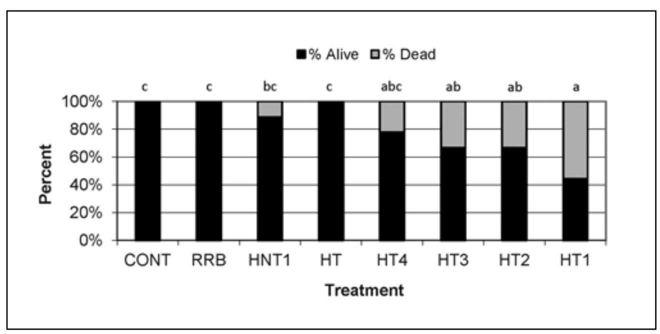

Figure 8.—Percent mortality of selected post oaks by treatment on the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK from 1983-2010. See Table 1 for treatment descriptions. Percent mortality by treatment with the same letters was not different (*P*=0.004).

respectively 25.1 m^2 ± 3.3, 11.0 m ± 0.4, 35.6 cm ± 1.2. Average crown area, height, and d.b.h. of post oak was 68.5 m^2 ± 4.3, 15.7 m ± 0.4, 23.6 cm ± 1.4, respectively.

It took 6 years for differences to be expressed in terms of blackjack oak physical characteristics following initial treatment. In subsequent years and during the period of above average rainfall, we found differences (P<0.050) in height in 3 years, d.b.h. in 4 years, and in crown area in 2 years for blackjack oak. When differences became apparent, the height was greater on HT2, RRB, and HT treatments than on Control treatments. However as the HT stand densified, height-growth slowed, and heights became similar to the Control by 1991 (P=0.36) but less than HT2 and RRB. Crown area and d.b.h. exhibited a similar pattern. After 1997 and throughout the period of below average rainfall, we found no differences in any of these variables between treatments for blackjack oak. Because of declining sample size and loss of some replications from both fire and disease mortality, the ability to detect meaningful differences was compromised somewhat as illustrated by average annual growth of blackjack oak across 28 years (Fig. 9).

We observed no treatment differences in average d.b.h. or crown area in any years in post oaks. However, we observed differences in average height by treatment 5 years following the ice storms. Post oaks from the HT treatment averaged greater heights than did the harvested and burned treatments. The treatments that were burned were much more open throughout the study, and crowns were more exposed to ice damage. The crowns of post oaks in the HT treatment were somewhat protected from the ice storms until 2007. Following that ice storm, post oaks in the HT treatment experienced severe crown damage and were near the lowest in height and were supplanted by the Control and RRB with highest average height. In contrast to height, average annual diameter growth was lowest for the HT treatment and highest for the HT4 (Fig. 10). Following ice storms and during the below average rainfall period, thinned and burned treatments had lower (P>0.050) heights because of top breakage in 4 years. Following the damaging storm in 2007 which affected trees across the board, we found no difference in height or crown area.

In 2010, average crown area, height, and d.b.h. of all blackjack oak across treatments was not different and was respectively 29.0 m^2 ± 4.8, 11.2 m ± 0.5, 31.4 cm ± 2.4. The same was true for post oak with average crown area, height, and d.b.h. of 78.1 m^2 ± 3.6, 15.5 m ± 0.3, 43.6 cm ± 0.9, respectively.

Acorn Production and Nutrient Content

In the first fall (1984) of the 7 years that we monitored post oak acorn production, we observed no differences among units treated alike because commercial harvest was accomplished by mid-summer and hardwood

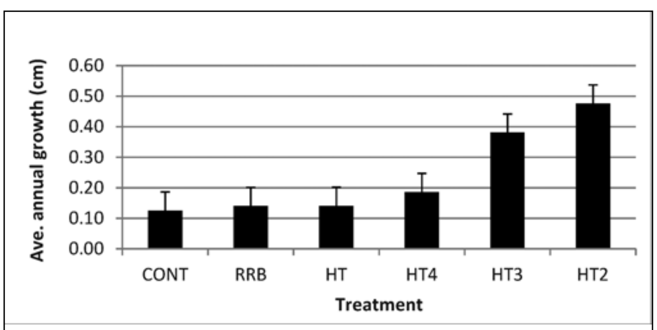

Figure 9.—Mean (± SE) blackjack oak average annual diameter growth from 1984-2010 by treatment on the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK. See Table 1 for treatment descriptions. Treatment means were not different (P=0.126).

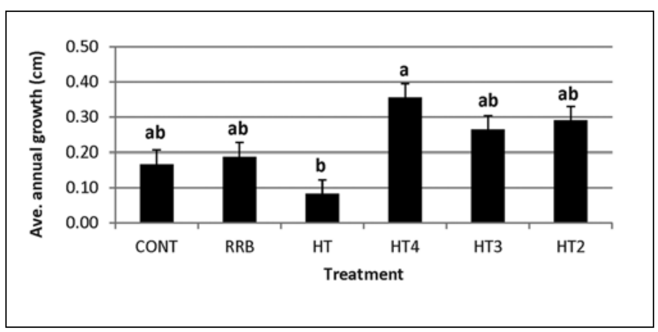

Figure 10.—Mean (± SE) post oak average annual diameter growth from 1984-2010 by treatment on the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK. See Table 1 for treatment descriptions. Treatment means with the same letter were not different (P=0.031).

thinning shortly thereafter. That year was a bumper acorn production year with an average of 105.2 kg ± 17.7 per tree across all units and the highest during the study years. Only in 1 year (1988) did post oak acorn production differ (*P*=0.0099) between treatments (Fig. 11). Here the thinned but unburned treatment (HT) had the highest production. The previous year showed a strong similar tendency in acorn production by treatment (*P*=0.071) with the HT treatment again being highest. Post oak acorn production varied widely by year exhibiting a strong year effect (*P*<0.0001) with 2 years of the 7 producing an average of <1 kg of acorns per tree across all treatments. Another year had very low production, and 2 years exhibited moderate production, one of which was 1988 (Fig. 9). We observed no year by treatment interaction (*P*=0.381) but did observe a tendency (*P*=0.115) for the HT treatment to have higher post oak acorn production when analyzed across all years, with the burned treatments similar in production to the control.

We found no differences in acorn production by blackjack oaks in any year as a response to thinning and burning. However this may have been a result of high fire-caused mortality following the first burn and mortality from canker on unburned controls, thus lowering sample tree number and number of replications for some treatments. When analyzed across all years that acorn production was monitored, we observed a strong tendency (*P*=0.052) for a treatment effect wherein the thinned (burned or unburned) treatments ranked higher in production than the control. Some blackjack acorn production, although somewhat limited at times, occurred in every year (*F*-test for year effect; *P*=0.086).

Nutrient content of sampled acorns generally varied more by tree within treatment units than between treatments (*P*>0.05) (Table 3). The only exception was moisture content which was different (*P*=0.030) depending on the time since burned and number of times burned (Table 3). Acorns from the unburned control and the burned unit three-growing seasons postburn had higher moisture content than the more frequently burned treatments. Crude protein was elevated slightly (*P*=0.120) in more frequently burned treatments.

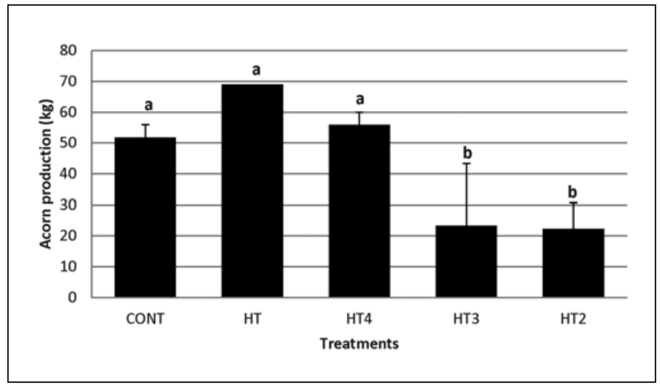

Figure 11.—Mean (± SE) post oak acorn production in 1988 by treatment on the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK. See Table 1 for treatment descriptions. Treatment means with the same letter were not different (*P*=0.0099).

Landscape Treatment and Wildlife Response

When initial thinning efforts were applied on a landscape level on PWMA beginning in 1978, fire frequency was >4-year intervals and inadequate for maintenance of savanna and woodland structure. Based on small plot study results, we began landscape application of frequent fire on a 1-3 year cycle in 1997, increased thinning in 1999-2001, then restoration thinning on the landscape in 2008 through present (Masters and Waymire 2012). By 2010, appropriate thinning regimes had been applied on 3,704 ha or 47.5 percent of the area. Since 1999 we have averaged 2,460 ha of prescribed burning annually. Both deer and elk populations have responded dramatically to landscape level application of thinning to presettlement basal areas and application of frequent fire across the area. Deer populations have increased from 7.3 deer/km^2 ± 3.8 in 1984 to 35.8 deer/km^2 ± 9.6 in 2010. However we note that since 1999 the average number of deer observations during spot-light counts has increased dramatically, partially a result of day-lighting of roads (complete tree removal 20-40 m

Table 3.—Mean nutrient content (% dry matter) of post oak acorns from control and thinned and burned units (1, 2 and 3 growing seasons postburn) from the Forest Habitat Research Area, Pushmataha Wildlife Management Area, OK in 1988. See Table 1 for general treatment descriptions, except that numbers represent number of growing seasons (gs) post-burn. Differences were not significant other than for percent moisture content. Means with the same letter or no letters were not different (*P* > 0.050).

	Number of Burns	Months Since Burns	Mean	SD	Max	Min
Percent Moisture						
CONT	0	>200	48.1 a	4.6	55.3	39.4
HT1 gs	2	7	40.6 b	1.9	42.1	38.5
HT2 gs	2	20	32.3 c	9.7	40.2	21.5
HT3 gs	1	42	46.8 a	7.3	55.0	40.9
Crude Protein						
CONT	0	>200	5.0	0.75	5.9	3.6
HT1 gs	2	7	6.0	0.90	6.6	5.0
HT2 gs	2	20	6.0	0.70	6.7	5.3
HT3 gs	1	42	4.8	0.89	5.5	3.8
ADF			27.9	5.5	38.0	19.5
Ash			2.7	0.3	3.4	2.2
TDN			70.5	5.0	78.1	61.2
Ca			0.44	0.13	0.86	0.29
P			0.09	0.01	0.12	0.07
M			0.10	0.03	0.20	0.08
K			0.77	0.17	1.23	0.55

either side of the road), thus inflating estimates to some extent. Never-the-less the trend is clear. Elk index numbers have increased from 6 in 1984 to 50 in 2010 (Masters and Waymire 2012).

On the PWMA portion of the BBS route from 1994-2009, a total of 70 different songbird species were reported. Average species richness per year was 39 ± 3. Of the 20 target species we selected, 8 were considered woodland-grassland associates, 6 were considered forest-shrub (edge) associated species, and 6 were considered forest interior species. All songbirds from the woodland-grassland species group except the pine warbler showed an increasing trend (Table 4). The trend was level throughout the study period. Of particular note was the strong increase for Bachman's sparrow, a species of special management concern across the southeastern United States, and in Oklahoma and Arkansas in particular (Cox and Widener 2008). The yellow-billed cuckoo was the only forest-shrub associated species to demonstrate a downward trend following landscape treatments. In this group, notable increases over the 16 year time-span for blue grosbeak and common yellowthroat were observed. For the forest interior group the summer tanager showed an increasing trend while the tufted titmouse, downy woodpecker and red-eyed vireo were level and showed no influence from the application of restoration treatments. On the other hand, we observed declining trends for only black & white warbler and chuck-will's-widow (Table 4).

Table 4.—Breeding bird trends of selected species from Pushmataha Wildlife Management Area, OK, 1994-2009 following landscape level application of restoration treatments. Trends are derived from only the Breeding Bird Survey Route data for Pushmataha County that occur on the Pushmataha Wildlife Management Area. A total of 37 stops out of the 50 for the county occur on the area. Key to trend symbol: + = increasing trend, – = decreasing trend, o = no or level trend.

Species Group/Species	r^2	Trend
Woodland-grassland		
Bachman's Sparrow[a]	0.61	+
Chipping Sparrow	0.12	+
Eastern Wood Pewee[a]	0.64	+
Indigo Bunting	0.52	+
Northern Bobwhite[a]	0.14	+
Pine Warbler	0.00	o
Prairie Warbler[a]	0.39	+
Red-headed Woodpecker[a]	0.12	+
Forest-shrub		
Blue Grosbeak	0.77	+
Blue-gray Gnatcatcher	0.40	+
Common Yellowthroat	0.81	+
Mourning Dove	0.37	+
Yellow-billed Cuckoo	0.22	–
Yellow-breasted Chat	0.26	+
Forest interior		
Black & White Warbler	0.62	–
Tufted Titmouse	0.00	o
Downy Woodpecker	0.10	o
Red-eyed Vireo	0.04	o
Summer Tanager	0.43	+
Chuck-will's-widow	0.46	–

[a] Species of special management concern in southeast U.S.

DISCUSSION

Presettlement forests in the Ouachita Highlands of eastern Oklahoma and western Arkansas were characterized by low-density and low basal area stands from open woodland to savanna-like conditions. They included a variety of forest cover types that expressed dominance according to topographic position and site conditions including shortleaf pine-bluestem, mixed oak-pine woodlands, and post-oak dominated savannas or oak woodlands (Foti and Glenn 1991, Masters and others 2007). Fire was a common factor that influenced density and likely a given stand's composition (Masters and others 1995, 2007).

Forest thinning and periodic fire is useful for returning natural stands to structural characteristics similar to presettlement times and managing for the full complement of native wildlife species (Masters and others 1996, 2002; Wilson and others 1995). However, following thinning the reintroduction of fire into a stand long-excluded from fire should be undertaken with some degree of understanding. Otherwise extensive mortality of the residual trees may result. Wade and Johansen (1986) concluded that reintroduction of fire into stands long excluded from fire may cause delayed mortality from stem girdling because low intensity fires may smolder in the accumulation of sloughed bark and litter around the immediate bole of the tree. Tree mortality was related to bark thickness, moisture content of the tree (Martin 1963), tree diameter, fire intensity, and residence time (Wade and Johansen 1986). Large-stemmed oaks are very resistant to fire induced mortality (Garren 1943, Komarek 1981, Kucera and others 1963, White 1986). This is due to the fact that the time for the cambium of trees to reach lethal temperatures increases with bark thickness (Hare 1965), and bark thickness increases with age (Davis 1953). Fire intensity is a causative factor in tree mortality, and mortality is determined largely by the extent of bark char and crown damage (Cain 1984) and is a function of fire type (backfire vs. headfire) (Fahnestock and Hare 1964) and season of burn (Waldrop and Van Lear 1984). The height of crown scorch on residual trees is a geometric function of fire intensity (Van Wagner 1973).

We observed a mortality pulse following introduction of fire into our experimental stands. This pulse of mortality affected blackjack oaks more severely than post oaks (Fig. 6) and was likely related to high amounts of residual logging slash, related fire behavior, and perhaps the shock of fire introduction after a long absence (Wade and Johansen 1986). However, throughout the study a higher proportion of total blackjack oak mortality was from disease rather than fire compared to post oak. Canker mortality was found in unthinned stands except for one tree in the harvested and thinned but unburned treatment (HT).

This treatment had increased in basal area 3-fold over the 25 years post-thinning. It is of note that mortality in the unthinned RRB treatment (4-year interval) was reduced compared to the control. In this treatment, some reduction of midstory hardwoods has occurred over the course of our study as a result of repeated burning. As our weather and mortality data suggested, mortality in all disease cases was likely a result of competition and late summer drought stress (Conway and Olson 2004). Our evidence suggests that thinning and burning together alleviated competition and drought stress on blackjack oak.

Aside from the high mortality rate from canker, the 100 percent blackjack mortality on the HT1 treatment but none on the HNT1 treatment also was unexpected. Because of the greater canopy cover on the HNT1 treatment versus other thinned and burned treatments, there was substantially less fuel and thus lower fireline intensity for burns on this treatment. Lower fuel loads particularly adjacent to tree stems prevented stem girdling. Post oak also exhibited similar differential mortality with annual burned treatments, but otherwise mortality was increased with increasing fire frequency.

Based on mortality alone, our thinned treatments with less frequent fire intervals (3- and 4-year or no fire) would appear to be recommended alternatives for oak-savanna restoration. However, given that over 28 years of the study, unburned treatments (HT) and the HT4 treatment had developed into immature forest stands (Table 2, Figs. 3 and 4) and were not judged viable alternatives for oak savanna restoration. Burn conditions during the first burns were lower than 40 percent relative humidity at 1400 hrs and considerable crown scorch resulted in approximately 11 percent mortality that occurred over the ensuing 3 years and beyond. If burns were conducted under higher relative humidity and firing patterns used generated less intensity than strip head fires, likely mortality could have been reduced substantially.

Conventional wisdom suggests that even mature oaks may respond with increased diameter, height, crown growth, and acorn production in response to full crown release (e.g., Cypert 1951, Drake 1991) although few studies have demonstrated this empirically and none other than this one that we are aware of with post oak or blackjack oak in the western part of its range on xeric sites.

Weather conditions such as periodic ice storms and drought is a stronger determinant of individual tree growth and development and associated characteristics of diameter, height, and crown area increment than credited in the literature, especially on xeric sites. We measured increases, due to thinning and thinning and burning to some extent, in crown expansion, diameter, and height of blackjack oak during a period of above average rainfall. These increases disappeared following a period of years with short-interval ice storms and during a period of significant below average rainfall. On an individual tree basis, those blackjack oak surviving our first burns grew vigorously on the thinned and burned treatments with 2- and 3-year intervals; however, with the loss of sample size and replication we cannot state this unequivocally. Greater development in open areas would not be surprising, however, as DeSantis and others (2010) suggested that blackjack oaks needed a growing environment with high light and further were more susceptible to drought and fire. Our data suggested that surviving blackjack oaks responded favorably in diameter growth in a high light environment, but also under drought and frequent fire conditions.

Post oaks were not as responsive to full crown release as blackjack oaks in terms of height growth and crown development. Our results were somewhat confounded by a series of ice storms that severely damaged some crowns. Blackjack oak was apparently more resistant to crown damage from ice than post oaks because of the stout and short nature of their limbs and compact nature of their crowns. We did find clear indications of higher average annual growth over 28 years on thinned and burned treatments.

Acorns are an important food item in the diet of many species of wildlife. As such any change in production and nutrient content as a result of management treatments may directly affect a given wildlife species nutritional status, especially during winter. We noted decreases in post oak acorn production with thinning and frequent fire but no difference from controls with either thinning or thinning and 4-year fire intervals (HT4). We only measured acorn production during the first half of the study period. Our trees were notably slow to respond on these xeric sites. Goodrum and others (1971) found that total mast failure never occurred during their 20-year study. We observed near complete mast failure in 2 of 7 years and low to moderate production in 4 years and 1 exceptional year. These xeric and nutrient poor oak-pine habitats in mountainous terrain such as in the Ouachita Highlands may not have the capacity for acorn production, especially in drought years, as in their study. Mast shortfalls have also been documented in the Ozark Plateau region of Arkansas (Segelquist 1974). The variability of individual trees in terms of acorn production and growth is well noted in the literature (Downs and McQuilken 1944, Drake 1991, Goodrum and others 1971) and our findings were no different. However, with increased diameter growth from thinning and burning, perhaps mast production should be revisited in a later study.

All nutrient parameters for sampled post oak acorns fell close to mean values and within the range of values reported in the literature (Short 1976, Halls 1977). However, moisture content for our samples was different according to time since burned. Samples three growing seasons postburn were similar to samples from unburned control units, and those one and two-growing season postburn were lower in moisture content. Lower moisture content may have some additional nutritional benefit to wildlife as crops with lower moisture content have been shown to have lower storage loss (from rot) and higher starch (carbohydrate) content through time compared to items with higher moisture content (Hornick 1992). If the same holds true for acorns then acorns should remain sound longer and have higher carbohydrate content during the late winter period of nutritional stress. Whether this is nutritionally significant remains to be seen. Acorns are generally very high in carbohydrates and are highly digestible, particularly those in the white oak group, and thus are an excellent energy source in winter (Kirkpatrick and Pekins 2002). Energy appears to be more important than protein or other nutrients at least for bobwhites (Giuliano and others 1996) in winter. Although a slight increase (1 percent) in crude protein was noted in acorns for the first two growing seasons postburn, that small increase may not be of biological significance given that crude protein of acorns is generally low compared to other foods (Kirkpatrick and Pekins 2002).

Implications of our fire and thinning treatments for landscape implementation of oak savanna and woodland restoration suggested fire regimes longer than every year because of higher cumulative oak mortality from annual burning in low density stands (<10 m^2/ha BA). However periodic annual fire may be a suitable application in stands above 10 m^2/ha BA for maintenance of oak savanna conditions. We found that a 3- or 4-year frequency allows for oak regeneration to develop into small trees, but the longer interval also allows forest development as also reported by Burton and others (2010). Therefore a rigid 4-year frequency is likely unsuitable for savanna maintenance.

Because oak recruitment was virtually non-existent in the annually burned treatments (R. Masters, unpublished data), 2-year and 3-year intervals are recommended to maintain open woodland or savanna conditions. The fire return intervals reported for the nearby McCurtain County Wilderness Area across a variety of sites ranged from 2.1-5.6 years with a range of 1-12 years (Masters and others 1995). Because the fire regime was historically variable, we implemented a fire regime across the landscape restoration units encompassing 1-4 years recognizing that north slope hardwoods, riparian corridors, and upland drainages associated with ephemeral streams will have effectively longer intervals as the understory will burn completely only in very dry conditions. With fire intervals longer than 4 years, mesic tree species begin to invade and change the character of the stands, and thus the potential for recurrent surface fire (Burton and others 2010).

Forest health issues have been raised particularly about densification of oak woodlands and savannas and oak decline, aside from the loss of herbaceous plants and associated animal species habitats. Oak decline is ascribed to various insect and disease agents associated with approaching physiological maturity (Hyink and Zedaker 1987). Relatively little attention, however, is paid to increased tree density through time, and thus increasing competition for moisture and nutrients, as an untoward influence on disease incidence and ecosystem health of fire suppression efforts (e.g., Hyink and Zedaker 1987, Oak and others 1986).

Fire is an integral process in oak or mixed oak-pine systems. Beneficial effects on ecosystem health include thinning of trees to reduce moisture and nutrient competition, thus increasing drought tolerance and lowering incidence of disease in a given stand, and for maintaining historic structure and function (Burton and others 2010, DeSantis and others 2010). Oak savanna and woodland restoration efforts as applied on a landscape level on PWMA are similar in site location, tree density, and composition to presettlement conditions. The essential test to determine the success of restoration efforts is the effects on wildlife populations. This view of restoration success is too often overlooked.

The dramatic response of deer and elk was not surprising given that a sporadic source of nutrition from acorns was supplemented with a much greater and more consistent amount of nutritious forage. However, at this point it is not clear if acorn production was enhanced, as we had postulated, to the point that it offsets that lost from thinning.

Songbirds provide rapid feedback on the effects of forest management and re-introduction of fire. Because they are closely associated with a particular arrangement of habitat structure, they respond in relatively short periods of time, and some are associated with narrow successional windows created by postfire succession in the understory in addition to forest canopy cover change (Cox and Widener 2008, Masters and others 2002, Wilson and others 1995). Woodland-grassland and forest-shrub birds have declined precipitously across the southeast United States because of fire exclusion and densification of forests (Cox and Widener 2008, Wilson and others 1995). The frequent fire connection is essential for woodland grassland birds in order to maintain suitable habitat structure (Cox and Widener 2008).

Not withstanding the needs of these groups of birds, we were also interested in any deleterious influence on forest-shrub and forest interior birds that were species of high interest although not species of special management concern. It is significant that only three species, yellow-billed cuckoo, black & white warbler, and chuck-will's-widow, showed declines over the 16 years following staged implementation of landscape level restoration. Regional trends have a strong influence on local populations of birds. Therefore, interpretation of results on a given route must be interpreted in light of regional trends. Yellow-billed cuckoo and chuck-will's-widow experienced strong regional declines from 2004-2007 (Sauer and others 2011). However, the black & white warbler exhibited a stable regional trend during this time period (Sauer and others 2011).

Our study has shown that small plot research can inform landscape level restoration and management through confirmation of the utility of restoration targeted thinning and confirmation of fire intervals used by land management as compared to presettlement landscape structure and fire regime. Studies of presettlement landscape characteristics such as structure, form, function, and fire chronology provide essential data for guiding restoration of oak woodlands and savannas as well as for other ecosystems. Researchers and land managers would do well to explore and learn from this diverse literature.

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APPENDIX

List of common and scientific names of plants, bird, and mammal species mentioned in the text and figures.

Common Name	Scientific Name (authority)
Plants	
Big bluestem	*Andropogon gerardii* (Vitman)
Black oak	*Quercus velutina* (Lam.)
Blackberry	*Rubus* spp.
Blackjack oak	*Quercus marilandica* (Muenchh.)
Bluestem grass	*Schizachyrium* spp., *Andropogon* spp.
Greenbriar	*Smilax bona-nox* (L.)
Hypoxylon cancer	*Hypoxylon atropunctatum* (Schwein.) Cooke
Indiangrass	*Sorghastrum nutans* (L.) Nash
Little bluestem	*Schizachyrium scoparium* (Michx.) Nash
Loblolly pine	*Pinus taeda* (L.)
Mockernut hickory	*Carya alba* (L.) Nutt.
Muscadine grape	*Vitus rotundifolia* (Michx.)
Panicums	*Panicum* spp., *Dicanthelium* spp.
Poison ivy	*Toxicodendron radicans* (L.)
Post oak	*Quercus stellata* (Wangenh.)
Sedges	*Carex* spp., *Scleria* spp.
Shortleaf pine	*Pinus echinata* (Mill.)
Switchgrass	*Panicum virgatum* (L.)
Tree sparkleberry	*Vaccinium arboretum* (Marshall)
Virginia creeper	*Parthenocissus quinquefolia* (L.) Planch.
Winged sumac	*Rhus copallinum* (L.)
Birds	
Bachman's Sparrow	*Aimophila aestivalis* (Lichtenstein)
Black and White Warbler	*Mniotilta varia* (Linnaeus)
Blue Grosbeak	*Guiraca caerulea* (Linnaeus)
Blue-gray Gnatcatcher	*Polioptila caerulea* (Linnaeus)
Chipping Sparrow	*Spizella passerina* (Bechstein)
Chuck-will's-widow	*Caprimulgus carolinensis* (Gmelin)
Common Yellowthroat	*Geothlypis trichas* (Linnaeus)
Downy Woodpecker	*Picoides pubescens* (Linnaeus)
Eastern Wood Pewee	*Contopus virens* (Linnaeus)
Indigo Bunting	*Passerina cyanea* (Linnaeus)
Mourning Dove	*Zenaida macroura* (Linnaeus)
Northern Bobwhite	*Colinus virginianus* (Linnaeus)
Pine Warbler	*Dendroica pinus* (Wilson)
Prairie Warbler	*Dendroica discolor* (Vieillot)
Red-cockaded woodpecker	*Picoides borealis* (Vieillot)
Red-eyed Vireo	*Vireo olivaceus* (Linnaeus)
Red-headed Woodpecker	*Melanerpes erythrocephalus* (Linnaeus)
Summer Tanager	*Piranga rubra* (Linnaeus)
Tufted Titmouse	*Baeolophus bicolor* (Linnaeus)
Yellow-billed Cuckoo	*Coccyzus americanus* (Linnaeus)
Yellow-breasted Chat	*Icteria virens* (Linnaeus)
Mammals	
Bison	*Bison bison* (Linnaeus)
Rocky Mountain elk	*Cervus elaphus* (Linnaeus)
White-tailed deer	*Odocoileus virginianus* (Zimmermann)

FIRE-ADAPTED NATURAL COMMUNITIES OF THE OZARK HIGHLANDS AT THE TIME OF EUROPEAN SETTLEMENT AND NOW

Paul W. Nelson

Forest Ecologist, Mark Twain National Forest, Supervisors Office, 401 Fairgrounds Road, Rolla, Missouri 65401. To contact call 573-341-7433 or email at pwnelson@fs.fed.us.

Abstract.—The Ozark Highlands Plateau of southern Missouri and northern Arkansas is home to more than 2,000 vascular plant species and at least 15,000 species of various animals of which over 150 are endemic. The Nation's most significant karst region occurs here, boasting the most springs of any state and more than 6,000 caves. Missouri's Ozark biota is sorted into 65 distinctive mappable natural communities. Misconstrued as "oak-hickory forest", analysis of Missouri's historic vegetation, fire history studies, plant adaptations, and responses to fire management reveal that the Ozark's was a landscape dominated by a complex mosaic of fire and topography-mediated grass and forb-rich savannas, woodlands, glades, forests, and fens. Much of today's native Ozark vegetation is deceptively out of character for its structure, composition, species richness and former landscape patterns due to having suffered the consequences of destructive post-European settlement overgrazing, poor farming, fire cessation, and soil loss. Fallacies abound that "resilient" Ozark ecosystems will recover, succeed, and migrate in the face of climate change and homogenizing vegetation transformation. Monitoring of remnant high quality natural communities reveals their susceptibility to irreversible conservative species loss and simplification. Managers must understand and distinguish between the achievement of true ecological restoration outcomes of high quality habitats over personal/professional biases including grazing woodlands, timber production, and singular species emphasis.

BACKGROUND

This paper primarily employs the ecological classification and ecosystem restoration approaches used by the Missouri Natural Areas Committee. The committee is made up of six state and federal agencies that have cosigned a Memorandum of Understanding to administer a system of natural areas. Natural areas by definition are areas of the landscape that protect high quality aquatic and terrestrial assemblages of plants and animals as they may have existed prior to European settlement (Nelson 2010). It is this time period that sets the stage for defining and describing natural communities and the sequence of European settlement factors that have altered and damaged Missouri's historical fire-adapted natural communities.

The objectives of this paper are:

1. To describe the primary fire-adapted natural communities that occurred prior to European settlement
2. To list the modern day disturbances that have altered their historic patterns
3. To introduce some misconceptions about ecosystem restoration

Nelson (2010) describes natural communities as distinct assemblages of native plants, animals, and microorganisms that occur in repeatable (mappable) patterns across the landscape and through time. These assemblages of biota occupy definable physical environments, which in turn influence the structure and composition of natural communities. In Missouri, 85 natural communities are described with approximately 65 occurring in the Ozarks. These natural communities fall into broad types including forest, woodland, savanna, glade, prairie, wetland, stream bank, cliff, and cave. High quality natural communities are generally rich in native plant and animal species including a host of "conservative species" that are sensitive to modern disturbances. Conservative species decrease or disappear in ecosystems degraded by clearing, plowing, overgrazing, over browsing, and fire cessation resulting in a loss of biodiversity. As many as 300 to 400 vascular plant species may occur in one natural community type. A second attribute of high quality natural communities is the abundance of flowers, seeds, and biomass cover produced by the rich groundcover plant species. In turn, a great variety of host specific and other invertebrates inhabit this ground layer, interacting as plant pollinators and providing food for predators higher in the food chain.

High integrity natural communities are characterized by woody vegetation of variable ages and patterns as determined and shaped by the presumed historical disturbance regime and climate. This includes differences in gap size, blow down, old growth character, spacing between trees, and the presence or absence of large dead or decaying wood. Ozark natural communities were further characterized by variable but connected landscape patterns differentiated by the distinctive features of a given ecological subsection. Those features included topographic expressions, bedrock type, drainage patterns, and disturbance processes.

Each natural community type is classified according to dominant or characteristic vegetation types, canopy height and openness, vegetation layers, rock type, age class variation, and disturbance processes. The author (and the Missouri Natural Areas Committee) recognizes the temporal nature of natural communities and the changes brought about by variations in climate and other disturbance processes. For the purposes of this paper, the Missouri Natural Areas Committee uses the pre-European time period as a reference point to differentiate the contemporary altered landscape from high quality ecosystems prior to their severe, modern day disruption (Nelson 2010).

THE OZARK LANDSCAPE SETTING

The Ozarks ecoregion encompasses nearly 34 million acres across the southern one half of Missouri, northern Arkansas, and small portions of Oklahoma and Illinois. This distinctive biogeographic region of southern Missouri and much of northern Arkansas is a low structural dome of variably aged rock strata with the dome center consisting of the oldest (1.5 billion years) igneous rock in the St. Francois Mountains (Nigh and Schroeder 2002). This elevated region of the central United States along with the Ouachita Mountains to the south is the highest elevated region of mid-continental North America. A quarter billion years of exceptional geologic erosion, wind transport, and subterranean karst dissolution has created a diversity of landforms that vary in degree of relief, dissection, and geologic parent materials. Ozark biota are characterized by an unusually high level of species disjuncts and endemism with more than 160 species endemic to the area. Among the diversity of plants and animals, 159 species are endemic, 77 are modal (meaning their primary population density exists there), 81 are globally rare, and 58 are disjunct. Of this number, 32 species are in decline with perhaps a dozen extirpated (Ozarks Ecoregional Assessment Team 2003).

Differences in landform, soils, and vegetation produce 16 ecological subsections in the Ozark Highlands of Missouri (Nigh and Schroeder 2002). These ecological subsections and their geologic, biologic, topographic, and historical Native American interactions prior to European settlement created mosaics of diverse fire-mediated prairies, savannas, woodlands, glades, fens, and forests.

OZARK VEGETATION SHAPED BY HISTORIC FIRE

Native Americans played a role in shaping changes across the Ozarks for as long as 12,500 years (Ray and others1998). When Europeans first arrived in Missouri, high integrity ecosystems had evolved with adaptations to disturbance processes. Nelson (2010) provides an in-depth accounting of the effects of Native American habitation ranging from the landscape application of fire to agriculture and settlement. The environment was radically different during the period between 8,500 and 4,500 years before present (YBP) when drought-adapted species expanded their range, and grasslands (among woodlands) dominated. Following 4,500 YBP, essentially modern vegetation and faunal patterns appeared. Extensive evidence of past fires started by Native Americans abound. Guyette and Dey (2000) and Guyette and others (2006) provide accurate records in the modeling of historic fires across North America and the relationship between historic human habitation, topographic relief, and climate. Abrams and Nowacki (2008) concluded that Missouri's subhumid Midwestern climate would not support prairie, savanna, and open woodland without the influence of Native Americans broadcasting fire over North America the past 5,000 years. Thus, the 65 distinct natural communities of the Missouri Ozarks were shaped by centuries of variable fire regimes.

THE OZARKS VEGETATION-DIVERSE ASSEMBLAGES OF FIRE-ADAPTED NATURAL COMMUNITIES, NOT OAK-HICKORY FORESTS

The mindset of early natural resource management by state and federal agencies was shaped throughout the early 1900s by the notion that fire was detrimental for plants and wildlife. Dr. Julian Steyermark who wrote "Flora of Missouri" (1963) and "Vegetational History of the Ozark Forest" (1959) falsely shaped this mindset by furthering the theory that the Ozarks was primarily a forested landscape succeeding from an earlier open savanna-woodland vegetation type. He attributed the "unnatural succession" of former open grass-dominated Ozark woodlands and savannas to the eventual mesification effects of a warmer-wetter climate following the xerothermic period (a postglacial interval of a warmer, drier climate that occurred 4,000-5,000 years ago). Dr. Steyermark never made the connection between the role of Native Americans in their broadcasting fire across all of North America and its profound effects in shaping fire-adapted natural communities. His reports to the Mark Twain National Forest and state foresters in the mid 1950s furthered the notion that post-European annual burning of the Ozarks was destructive to native flora. Later, natural features inventories conducted by the Missouri Natural Areas Committee (listed in Nelson 2010) would demonstrate that it was the combination of open-range overgrazing coupled with annual burning of the now logged over Ozarks that led to the widespread loss of the once rich grass-forb groundcover associated with ancient open woodlands, savannas, and glades.

Many early explorers in Missouri chronicled numerous accounts of periodic fires and their influence in shaping the open, park-like character of the Ozarks. Schoolcraft (1819) observed, "The Ozarks…covered with a very uniform growth of black oaks and post oaks, in the summer season by a vigorous growth of wild grasses, flowers and vines." Louis Houck (1908) states, "Open woods and a growth of wild prairie grasses and flowers filling the broad spaces between the trees. All the forests were free from undergrowth, and open and park like in appearance." George Swallow (1859) writes, "This growth is not due to the poverty of the soil but to the fires which have annually overrun this country since the earliest dates of the Indian traditions…" Table 1 shows the transitional continuum and breaking points between the primary fire-adapted natural communities in the Ozark Highlands.

Prairie

While over 13 million acres of prairie occurred across Missouri, its presence in the Ozarks is restricted primarily to the Central Plateau Subsection. Prairies were essentially treeless, primarily dominated by a

Table 1.—The Prairie, Savanna, Woodland to Forest Continuum for the Ozark Highlands

Attributes	Mesic Forest	Dry-mesic Forest	Dry-mesic Woodland	Dry (open) Woodland and Flatwood	Savanna	Prairie and Glade
Vegetation Layers	Multiple >4	Multiple >3	2 to 3	2 to 3	2	1
Age Classes	Variable - uneven age	Variable - uneven and even	Uneven/even	Multiple even age	Multiple even age	Not applicable
Tree Form/ Height	Narrow crowns, clean trunks 90+	Narrow crowns, clean trunks 70-90	Somewhat spreading crowns, clean trunks 60- 90	Spreading crowns and lower branches 20-60	Wide-spreading crowns, 20-60	Not applicable
Canopy Closure	90-100	90-100	80+	30-80	10-30	0-10
% Understory Cover	50-100 dense	50-100 dense	30-50 patchy	10-30 scattered	5-10 sparse	0-10
Ground Layer Cover	Dense in spring, patchy to sparse by mid-summer	Dense to patchy in spring, patchy to sparse by mid-summer	Dense to patchy in spring, patchy to dense by mid-summer	Patchy to dense all season	Dense all season	Dense all season
Ground Layer Plants	Rich diversity of spring ephemerals, and ferns; few summer/fall forbs	Moderate to low diversity of spring ephemerals, and ferns, few summer/fall forbs	Moderate to low diversity of spring ephemerals, and ferns; abundant C3 grasses, sedges, and summer/fall forbs	C3 and C4 grasses, sedges, diversity of forbs all season	C4 grasses, sedges, diversity of forbs all season	C4 grasses, sedges, diversity of forbs all season
Topography and Landform	Protected valleys, ravines, bluff bases, lower slopes of northerly aspects, fire shadow areas	Mid and upper slopes of northerly aspects, ravines, fire shadow areas	Mid and upper slopes of southerly aspects, fire prone landscapes	Steep upper slopes of southerly aspects, narrow ridges, broad ridges, fire prone landscapes	Level to gently rolling topography, steep loess hills, broad ridges	Level to gently rolling plains, steep loess hills, broad ridges; steep sw slopes-glades
Soils	Deep (>3') loams, nutrient rich, high organic matter	Moderate depth (24-36") silt loams, moderate organic matter	Moderate depth (20-36") silt loams, moderate organic matter	Shallow depth (<20"), droughty, often rocky and or nutrient poor	Wide range of soil types from shallow to deep, variably rocky	Wide range of soil types from shallow to deep, variably rocky
Fire Regime	Very infrequent (30+ years), and/or low intensity fires	Infrequent (20+ years), and/or low intensity fires	Low to moderate intensity fires every 3 to 15 years	Low to moderate intensity fires every 3 to 5 years	Moderate intensity fires every 1 to 3 years	Moderate to high intensity fires every 1 to 3 years
Dominant Trees	Red oak, sugar maple, ash, basswood, walnut	White, red, and black oaks, hickories	White, black, scarlet, chinkapin oaks, hickories, shortleaf pine	Post, blackjack, chinkapin, bur, white oak, pine	Bur, chinquapin, swamp oak, white oak	Shrubs
Basal Area # mature trees/ acre	70-80 >30	70-80 30	60-80 20-30	30-60 <10	<30 <5	<10 <2

mix of grasses, forbs, and low shrubs. The often level to gently undulating plateau was conducive to rapidly spreading, intense fires occurring as often as every 2-3 years, thus limiting tree growth. Only one occurrence of a remnant Ozark prairie exists today.

Savanna

Some 6.5 million acres of savanna occurred across the Ozarks, primarily on the Springfield Plateau and Central Plateau Subsections. Savannas were Serengeti-like landscapes of widely spaced, orchard-like trees of variable age classes ranging from old growth trees to islands and scatterings of oak regeneration. Tall grass species and forbs dominated the groundcover. Principle tree species included post oak, white oak and bur oak (scientific names listed in Appendix). The greater undulation and fragmentation of plateaus and plains by streams and rivers decreased the effects of rapidly spreading fires, thus leading to the increase in tree species, particularly post and white oak. Unfortunately, very little savanna remains in the Ozarks, having succumbed to exotic cool season grasslands used for pasture.

Woodland

By far the most diverse, 13 woodland types occur across the Ozark Highlands. This was the predominant natural community type with at least 11 million acres occurring. Fortunately, this is also the most recoverable, restorable natural community in the Ozarks with the potential of restoring 6 million acres (Spencer and others 1992). Woodland types often have a variable canopy openness ranging from 30 to 90 percent canopy cover depending on restoration objectives and fire effects. Important woodland types include post oak-black oak woodlands, mixed oak woodlands, shortleaf pine-bluestem woodlands, chinquapin oak woodlands, and bottomland woodlands. Many examples of woodland restoration abound across the Ozark Highlands as agencies and conservation groups continue actively restoring landscapes. The Central Hardwoods Joint Venture considers woodlands, particularly their early seral, grass-dominated patterns, vital toward the recovery of early seral bird species including blue-gray gnatcatcher, indigo bunting, prairie warbler, chipping sparrow, field sparrow, and bobwhite quail. Vaughn (personal communication) demonstrated the importance of the attainment of quality restored woodland vegetation structure and the recovery of diverse bird communities.

Forests

In the general absence of fire, trees formed a closed canopy interspersed with multi-layered shade-tolerant subcanopy trees, shrubs, vines, and scattered herbs. Trees attained their greatest height, forced to grow high, often competing for light in forest gaps. Ground flora often consisted of a rich assemblage of fire-intolerant spring ephemeral forbs, shade-tolerant sedges, grasses, and ferns. Much of Missouri's woodland landscape was regarded as "forest" (Braun 1950, Steyermark 1959) when in fact the present-day closed, dense canopy is an artifact of domestic livestock overgrazing and fire suppression (Pyne 1982). In the Ozarks, forest is confined to the most rugged, deeply dissected hills and breaks of the Current River and Eleven Point River watersheds with lesser amounts in protected valley coves, large river floodplains, and the base of bluffs elsewhere (Nigh 2002).

Glades

Five glade natural communities occur in the Ozark Highlands. Glades are essentially treeless shallow bedrock openings in woodlands ranging in size from one-half acre to 1,500 acres. Despite shallow bedrock, as many as 400 vascular plant species inhabit glades, including several endemic and restricted species. The extremely dry, desert-like condition of glades attract animal species with a primary distribution in the more arid southwestern United States including the eastern collared lizard, flat-headed snake, southern coal skink, tarantula, and greater roadrunner. The Ozark Highlands contain the highest number and acreage of glades in Eastern North America (Nelson and Ladd 1983). Approximate glade acreages and types, according to Nelson and Ladd, include: 200 acres of chert glades (Springfield Plateau only); 5,000 acres of limestone glades (Springfield Plateau); 12,000 acres igneous glades (St. Francois Mountains); and 200,000 acres dolomite glades (located throughout the Ozark Highlands but greatest abundance in White River Hills, Osage River Hills, and Inner Ozark Border). Guyette (1982) estimated historical fire to occur every 3-4 years on glades in southwest Missouri.

NATURAL COMMUNITY PATTERNS ACROSS THE OZARK HIGHLANDS

Early interpretations of the falsely applied concept of "Oak Hickory Forest" portrayed the Ozarks as covered primarily in forest and dominated by mixtures of oak and hickory. Modern interpretations now cite several valuable sources. The Missouri Historic Vegetation Survey, Geographic Resources Center, University of Missouri, Columbia has attributed the characteristics of over 400,000 individual trees recorded by Government Land Office early land surveyors during the early to mid 1800s. The Geographic Resources Center has queried these data across a variety of stratified applications, including relative importance values for all tree species by ecological subsection, the relative openness of the tree canopy, and tree associations. Results reveal that the Ozark Highlands contained over 25 different tree associations, many of them attributed to the influences of fire, topography, and geology (Batek and others 1999). Oaks were important but so was shortleaf pine. Hickory was not an important (dominant) tree species. When modeling witness tree structure and openness (a measure of distance and diameter from section corners and section lines), much of the Ozarks was open in character, thus confirming the historical presence of savanna and open woodlands. Again, forest was confined to dissected river breaks. Table 2 shows the most important oak-dominated natural communities, landforms, and estimated fire interval. Table 3 lists the oak species known to occur in the Ozark Highlands and their optimal natural community (Nelson 2010).

THE PRESENT-DAY OZARK HIGHLANDS: OUT-OF-CHARACTER AND DEGRADED

Biotic homogenization (McKinney and Lockwood 1999) happens when native localized ecosystems are assimilated by widespread exotic or weedy native species, thereby increasing their compositional similarity. Biotic distinctiveness gradually dissolves. An example might be the replacement of 300 native plant species associated with a high quality, pre-European glade by 40 native species and dozens of exotics within the same landscape area. Present day homogenization processes include:

- Importation of exotic plants for agriculture, wildlife plantings, and landscaping/gardening
- Globalized transport of exotic pets (legal and illegal) along with accidental insect/diseases

Table 2.—Important oak-dominated natural communities of the Ozarks, their landform and presumed fire interval

Communities	Landform	Fire Interval
Post oak bluestem flatwoods	Gentle fragipan plains and ridges	Fire 3-5 years
Post oak and bur oak savanna	Gentle plains	Fire 2-4 years
Chinquapin oak bluestem woodland	Dolomite slopes and hills	Fire 3-10 years
White, black, and blackjack oak woodlands	Chert plains	Fire 3-5 years
Blackjack oak xeric woodland	Igneous rock outcrops	Fire 5-15 years
Post and chinquapin oak-smoketree glades	Dolomite bedrock of dissected hills	Fire 3-4 years
White oak-pine woodlands	Dissected hills and breaks	Fire 10-50 years
Mixed bottomland oak forests	Larger river floodplains	Fire 25-50 years
White, northern red oak/sugar maple forest	Deep coves and valleys	Fire 15-30 years

Table 3.—List of Missouri oak species and their optimal natural community type (Nelson 2010)

Oak species	Savanna	Open Woodland	Closed Woodland	Flatwoods	Upland Forest	Bottomland Forest	Glade
Blackjack oak		x					
Black oak		x	x				
Bur oak	x					x	
Cherrybark oak			x		x		
Dwarf chestnut oak	x						
Chinquapin oak		x					x
Nuttall's oak						x	
Overcup oak						x	
Pin oak						x	
Post oak	x	x		x			x
Red, northern oak			x		x		
Red, southern oak	x	x					
Scarlet oak		x	X		X		
Schneck's oak							x
Shingle oak						x	
Shumard oak						x	
Swamp white oak						x	

- Land fragmentation and parcelization
- Overhunting and wildlife persecution
- Agriculture
- Fire suppression
- Overgrazing (past and present)
- Modern developments; housing, reservoirs, roads, industry
- Variations in management philosophies

The consequences of these processes have impacted the Ozark Highlands in the following primary ways:

1. Overgrazing has decimated the once lush mantle of rich grass/forb groundcover and resulted in soil loss-erosion of gravel into Ozark streams (Smith 2003).
2. Soil erosion has exasperated watershed hydrology degradation with runoff increasing (even in dense, overstocked canopies) and subsequent landscape dehydration.
3. Loss of groundcover species has dramatically reduced or eliminated flower and seed production, thereby decreasing the abundance of insect populations important for foraging by birds and bats (Beilmann and Brenner 1951).
4. Fire suppression coupled with grazing and logging has led to dense overstocking of increasingly undesirable tree species (Ladd 1991, McCarty 1998, Nigh 1992).
5. Loss of topsoil has changed soil chemistry and dehydrated the landscape causing increased runoff and decreased spring flows.
6. Black and scarlet oak (preferred species for timber markets) have increased, but so have oak decline, oak wilt, and oak borer diseases (Law and others 2002, USDA 2005).
7. Exotic species are on the increase in especially vulnerable damaged woodlands. Of Missouri's 800 introduced nonnative plant species, 32 are recognized as invasive on the Mark Twain National Forest (USDA 2012). Feral hogs are increasing across the Ozarks.
8. Certain native species populations are out of balance including invasive red cedar on glades and isolated occurrences of heavy white-tailed deer overbrowsing. Rooney (2004) attributed biotic simplification of 62 forests in Wisconsin to the keystone effects of too many deer.

9. There is an increase in the listing of species of conservation concern, particularly those once associated with fire-adapted natural communities.

Over 75 percent of Missouri's 44 million acres of historic vegetation is completely transformed; most destroyed. The remaining 25 percent is fragmented, out-of-character, damaged, fire-suppressed, and subject to exotic species invasion. Of this percentage, the following acreages of various oak-dominated natural communities are estimated to be restorable to high quality natural community standards on the Mark Twain National Forest: 4,000 acres of post oak bluestem savanna; 15,000 acres post oak-chinquapin oak dolomite glade and woodland; 2,000 acres post oak igneous glade; and 150,000 acres of various oak and pine woodland types. The most threatened oak systems across the Ozark Highlands include mixed oak bottomland forests with few high quality examples occurring anywhere.

MISCONCEPTIONS OF ECOSYSTEM SUCCESSION AND MIGRATION

The old botanical theories of plant succession, species migrations, and reaccrual of species-rich ecosystems on the move in the face of climate change are not going to operate effectively, at least in the Midwest. Rooney (2004) and others observed that in the face of changing vegetation structure and patterns, species richness declines and there's no corresponding accrual of new species adapted to the different environment. We do not have any examples of an unmanaged savanna, woodland, or prairie that, when left to its own devices, has recovered or succeeded to an equally species rich ecosystem. Also a myth is the old concept that natural communities will migrate or succeed in response to shifts in climate (Steyermark 1959).

CONSERVATION STRATEGIES

The Nature Conservancy's Ozark Ecoregional Conservation Assessment (Ozarks Ecoregional Assessment Team 2003) has identified high priority portfolio areas within the Ozark Highlands to concentrate protection/management strategies. The Mark Twain National Forest used this assessment to design and delineate 19 management areas across distinctive ecological regions of the Mark Twain, emphasizing restoration of ecosystems. The 2005 Forest Management Plan for the Mark Twain identified objectives to restoration more than 150,000 acres to desired conditions for healthy natural communities. The Mark Twain has also increased prescribed burning to 40,000 acres annually. At the 2009 Missouri Natural Resources Conference, a multi-agency workgroup concluded that the best conservation strategy in response to climate change was to make ecosystems more resilient by restoring their historical condition and maintaining or emulating critical disturbance processes, especially fire.

CONCLUSIONS

In conclusion, much of the Ozark landscape was mantled in a park-like growth of mixed oak and pine interspersed with a nearly continuous ground cover of deep rooted perennial grasses and forbs. Most vegetation composition and structure directly reflected the effects of variable fire regimes, excepting the most deeply dissected river breaks. Open range overgrazing, soil erosion, poor farming, and fire suppression have obliterated the historic character of former fire-mediated woodlands, savannas, and glades resulting in population shifts for many plant and animal species. Hands-off management will result in declines in species richness for landscapes once dominated by fire-adapted ecosystems due to the homogenization process. Threats abound. To make matters worse, the preservation of biodiversity through ecosystem restoration is not a primary management goal of most land managing agencies. Managers differ in their land management aspirations, personal interests, philosophies, and management styles.

We have the resources: people, money, time. We have strategies: state park resource stewardship plans and policies, Missouri Department of Conservation Comprehensive Wildlife Strategy, Mark Twain National Forest Plan ecosystem restoration objectives, and The Nature Conservancy's Ozark Ecoregional Conservation Assessment. But the statewide scorecard for ecosystem restoration work accomplished is far from achieving its objectives. Resistance prevails in many forms: budget cuts, layoffs, other priorities, manager philosophies, changing leadership, and government agency decentralization.

Perhaps toughest of all is our not accepting that sensitive plant and animal populations and quality ecosystems will in many places fall victim to the homogenization process. Protecting ecosystems is time consuming, expensive, and requires a constant dedicated commitment. This commitment requires that we institutionalize the idea of ecosystem restoration as a primary driver behind what land managing agencies and private nature organizations do. But changing politics, new leaders with their own aspirations, and the eventual "retirement" of those with passion and initiative to advocate make this extremely challenging.

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APPENDIX

List of scientific plant names (Yatskievych and Turner 1990) for cited common plant names.

Common name	Scientific name	Common name	Scientific name
Blackjack oak	*Quercus marilandica*	Post oak	*Quercus stellata*
Black oak	*Quercus velutina*	Northern red oak	*Quercus rubra*
Bluestem	*Andropogon* species	Scarlet oak	*Quercus coccinea*
Bur oak	*Quercus macrocarpa*	Schneck's oak	*Quercus schumardii* var. *schneckii*
Cherrybark oak	*Quercus pagoda*	Schumard's oak	*Quercus schumardii*
Dwarf chestnut oak	*Quercus prinoides*	Shingle oak	*Quercus imbricaria*
Chinquapin oak	*Quercus muehlenbergii*	Shortleaf pine	*Pinus echinata*
Nuttal oak	*Quercus texana*	Smoketree	*Cotinus obovatus*
Overcup oak	*Quercus lyrata*	Sugar maple	*Acer saccharum*
Pin oak	*Quercus palustris*	Swamp white oak	*Quercus bicolor*

FIRE CHRONOLOGY AND WINDSTORM EFFECTS ON PERSISTENCE OF A DISJUNCT OAK-SHORTLEAF PINE COMMUNITY

Michael D. Jones and Marlin L. Bowles

Research Associate (MDJ), The Morton Arboretum, and Christopher B. Burke Engineering, Ltd.; Plant Conservation Biologist (MLB), The Morton Arboretum, 4100 Illinois Rt. 53, Lisle, IL 60532. MLB is corresponding author: to contact, call 630-719-2422 or e-mail at mbowles@mortonarb.org.

Abstract.—We investigated effects of a human-altered fire regime and wind storms on persistence of disjunct oak-shortleaf pine vegetation occurring along 5.5 km of xeric habitat on the east bluffs of the Mississippi River in Union County, IL. In 2009, we resampled vegetation transects established in seven stands in 1954 and obtained 26 cross sections containing fire scars from pines downed by windstorms in 2008 and 2009. These scars revealed 81 years with fires between 1767-1991. Only presettlement fire years corresponded with drought conditions indicated by the Palmer Drought Severity Index. After settlement, fire return intervals measured within stands increased significantly from 4.55 years during the prelogging era (1825-1875) to 8.19 years during the postlogging era (after 1935). After 1975, only one wildfire and two of three management fires were detected. Pine recruitment and the percentage of scarred trees, a proxy for fire intensity, peaked during the prelogging era when both pine and oak recruitment were also positively correlated with fire frequency. Shortleaf pine (*Pinus echinata* Mill.), black oak (*Quercus velutina* Lam.), and post oak (*Quercus* stellate Wangenh.) were codominant in 1954, but by 2008, pine had become subdominant to black oak, and post oak had declined significantly. The 2008-2009 windstorms apparently compensated for recent fire exclusion by selective removal of black oak, allowing pine to regain codominance. At this site, longevity of shortleaf pine coupled with periodic canopy disturbance and favorable fire history appear to have been critical factors maintaining this species in optimum xeric habitat. However, unless fire processes are restored, post oak and other shade-intolerant species may disappear, and shade-tolerant woody species recently established in the understory may further influence future canopy composition.

INTRODUCTION

Fire and windstorms were two of the most important disturbance processes affecting composition and structure of presettlement forests (e.g., Frelich and Lorimer 1991, Peterson and Pickett 1995, Schulte and Mladenoff 2005). Postsettlement fire suppression has significantly altered oak forests by reducing oak regeneration and allowing establishment of fire-intolerant woody vegetation; however, this effect may be moderated on drier habitats (Abrams 1992). As a result, understanding how human-altered fire regimes and their interactions with other disturbances affect long-term vegetation change is critical for management of disturbance-dependent woody plant communities. To resolve this need, managers often seek information on fire history dynamics and reference stand conditions to develop management and restoration goals (Stambaugh and Guyette 2006, Swetnam and others 1999). Fire chronologies determined from tree rings and fire scars can reveal historic fire regimes that have maintained fire-dependent woody plant communities and can help set management strategies to maintain this vegetation (Wolf 2004). For example, Midwest fire chronologies for oaks have revealed comparatively long fire return intervals associated with landscape-scale fires before and during early settlement, followed by a shift toward more localized fire occurrences and fire suppression (reviewed in McClain and others 2010). Likewise, in the Missouri Ozarks, shortleaf pine (see Appendix I for nomenclature and common names) fire scars indicate that presettlement fires occurred during years of extreme drought, while more frequent postsettlement fires, as well as fire protection, mask the influence of drought (Engbring and others 2008, Stambaugh and Guyette 2006).

Severe windstorms alter stand structure by reducing basal area and increasing canopy openness (Hanson and Lorimer 2007). They range from narrow tornado paths or downbursts to wide-scale straight-line winds or derechos (Peterson 2000). Impacts correlate with wind speed and vulnerability due to topographic relief and range in scale from small gaps that are captured by advance regeneration of shade-tolerant species to stand-wide disturbances that may reset or advance successional stages, depending upon understory vegetation (Arevolo and others 2000, Canham and others 2000, Dunn and others 1993, Peterson 2000, Peterson and Pickett 1995, Van Couwenberghe and others 2010). Canopy disturbance return times may occur within the life span of a tree cohort in northern hardwood forests (Frelich and Lorimer 1991) as well as in the Missouri Ozark forests (Rebertus and Meier 2001). Under such disturbance regimes, variability in canopy species susceptibility to windthrow, as well as storm season, can determine their relative abundance in stand composition (Arevalo and others 2000, Fumiko and others 2006, Peterson 2000).

In this paper, we develop a fire chronology for disjunct oak-shortleaf pine vegetation in southwest Illinois. We correlate oak and pine recruitment with fire frequency and compare long-term vegetation change between 1954-2008 with short term change following windstorms in 2008 and in 2009. Our goal was to understand long-term vegetation change in relation to exogenous disturbance processes and to provide management recommendations to help sustain this vegetation. We asked the following research questions:

1) What are the chronology and characteristics of fire, and have they changed during presettlement, prelogging (1825-1975), logging (1876-1935), and postlogging eras?
2) How have pine and oak recruitment interacted with fire history over time?
3) How have canopy and sub-canopy vegetation structure and composition changed since 1954?
4) How did windstorms alter canopy structure and affect species persistence?

Because of its fire-dependence, we expected that pine dominance and reproduction would decline with reduced fire frequency, but that this effect would be moderated by xeric habitat conditions.

STUDY SPECIES AND STUDY AREA

Shortleaf pine is characteristic of the fire-adapted vegetation of the Missouri Ozark Plateau, which extends into Illinois along 5.5 km of southwest facing limestone bluffs of the Mississippi River at the LaRue-Pine Hills Forest Service Research Natural Area, Union County, IL (Fig. 1). We hereafter refer to the study area as Pine Hills. Shortleaf pine stands occupy cherty soils on convex slopes dissected by ravines; this xeric habitat averages 57 percent slope, with 21 percent canopy openness, 79 percent ground cover by litter, and 21 percent exposed mineral soil or moss (Jones and Bowles 2010). Shortleaf pines were recorded at Pine Hills by the U.S. Public Land Survey in 1810 and were sampled by Shake (1956), Suchecki (1987), and Jones (2003). In a dendrochronological analysis of shortleaf pine tree cores from Pine Hills, Jones (2003) indicated that most of the older trees established following major disturbance in 1811-12 and attributed episodic releases in the 1900s to tree harvest, natural canopy disturbance, and favorable precipitation. Both Suchecki (1987) and Jones (2003) projected the shortleaf pine population to decline with continued fire-protection. Concern for shortleaf pine population viability prompted this investigation (Jones and Bowles 2010).

Figure 1.—Distribution of shortleaf pine in southeastern North America, indicated by shading. Source: (http://esp.cr.usgs.gov/data/atlas/little/). Arrow indicates location (circled) of disjunct stands at Pine Hills.

Variation in fire frequency is critical for maintenance of shortleaf pine. In Missouri, pine seedlings establish with 1-4 year fire return intervals but require 8-15 year fire-free intervals for survival (Stambaugh and Muzika 2007). Adult pines are fire-resistant, but record fires by developing characteristic callus tissue over injured cambium, allowing development of fire chronologies (e.g., Stambaugh and Guyette 2006). Shortleaf pine occurs with black oak, post oak, and blackjack oak at Pine Hills, as well as in dry forests and savannas in the Missouri Ozarks (Kabrick and others 2004). Post oak, a member of the white oak group, is shade intolerant (Stransky 1990) and fire tolerant (Johnson and Risser 1975). It occupies fire-maintained savanna and woodland in the Cross Timbers (Stambaugh and others 2009) and appears to be declining in southern Illinois due to succession following fire protection (McClain at al. 2010). Blackjack oak, a member of the black oak group, also tends to be shade intolerant and may occupy an earlier-successional niche than post oak (DeSantis and others 2010, Fumiko and others 2006).

European settlement began at Pine Hills during the 1820s, followed by the initiation of a logging era in the mid 1870s, and acquisition as a Forest Service natural area between 1934-1937. The 1908 Union County plat book shows five saw mills within 1 mile of Pine Hills and six property owners of the seven shortleaf pine stands sampled for this study. The 1929 county plat indicates that ownership of these stands was halved within 21 years. Annual fall fires in the late 1800s and early 1900s reportedly did little damage to large trees at Pine Hills (Krause 1985). Nevertheless, concern for damage from frequent forest fires in this area lead to increasingly greater fire protection in southern Illinois (Miller 1920), thereby reducing the extent of fire coverage. For example, although an average of one fire per year was reported from the Pine Hills area between 1936 and 1968, they occurred mostly along roads and railroads (Suchecki 1987). The Shawnee National Forest conducted prescribed burns in 1989, 1991, and 1993.

Seven shortleaf pine stands at Pine Hills sampled in 1954 by Shake (1956) established that canopy importance was shared by shortleaf pine, post oak, black oak, and blackjack oak, with *Vaccinnium* and *Rhododendron* species in the shrub layer. In 2009, we resampled transects mapped by Shake (1956). This sampling followed 2008 and 2009 windstorms that downed both pine and oak canopy trees. Most pines were uprooted, while oaks were broken off near their bases. The 2008 storm may have been 80-130 km/hr straight-line winds that were associated with the 5-6 February Super Tuesday Tornado outbreak (CIMSS 2008). The 2009 storm was a cyclonic derecho with peak 145-160 km/hr winds that crossed eastern Missouri and southern Illinois on 8 May (Brown 2009, CIMSS 2009). The 2009 storm had far greater impact on Pine Hills than did the 2008 storm (M. Jones, personal observation).

METHODS

Data Collection and Processing

To develop a fire chronology, we used a chain saw to obtain 58 cross sections from the bases of pines downed by windstorms and from previously cut pine stumps. Downed pines were unavailable in one stand. We also extracted cores to pith from 71 standing shortleaf pine and 28 oaks representing 10 cm size classes in sampling plots. All samples were sanded with progressively finer grades of sandpaper (1200 minimum grit) to reveal annual rings and fire scars. Twenty-six of the cross sections contained 113 fire scars and were used to develop a stand fire chronology. Following Stambaugh and Guyette (2006), a radius of each cross section with the least variability due to fire scarring was used for measurement, and plots of ring-width series were used for visual cross-dating of "signature years" and for locating missing or false rings. Once dating was completed, series accuracy was verified on the COFECHA program (Grissino-Mayer 2001a) and combined into a master stand chronology represented by a standardized ring-width index using the ARSTAN program (http://www.ldeo.columbia.edu/res/fac/trl/public/publicSoftware.html). FHX2 software (Grissino-Meyer 2001b) was used to graph fire scars, and to calculate fire intervals for each series, the percentage of scarred trees for each year of a fire occurrence, and Jaccard and Yule similarity indices of fire synchroneity among stands. Linear distances among stand midpoints were measured at a horizontal scale on a 7.5' USGS quadrangle map and ranged from 0.25-5 km, averaging 1.93 km in a pair-wise matrix.

Transect locations were mapped by Shake (1956) on the Wolf Lake, Illinois 7.5' USGS quadrangle. These transects were digitized onto a digital elevation model of the Wolf Lake quadrangle, from which Global Positioning System (GPS) positions were retrieved for field locations. The initial small mapping scale prevented precise relocation of transects, and our data represent random samples of stands over time. We sampled trees using the original point-center-quarter (PCQ) method (N = 210 points), as well as 0.025 ha tree plots (N = 56) in which were nested 0.001 ha sapling and 1-m^2 juvenile plots. PCQ tree sampling included the identity and diameter of the nearest tree (>10 cm d.b.h.) in each quadrant, including both standing trees and those toppled by the 2008 and 2009 windstorms. The nearest live saplings (2.54-10 cm d.b.h.) were also recorded in each quadrant. Plot sampling included trees >10 cm d.b.h. in 0.025 ha plots, saplings >1 m high and > 2.54-10 cm d.b.h. in 0.001 ha plots, and juvenile trees <1 m high in 1-m^2 plots. All windthrown trees originating within sample plots were sampled to reconstruct pre-storm (2008) stand structures. Following Shake (1956), PCQ data were used to calculate relative density and relative basal area (BA) for 2.5-25.4 and >25.4 d.b.h. size classes. Pre- and post-storm plot samples were used to calculate basal area and dominance (relative basal area) for each stand. Average 2008-2009 density was also calculated for trees, saplings, and juvenile tree species.

Statistical Procedures

What are the chronology and characteristics of fire, and have they changed during presettlement, prelogging, logging, and postlogging eras?

Superposed Epoch Analyses (SEA) was used on FHX2 software (Grissino-Meyer 2001a) to test whether fire years were associated with years of extreme drought during the presettlement, prelogging, logging, and postlogging eras. Proxy climate data were Palmer Drought Severity Indices (PDSI) from Grid 112 for southeast Missouri (Cook and others 2004); data were bootstrapped for 1000 simulations to derive 99 percent and 95 percent confidence intervals for normal drought conditions and were compared for each of the 5 years proceeding and 4 years following the year assigned to a fire scar. We used regression analysis to test whether stand ring-width indices had significant linear relationships with local 1901-2009 precipitation and temperatures in years t and t-1 for fall, winter, spring, and summer months. Data were obtained for Anna, IL from the Illinois State Climatologist Office and Illinois State Water Survey (Illinois State Water Survey 2012).

A Kruskal-Wallis one-way analysis of variance (ANOVA) was used to test whether mean fire return intervals differed among eras. For this test, fire return intervals were measured within stands. Stands were not included in a factorial test because all stands were not represented by fire intervals within each era. A separate Kruskal-Wallis test was used to compare fire return intervals among stands. A Kruskal-Wallis one-way analysis of variance was also used to test whether the mean percentage of fire scarred trees (a proxy for fire intensity) differed among eras. Regression analysis was used to test whether significant linear relationships occurred between stand distance and fire synchroneity between stands, as measured by the Jaccard and Yule indices. This test was applied within eras.

How have pine and oak recruitment interacted with fire history over time?

Regression analysis was used to test whether significant linear relationships occurred between both pine and oak recruitment numbers and decadal fire frequency calculated in a sliding 10-year window (following Stambaugh and Guyette 2006) within eras. For this test, we based recruitment on trees aged from cores taken from sample plots and lagged fire by one decade to accommodate growth to d.b.h. in xeric habitat. The sample size for oaks was limited by availability of cores to pith because heart-rot was common among black oaks.

How have canopy and sub-canopy vegetation structure and composition changed since 1954?

Shake (1956) provided relative basal area and relative density data for each stand, which required converting modern data to relative values in order to test for significant temporal changes in these categories. Repeated measures analysis of variance (RMANOVA) was used to test whether relative basal area and relative density of trees > 10 cm d.b.h. changed between 1954 and 2008 (pre-storm), using the 7 stands as replicates. These tests were applied at the species level to shortleaf pine, white oak, blackjack oak, post oak, black oak, and hickory, as well as a single group representing subordinate species that were poorly represented as canopy trees. For each stand, Shake also provided tallies of tree species stem numbers in 2.5-25.4 cm and in >25.4 cm size classes. We converted these data to stand point sample densities for each species by dividing species tallies by the number of plots in each stand. They were analyzed by RMANOVA to test whether species point density changed between 1954-2008 (pre-storm) within size classes. This provided a validation of changes in relative density. We also used linear regression to test whether 0.025 ha plot density of post oak, which declined significantly, was related to canopy openness (measured with a digital camera and hemispherical lens, Jones and Bowles 2010) and whether its basal area was inversely related to that of black oak, the dominant canopy species.

How did windstorms alter canopy structure and affect species persistence?

To assess the 2008 and 2009 storm impacts on tree species density, we used RMANOVA to test whether significant temporal change occurred in basal area and relative basal area (dominance) of dominant tree species, as well as other subordinate species treated as a single group.

RESULTS

Chronology of Fire and Tree Growth

Cross sections from 25 windthrown trees contained 110 fire scars, while a single cross-dated stump cross section contained 3 scars, resulting in 113 scars dating between 1767 and 1993 on 26 trees (Fig. 2). Seasonality of presettlement fire scars was difficult to assess because of poor conditions of the cross sections. However, the majority of scars occurred during the dormant season, indicating most fires were between September-March. The total number of years during which fires were detected included 6 years in the presettlement era (1767-1825), 22 years in the prelogging era (1826-1875), 31 in the logging era (1876-1935) years, and 22 in the postlogging era. After 1975, only one wildfire and two of three management fires were detected, and no fires were

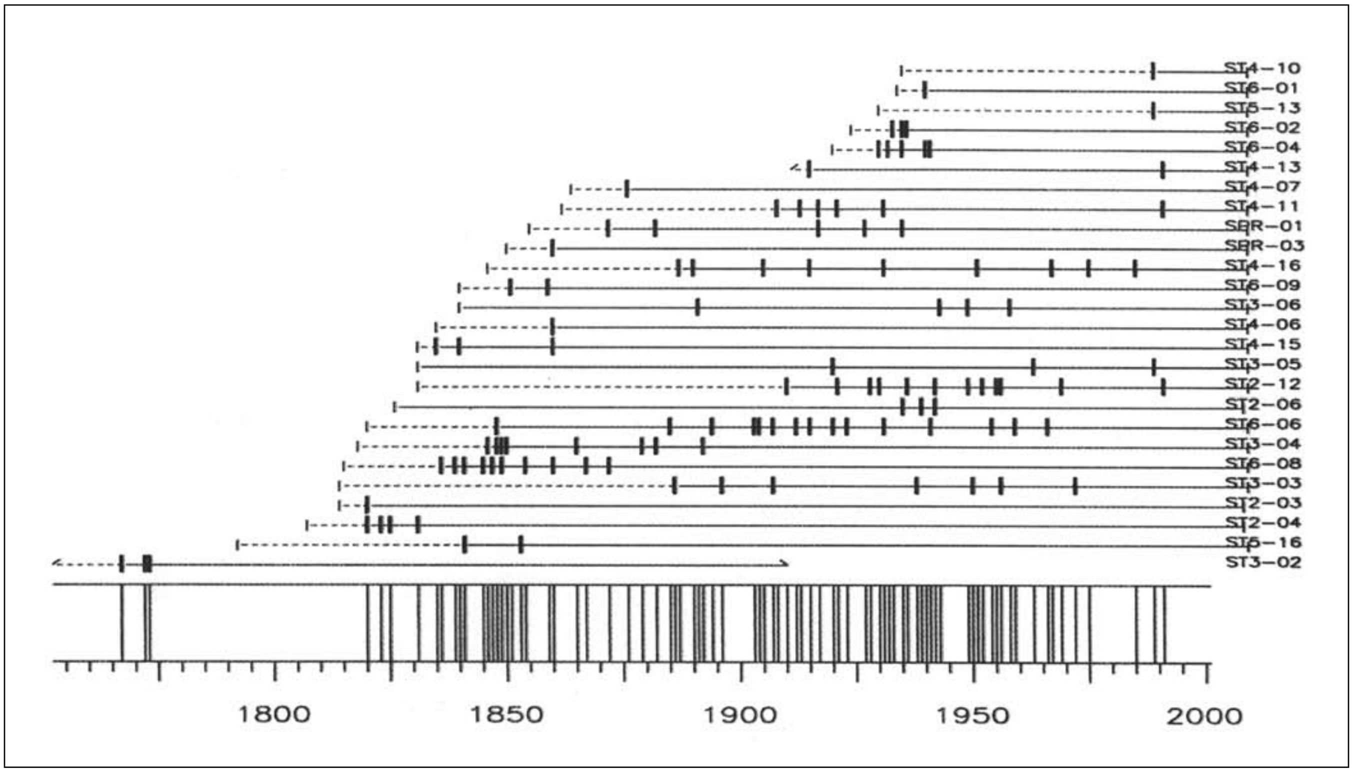

Figure 2.—Fire chronology of shortleaf pine at Pine Hills. Vertical bars represent fire scars on cross sections from 25 windthrown trees and a single cross-dated stump between 1767-1993. Vertical axis indicates stand and tree numbers.

recorded after 1993. At the landscape level, fires were recorded about every 2 years between 1820 and 1975. Fire-return intervals measured within stands increased significantly over time, averaging 4.55 (+1.1 se) years in the prelogging era, 6.57 (+1.2 se) years in the logging era and 8.19 (+1.5 se) years in the postlogging era (Fig. 3). These intervals also varied significantly (Chi-square = 13.06, $P = 0.011$) among stands, ranging from 3.7 (0.58 se) to 16.7 (8.69 se) years. The percentage of fire-scarred trees, indicating fire intensity, was greater during the prelogging era (Fig. 3). Within eras, there was no significant relationship between distance among stands and fire synchroneity. SEA indicated that fires during the presettlement era occurred during significant dry years, as indicated by the Palmer Drought Severity Index (Fig. 4). No significant relationships occurred between dry or wet years and fire during the postsettlement, logging, or postlogging eras. The standardized ring-width chronology was significantly positively correlated ($r^2 = 0.145$, $P < 0.001$) with summer precipitation and negatively correlated ($r^2 = 0.259$, $P < 0.001$) with summer temperature. There were no significant correlations with other seasonal precipitation or temperature variables, nor with any variables in the previous year.

Shortleaf pine and oak establishment based on plot-sampled tree cores were recorded from 1810-1970 (Fig. 5). The greatest percentage of pine recruitment occurred during the prelogging era when it was significantly positively correlated with decadal fire frequency (Fig. 6). Pine recruitment tended ($P < 0.07$) to be negatively correlated with fire frequency during the logging era and positively correlated during the postlogging era. Pine recruitment also underwent a moderate increase during the early decades of the postlogging era. Oak recruitment also was positively correlated with fire frequency during the prelogging era. However, a greater percentage of oak recruitment occurred during the last decade of the logging era and continued into the early postlogging era, when it was marginally ($P < 0.07$) correlated with fire frequency. Among oaks, white oak recruitment predominated during the prelogging era, black oak and blackjack oak predominated during the logging era, and white oak and black oak predominated during the postlogging eras (Fig. 5). Post oak recruitment was represented by single occurrences during the prelogging and logging eras.

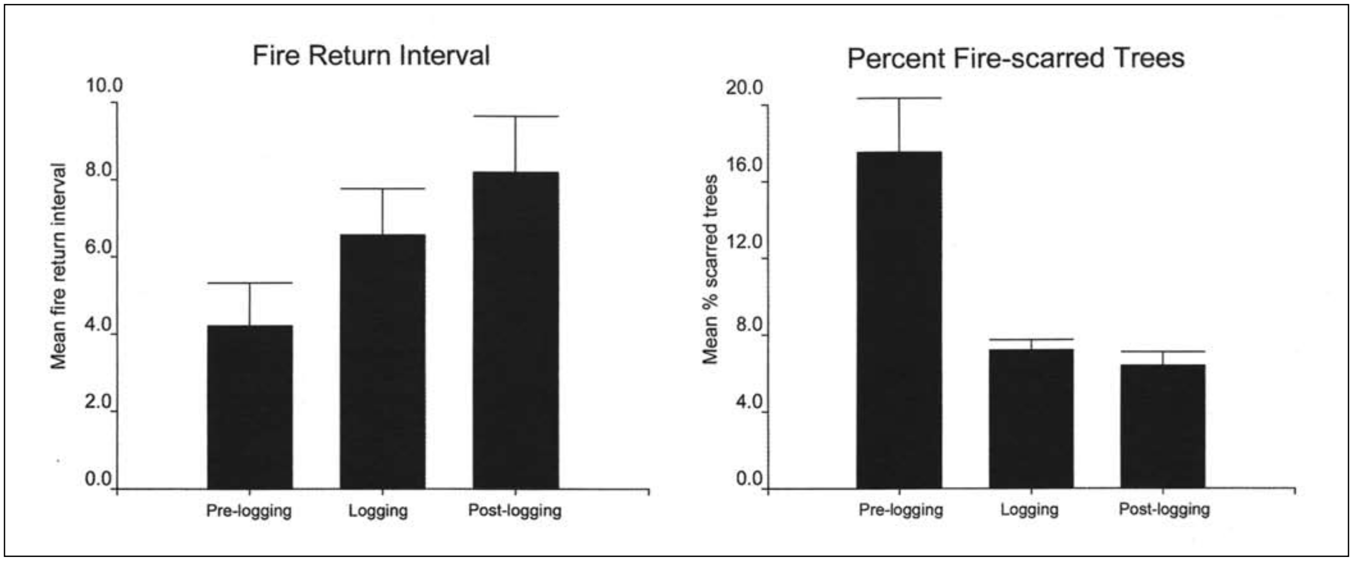

Figure 3.—Relationship between mean (+se) fire return interval and mean (+se) percentage of fire-scarred trees across eras at Pine Hills. Fire return interval: Chi-square = 6.15, P = 0.046. Sample size: Prelogging (N = 19), Logging (N = 30), Postlogging (N = 21). Percentage of fire-scarred trees: Chi-square = 31.56, P < 0.001. Sample size: Prelogging (N = 23), Logging (N = 30), Postlogging (N = 26).

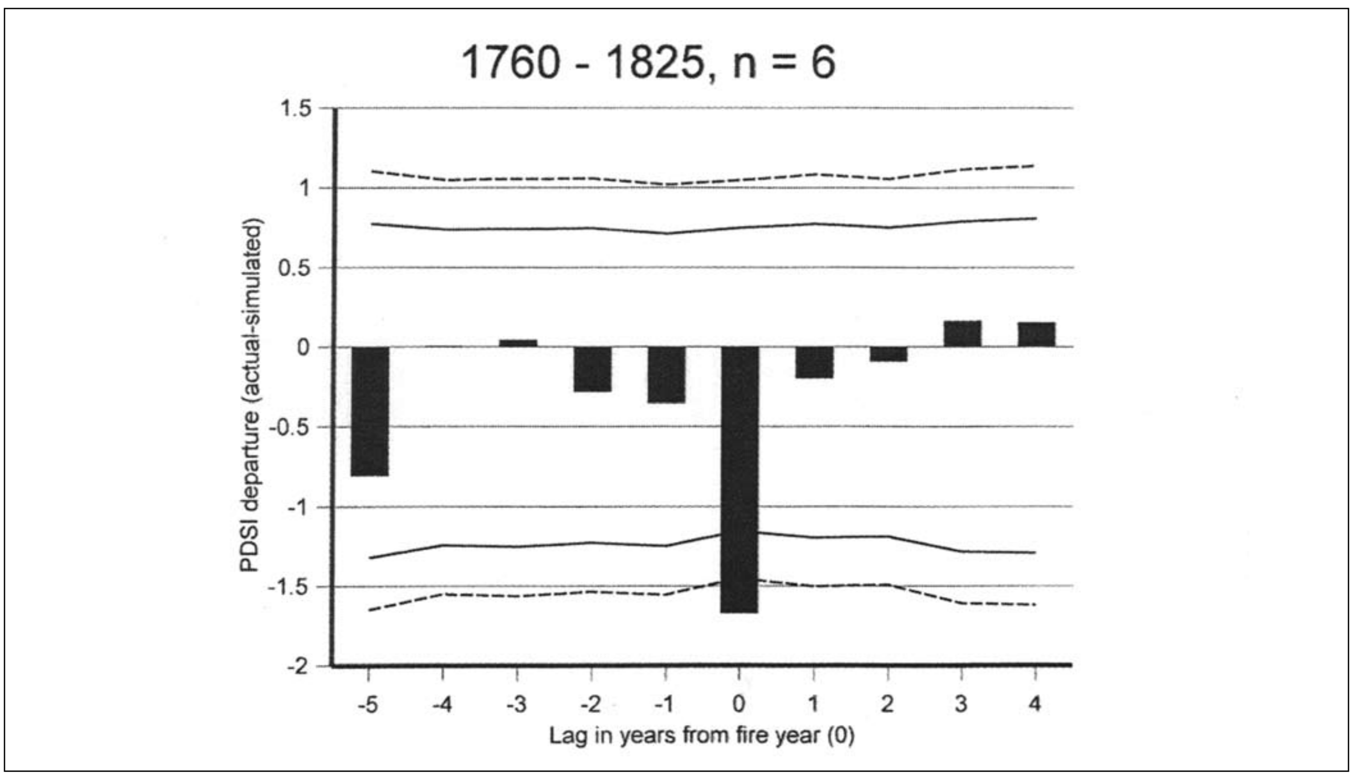

Figure 4.—Superposed epoch analysis (SEA) during the presettlement era at Pine Hills. Horizontal axis represents lag in years from fire year (0). Bars represent deviation from normal drought conditions based on 1000 simulations. Solid and dashed lines are 95 and 99 percent confidence intervals, respectively.

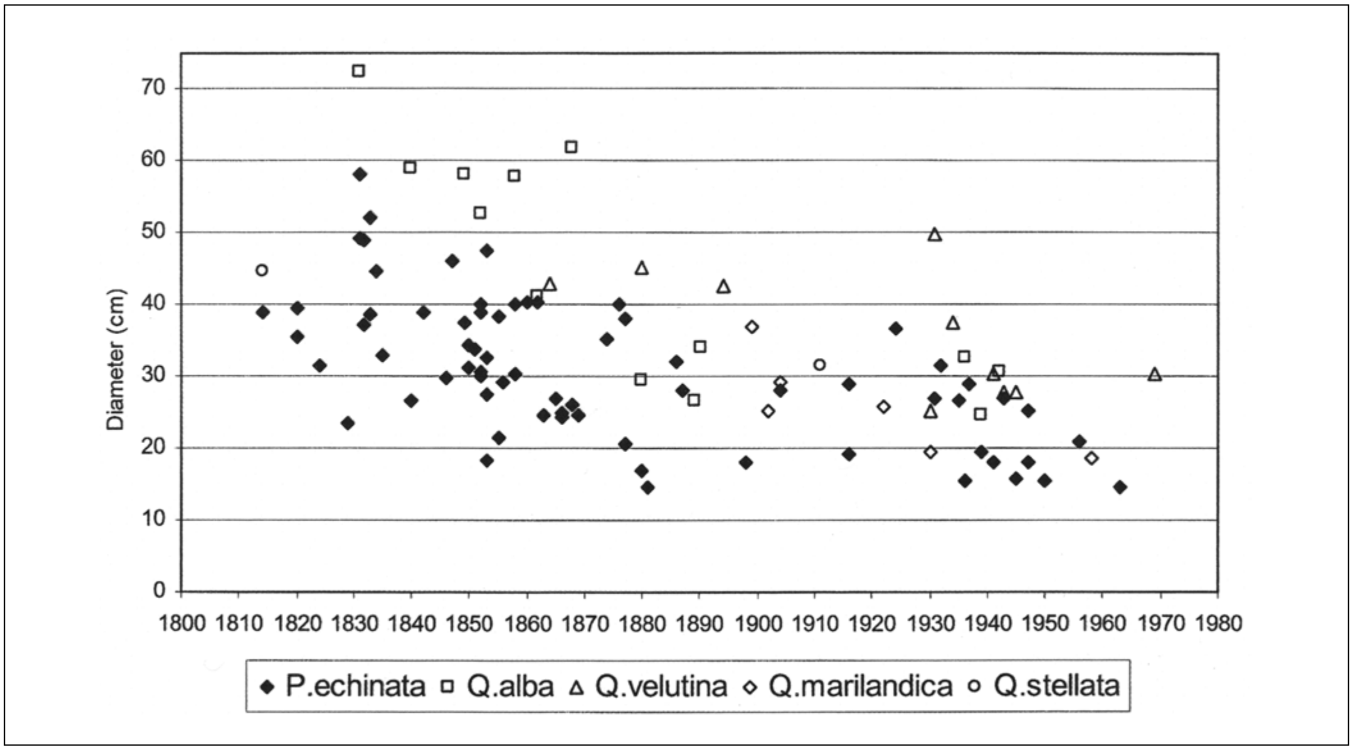

Figure 5.—Recruitment chronology and age-diameter relationships for shortleaf pine (r^2 = 0.293, P = < 0.001) and oak species (r^2 = 0.610, P < 0.001) at Pine Hills.

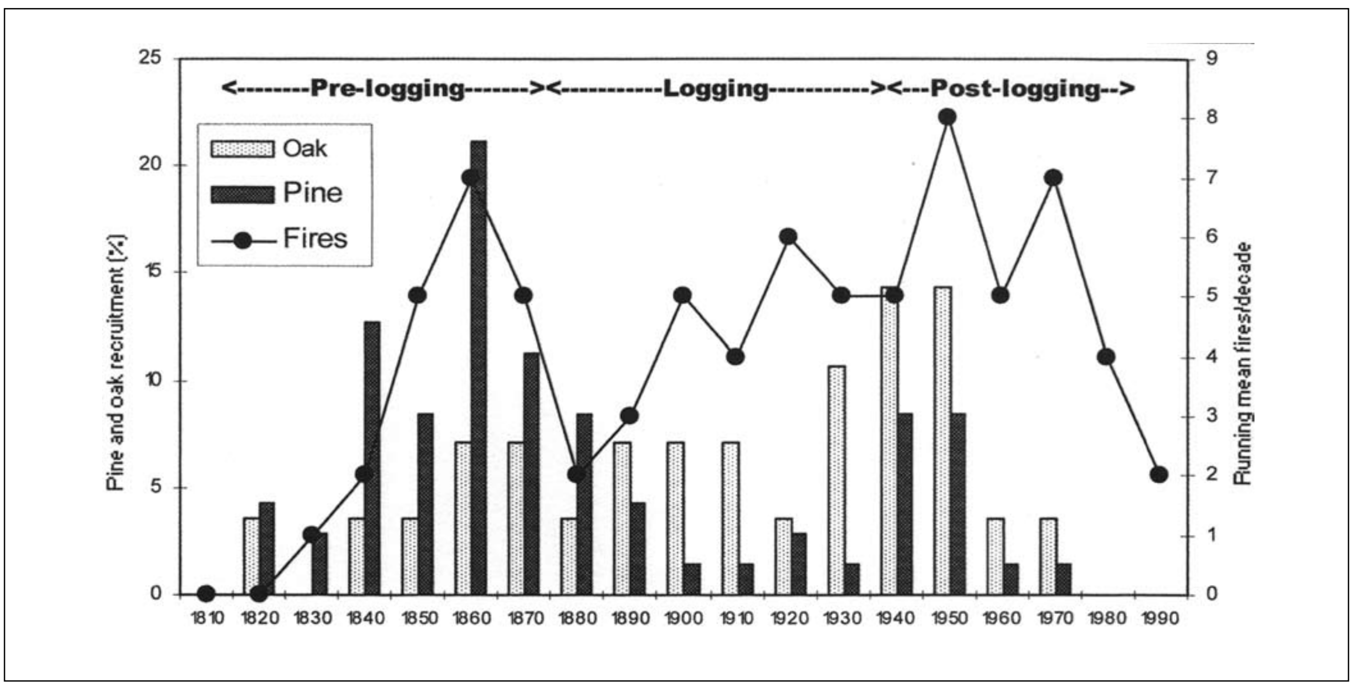

Figure 6.—Decadal relationship between running mean fires/decade and recruitment of shortleaf pine and oak at Pine Hills. Years represent end of decade. Fires are lagged 1-decade. Correlations: Prelogging (pine r = 0.869, P = 0.005; oak r = 0.832, P = 0.01); Logging (pine r = -0.83, P = 0.066; oak r = 0.211, P = 0.689); Postlogging (pine r = 0.788, P = 0.063); oak r = 0.788, P = 0.0625).

Structural and Compositional Change

By 2008, there were significant declines in relative BA and relative density of post oak and an increase in these measures for the group of subordinate species, including elm, sassafras, service berry, tulip tree, red maple, flowering dogwood, ironwood, white ash and American beach (Fig. 7, Appendix I). The most abundant of these species were subdominant with white oak, black oak, and shortleaf pine in juvenile and sapling size classes (Appendix I). Post oak also declined significantly in density in the 2.5-25.4 cm and > 25.4 cm size classes (Fig. 8). This change was accompanied by significant increases in white oak and other subordinate species in the 2.5-25.4 cm size class. In 2009, post oak density had a significant positive linear relationship with canopy openness ($r^2 = 0.5546$, $P = 0.0022$), and its basal area had a significant negative relationship with that of black oak ($r^2 = 0.2337$, $P = 0.0264$).

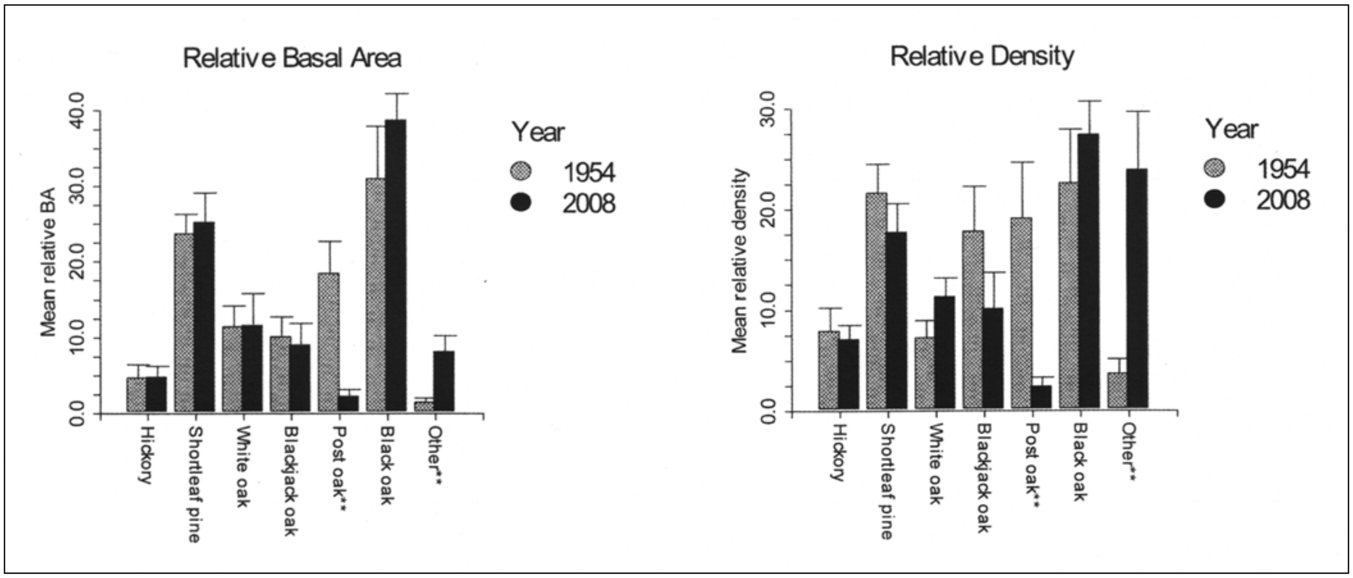

Figure 7.—Temporal change in tree species mean (+se) relative basal area (BA) and relative density at Pine Hills. Relative BA: post oak (F = 23.61, P = 0.0028), other (F = 14.78, P = 0.0086). Relative density: post oak (F = 18.26, P = 0.0052), other (P = 15.84, P= 0.0073).

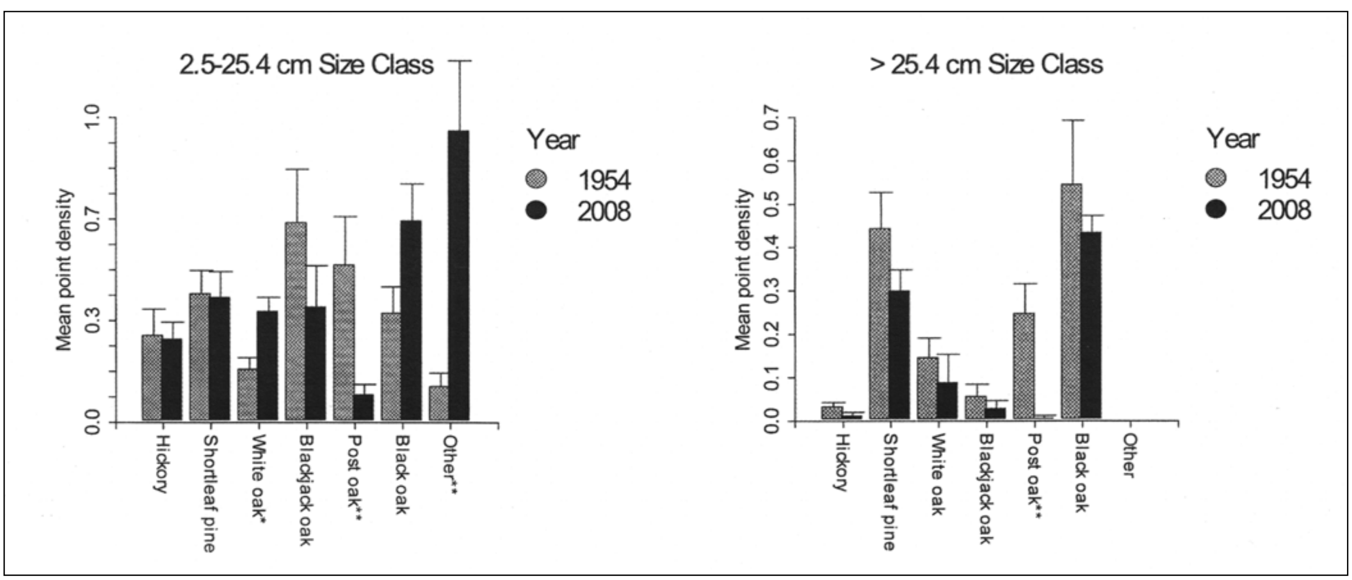

Figure 8.—Temporal change in species mean (+se) sample point density by size class at Pine Hills. 2.5-25.4 cm size class: white oak (F = 11.38, P = 0.0150), post oak (P = 11.90, P = 0.0016), other (F = 18.73, P = 0.0049); >25.4 size class: post oak (F = 29.32, P = 0.0016).

Before the windstorms, black oak had become the single dominant tree with subdominance by shortleaf pine and white oak and minor representation by blackjack oak and post oak (Fig. 9). Windstorm damage caused significant declines in BA of black oak and, to a lesser degree, shortleaf pine (Fig. 9). Because of the greater reduction of black oak BA, the storm damage resulted in increased relative BA (dominance) of hickory and shortleaf pine (Fig. 9).

DISCUSSION

Chronology and Change in the Fire Regime

At Pine Hills, average stand level fire return intervals increased from 4.55 years during the prelogging era to 8.19 years during the postlogging era. These intervals fall within the ranges reported for other Midwest oak-dominated stands with human ignition sources (McClain and others 2010) and indicate a strong human influence on the Pine Hills fire regime. The low fire synchroneity and high variability in fire return intervals among stands is probably due to spatially independent ignitions, as well as the site's dissected topography, which would reduce the landscape spread of fire. Indeed, the maximum number of stands with recorded wild fires during single years was three stands, which occurred only during 1860 and 1935. Ignitions also may have been independent of ownership by the early 1930s, when the seven stands had only three landowners. Cultural conditions may strongly affect fire return intervals. Wolf (2004) and McClain (2010) report three-decade fire return intervals during the mid to late 1800s in southeastern Wisconsin and in southern Illinois, and indicate that they correspond to periods of accelerated settlement and fire protection in the forest-prairie transition region. Although the increasing fire return intervals at Pine Hills support these findings, the absence of extended periods without fire indicates either more frequent ignitions or more moderate levels of fire suppression. The continuation of burning until about 1975 parallels results from the Ozark Wilderness of the Missouri Ozarks, where fires may have been set in protest against federal acquisition (Stambaugh and Guyette 2006). Similar actions could have taken place at Pine Hills after it was acquired as a Forest Service natural area.

Although our presettlement sample depth was limited to two trees, the lower fire frequency during this period is not inconsistent with other Midwest

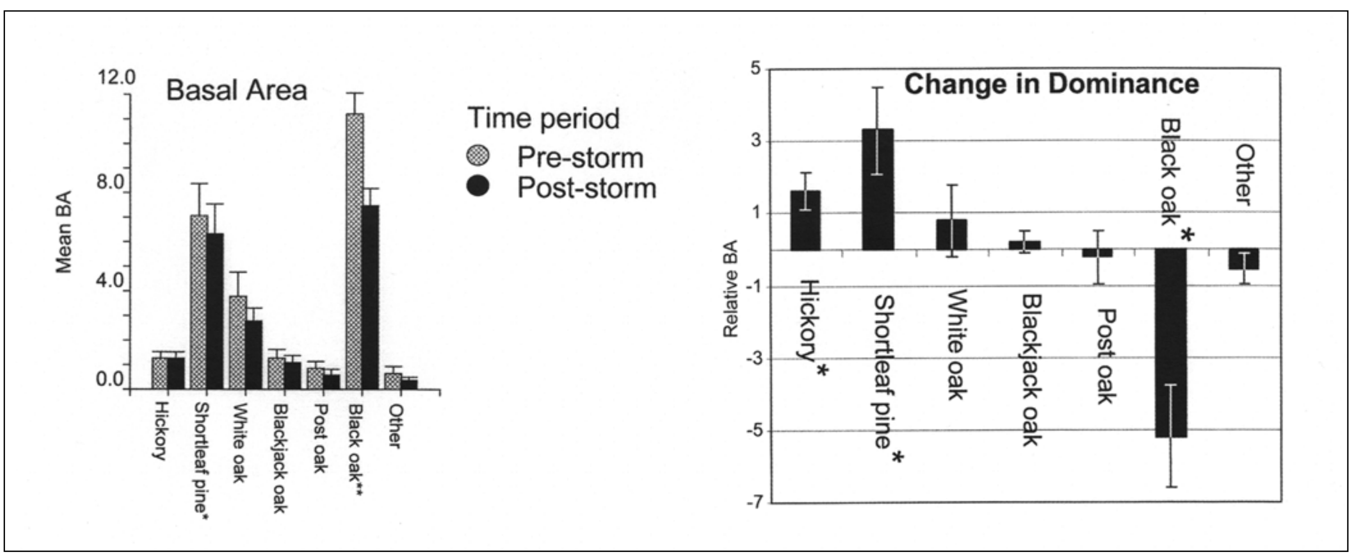

Figure 9.—Species mean (+se) pre- and post-storm basal area, and post-storm change in relative basal area (dominance) at Pine Hills. BA: shortleaf pine (F = 6.10, P = 0.0485), black oak (F = 30.51, P = 0.0015). Dominance: hickory (F = 10.34, P = 0.0183), shortleaf pine (P = 7.57, P = 0.0332), black oak (F = 13.7, P = 0.01).

studies (e.g., McClain and others 2010), which would suggest that less frequent fires characterized this period at Pine Hills. Our data also support findings of Stambaugh and Guyette (2006) that less frequent fires in the presettlement era corresponded to extreme drought conditions, while greater fire frequency during the postsettlement era masked the effects of drought on fire occurrence. Infrequent drought-driven fires may have been comparatively intense due to fuel accumulation between fire years and greater flammability during drought conditions. The occurrence of a greater percentage of scarred trees, a proxy for greater fire intensity, during the prelogging (1825-1875) era also suggests that high intensity fires would have continued through this era. The composition, structure and fuel loads of forests during this period also would have been more similar to presettlement conditions than to those that developed during the logging period. Increasing fire suppression during the 1900s may have contributed to less frequent burning as well as lower percentages of scarred trees.

Tree Recruitment and Interaction with Fire

The greater recruitment of shortleaf pine during the prelogging era, its positive correlation with decadal fire frequency, and the greater percentage of scarred trees during that era suggest that fire has had a strong positive influence on population dynamics of shortleaf pine at Pine Hills. Shortleaf pine seedling establishment requires 1-4 year fire return intervals, and 8-15 fire return intervals for subsequent survival and development (Stambaugh and Muzika 2007). At Pine Hills, temporal variation in fire return intervals falls within that range, and interactions with spatially variable topography and fuel loads may have facilitated pine regeneration and population maintenance. Nevertheless, the lower percentage of scarred trees during the logging and postlogging eras may indicate that reduced pine recruitment during these eras resulted from lower fire intensity. Slash from logging is often attributed to causing greater fuel loads and greater fire intensity, but no data are available on the intensity of logging in pine stands and the levels of fuels that might have accumulated. Although oak recruitment also had a positive relationship with fire frequency during the prelogging era, it had greater recruitment during the late logging era, as well as the early postlogging era. This process could have been enhanced by sprouts from cut stumps following logging, as well as large canopy gaps that would have persisted for a decade or more following logging. The lower recruitment of both oak and pine in the latter part of the postlogging era appears to correspond to decreasing fire frequency during that era.

Compositional Change and Compensatory Effects of Windthrow

Xeric habitats have been suggested to be resistant to successional replacement of oaks (Abrams 1992). However, long-term data have demonstrated successional change in western post oak-blackjack oak forests (DeSantis and others 2010). Likewise, our data suggest that reduced fire frequency and intensity has lead to replacement of post oak and shortleaf pine by black oak, with increased advance regeneration of shade-tolerant species. Canopy closure may be a strong causal factor in decline of post oak because of its shade intolerance (Stransky 1990), and has been suggested as a cause of post oak decline in southern Illinois (McClain at al. 2010). At Pine Hills, the positive relationship between post oak density and canopy openness and the negative relationship between post oak and black oak basal area suggest that post oak may be declining due to competition for canopy light caused by increasing canopy closure from black oak. The greater resilience of blackjack oak, white oak, and black oak may result from species specific adaptations to different habitat conditions. Blackjack oak may be adapted to more xeric or open early-successional habitat, which would allow it to persist in a comparatively narrow habitat range at Pine Hills where other species are less competitive. Black oak and white oak tend to have greater shade tolerance than blackjack oak (Dey 2003), and appear to tolerate a greater range of overlapping soil moisture conditions. This would promote their survivorship across a wider habitat range under more dynamic canopy conditions.

If fire protection continues, the increases in smaller size classes of shade-tolerant species may provide the basis for eventual successional replacement by many of these species, as they comprise well over 50 percent of the advance regeneration in oak-pine stands at Pine Hills. Similar successional change in other systems has been linked with fire protection. In Oklahoma, sapling densities of winged elm increased while densities of post oak and blackjack oak decreased (DeSantis and others 2010). Red maple is shade tolerant and fire sensitive and is currently expanding in eastern forests (Abrams 1998, Green and others 2010, McDonald and others 2002). Flowering dogwood and sassafras also colonize old fields in southwest Illinois but may not persist with succession (Ashby and Weaver 1970). Work is needed to understand whether restoring fire processes can prevent additional seedling establishment and canopy replacement by shade-tolerant species and also prevent competition by these species from excluding both pine and oaks seedlings.

Interactions between fire and windstorms have probably been important historic factors maintaining shortleaf pine at Pine Hills (Jones 2003). Our data suggest that in the recent absence of fire, the 2008 and 2009 windstorm effects have been compensatory, as they restored the former codominance of shortleaf pine with black oak (Shake 1956). Seasonality of windthrow was also critical, as the 2009 growing season storm selectively removed greater oak than pine basal area. This apparently resulted because broadleaf oak crowns are more vulnerable to windthrow than are needle-leaf pine crowns. Such stochastic exogenous disturbance processes may be important factors operating in combination with longevity of shortleaf pine and presence of optimum habitat in maintaining this disjunct pine population. The steep slopes and exposed soil at this site probably facilitated seedling germination during the presettlement fire regime and may have reduced local fuel loads and allowed seedling establishment during more frequent postsettlement fires. Although projection of increased storm damage is problematic (Peterson 2000), climate change models predict greater frequencies of severe storms (National Assessments Synthesis Team 2004) which could continue to favor shortleaf pine over hardwoods. However, as the windstorms had no direct effect on sapling or juvenile tree species, successional processes involving advance regeneration may determine future canopy composition (Peterson 2000). The size class distribution of shortleaf pine indicates that pine advance regeneration is present, and more work is needed to determine whether pine will access canopy openings created by windthrow.

ACKNOWLEDGMENTS

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APPENDIX

Species nomenclature and stems/ha for tree species in juvenile (<1 m high), sapling (>1 m, 2.54-10 cm d.b.h), and tree (>10 cm d.b.h.) size classes at Larue-Pine Hills Natural Area, ranked by juvenile density. Densities represent averages of pre- and post- 2008-2009 storm data.

Scientific name	Common name	Juvenile	Sapling	Tree
Ulmus sp. L.	Elm	5826.19	553.23	12.36
Quercus alba L.	White oak	2127.72	220.31	62.81
Sassafras albidum (Nutt.) Nees.	Sassafras	2117.30	470.96	7.32
Quercus velutina Lam.	Black oak	2036.37	307.38	141.20
Amelanchier arborea (Michx. f.) Fern.	Service berry	1714.12	159.58	2.90
Liriodendron tulipifera L.	Tulip tree	1654.14	0.00	0.41
Acer rubrum L.	Red maple	1531.75	95.92	2.94
Pinus echinata Mill.	Shortleaf pine	1475.31	228.56	120.31
Quercus marilandica Muenchh.	Blackjack oak	1164.06	119.93	50.94
Cornus florida L.	Flowering dogwood	931.96	354.52	8.17
Ulmus rubra Muhl.	Red elm	864.66	14.29	0.00
Carya glabra (Mill.) Sweet.	Pignut hickory	747.71	148.02	21.17
Diospyros virginiana L.	Persimmon	294.97	24.85	7.33
Carya ovata (Mill.) K. Koch.	Shagbark hickory	168.94	24.87	0.00
Ostrya virginiana (Mill.) K. Koch.	Ironwood	144.23	43.65	0.00
Fraxinus Americana L.	White ash	134.71	20.41	0.00
Fagus grandifolia Ehrh.	American beech	112.44	37.70	0.00
Prunus serotina Ehrh.	Black cherry	75.19	0.00	0.00
Quercus stellata Wangenh.	Post oak	64.94	14.29	19.43
Carya sp Nutt.	Hickory	59.52	0.00	0.00
Juniperus virginiana L.	Red cedar	59.52	0.00	0.00
Quercus rubra L.	Red oak	8.93	65.56	7.20
Carya texana Buckl.	Black hickory	4.46	37.68	17.61
Carya ovalis (Wangenh.) Sarg.	Sweet pignut hickory	0	0	8.51
Cercis canadensis L.	Redbud	0.00	10.20	0.00

AN OVERVIEW OF PRESCRIBED FIRE IN ARKANSAS AND OKLAHOMA OVER THE LAST 40 YEARS

Roger D. Fryar

Fire Management Officer (retired), Ozark National Forest, U.S. Forest Service, 377 Sturgeon Circle, Russellville, AR 72802. To contact, call 479-880-9592 or email at rogerfryar@yahoo.com.

Abstract.—Over the last 40 years, prescribed fire programs on National Forest lands have grown from relatively insignificant to a major emphasis area of natural resource management. During this same period, social, economic, and cultural values have greatly changed. The public's environmental awareness has likewise grown. How prescribed fire programs in Arkansas and Oklahoma have fared during this time is a subject of some interest. Scientific research in fire ecology and fire history has aided managers, enabling them to better explain the need for prescribed fire programs.

INTRODUCTION

While prescribed fire objectives can vary, restoring, enhancing, or maintaining ecosystem health has been of primary importance over the last decade. Research in the use of fire by people prior to and after European settlement and scientific discussions regarding the regenerative and restorative ecological effects of fire have aided fire managers in obtaining public confidence and support for burning programs. Today, prescribed fire in Arkansas and Oklahoma represents a large part of prescribed fire operations in the southeastern United States. The objective of this paper is to explore the history of prescribed fire programs. This comparison can provide a context and perspective into how the public perception of prescribed fire has found relatively wide acceptance where other vegetation treatment practices, such as clearcutting and herbicide use, have not.

The link between timely, pertinent research providing key rationale for why wildland fire is needed and resultant positive public perception remains key to seeing prescribed fire programs continue to grow in the future.

The utilization of prescribed fire was a small part of ecological restoration and management in Arkansas and Oklahoma 40 years ago. However, the next four decades witnessed a dramatic increase in the practice. Publicity surrounding historic natural catastrophic events such as Mount St. Helens and the Yellowstone fires may have helped the public to better understand ecosystem disturbance dynamics, resiliency, and the role of fire in forests. These natural catastrophic events provided a contrast to the many man-made disasters that occurred during the same time frame (e.g., Three-mile Island, Times Beach, Bhophal, Chernobyl, and Exxon Valdez). The man-made disasters were perceived by the public as "unnatural" and the result of mistakes in the imprudent use of science and technology. Natural catastrophic events were often perceived differently and lent themselves to use as "teachable moments" for scientists to better explain how the natural world works. Major fire seasons with wildfires affecting the urban interface, particularly in the West and in Florida, led scientists and politicians to address why wildfires seemed to be causing so much more damage than in previous decades. Ultimately, these assessments led to the National Fire Plan, and shortly afterward came the President's Healthy Forest Initiative (USDOI and USDA 2011). After that, a bipartisan led Congress passed the Healthy Forest Restoration Act (United States 108th Congress 2003). This Act focused on treatments to vegetation designed to restore "reference" ecological conditions and lower the threat of catastrophic wildfires in fire-adapted ecosystems. Prescribed fire was recognized as a primary vegetation management tool to accomplish these goals. The Fire Regime Condition Class (FRAMES 2011) and LANDFIRE (USDA and USDOI 2011) projects soon became available to help land managers identify and rank how degraded ecosystems had become, comparing reference conditions to current conditions.

HISTORY OF PRESCRIBED FIRE IN THE REGION

The Ouachita and Ozark-St. Francis National Forests in Arkansas and Oklahoma are within the Interior Highlands and are the focus for a significant amount of prescribed fire for ecological restoration, maintenance, and enhancement. Public land within this geographic area contains a high percentage of fire dependent plant communities. Shortleaf pine-bluestem and oak woodlands are two representative fire-dependent plant communities common in this area. Natural fire regimes for these communities are generally characterized by frequent, periodic fire of low to moderate severity. Current conditions are far removed from the reference condition with most lands in condition class three (the most highly departed). Federal agencies, (e.g., U.S. Department of Agriculture, Forest Service; U.S. Department of Interior, National Park Service, and Fish and Wildlife Service) state agencies, and private land trusts (e.g., The Nature Conservancy [TNC]) have developed large prescribed fire and fuel management programs. TNC provides key leadership and a clear voice to the public on prescribed fire issues. Research is conducted by universities, the Forest Service, the Fish and Wildlife Service, and National Park Service. Technology and information transfer is accomplished through a variety of means including Joint Fire Science Projects, The Fire Learning Network, research publications, Fire Effects Information System, Smoke Management Portals (e.g., Forest Consortium for the Advanced Modeling of Meteorology and Smoke [FCAMMS]), and various ad hoc councils, committees and planning efforts. A summary of some of this work is documented in Spetich (2004). These findings provided rationale for decisions in Land Management Plans for the National Forests and help guide development of reference condition descriptions in both Fire Regime Condition Class (FRCC) and LANDFIRE efforts.

There are a number of private citizen groups with a variety of views regarding prescribed fire. Some groups such as Audubon, Quail Unlimited, and The National Wild Turkey Federation serve as strong advocates for burning. Others such as Sierra Club and Wilderness Society vary in views, with some local chapters advocating the prudent use of fire while others are resistant to burning.

During the 1970s, most burning objectives for the National Forests in the southeast were tied directly to range or game habitat improvement, with programs averaging a few hundred acres to a few thousand acres annually. Management of habitat for the federally endangered red-cockaded woodpecker (RCW) led to an increase in prescribed fire and an affirmation by scientists (and the courts) that RCW was a fire dependent species whose optimal habitat required frequent burning. While normally dynamic enough to sustain themselves in the face of natural biotic and abiotic events, forest insect and disease outbreaks in Arkansas and Missouri have lent credence to the idea that disturbance driven forest ecosystems could not be sustained as host tree density rapidly increased. The resulting epidemics may have created ecologically unsustainable conditions along with both biological and economic loss. To much of the public, prudent but active management (rather than a "hands-off" approach) may have become more popular.

Few prescribed fires were occurring on federal lands in the 1970s and into the early 1980s. The hiring of wildlife biologists to work at the district level on the National Forests in the late 1970s resulted in a significant growth in prescribed fire programs. Some of the first landscape-scale burns were conducted in the late 1970s.

On private land, timber companies managed land primarily for timber production in the 1970s but began leasing lands for hunting in the late 1970s (Arkansas). Some of these leases were burned specifically to improve habitat for deer and upland birds. The burning had positive effects in maintaining fire-adapted plant and animal species. Prescribed fire by industry began diminishing as liability concerns (escapes and smoke) caused companies to rethink vegetation management alternatives. Consequently, herbicide use increased on many industrial forested lands.

In the 1980s, burning for RCW began in earnest. "New Perspectives" initiatives merging research efforts at landscape-scale projects took root in the late 1980s and 1990. Arkansas occupies a unique and important place in the history of new perspectives and ecosystem management. A historic visit to the Ouachita National Forest by Senator David Pryor (D-Arkansas) in August 1990, thereafter called the walk in the woods, served as an opportunity to shift the Ouachita's style of management in a manner that has served as a model for other national forests in the Nation (Guldin 2004). Eventually "new perspectives" was replaced with ecosystem management as the byword for how U.S. Forest Service lands were to be managed.

The 1990s saw significant growth in prescribed fire programs on all Federal lands as the National Fire Plan emerged and additional funding was made available. The size of programs along with increases in prescribed fire incidents led to more agency oversight and policies, and state regulation of prescribed fire. In 1999 the Ozark-Ouachita Highlands Assessment was done as a prelude to forest planning efforts. Findings regarding the role of fire in that assessment mirrored historical references by early explorers that described vegetation. They concluded that both pine and oak woodlands benefited from fire (USDA FS 1999).

The year 2000 marked the third year of drought in Arkansas thought to have contributed to a historic outbreak of red oak borer that affected thousands of acres in the national forests of Arkansas, Oklahoma, and Missouri. The prevailing low-disturbance fire suppression regime was cited as likely leading to oak forests being replaced by shade tolerant hardwood trees (Starkey 2004).

The Land and Resource Management Plans for both the Ouachita and Ozark-St. Francis National Forests were approved in 2005 (USDA FS 2005a, 2005b). Both plans called for an increase in prescribed fire as compared to former plans, with burning recommended for managing, restoring, and sustaining old-growth shortleaf pine, enhancing federally endangered RCW habitat and Indiana bat habitat, and responding to other ecological and social issues.

The surprising growth in prescribed fire (Table 1) can be largely attributed to how the practice has been perceived as a land management tool. Timely and pertinent research findings coupled with the relative absence of significant mishaps (escaped burns or smoke-related incidents) have also helped. Key research documenting mean fire return intervals generated through tree ring chronologies and General Land Office descriptions of pre-European settlement vegetation have greatly aided managers.

Symposia such as this and others provided a forum for the presentation of such research findings (Dickinson 2005, Powers 2007, Spetich 2004). Reference condition descriptions and modeling generated by Fire Regime Condition Class (FRAMES 2011) and LANDFIRE (USDA and USDOI 2011) programs further provided a scientific basis for rationale supporting the role of fire in ecological communities found throughout the Ouachita, Boston, and Ozark Mountains.

Table 1.—Prescribed Burning on the Ouachita and Ozark-St. Francis National Forests 1986-2010

Year	Acres burned	Year	Acres burned
1986	27,754	1999	135,041
1987	33,278	2000	132,859
1988	49,785	2001	79,653
1989	28,885	2002	120,854
1990	30,561	2003	180,644
1991	33,202	2004	202,490
1992	31,726	2005	134,957
1993	44,928	2006	116,118
1994	37,643	2007	215,483
1995	43,732	2008	187,895
1996	59,139	2009	183,163
1997	107,552	2010	197,259
1998	155,181		

The ability of fire managers to clearly articulate to the public the need to burn and to demonstrate the effectiveness of burning have greatly assisted in program acceptance by the public and subsequent growth. The Forest Service Land and Resource Management Plans of 2005 used a "best-science" approach to describe the role of fire in the ecosystem and need to do prescribed fire in an effort to accomplish specific ecological restoration. Catastrophic events of national significance captured both the public and political interest in the need to better reduce the risk of catastrophic wildfire and understand the role of fire in fire-adapted ecosystems. The National Fire Plan and Healthy Forest Initiative (USDOI and USDA 2011) and Healthy Forest Restoration Act (United States 108th Congress 2003) have further provided impetus to the need to burn.

THE FUTURE OF PRESCRIBED BURNING IN THE REGION

The future for increased burning in the Ozark/Ouachita Highlands may well depend on the ability of managers to conduct burns without incident (nuisance smoke, escapes and/or negative press from other adverse impacts). Political oversight and the public will continue to need added and ongoing evidence that burning programs are lowering the risk of catastrophic fire, resulting in the restoration of ecosystems and their fire dependent species. Such evidence could be depicted in updates in LANDFIRE mapping (LANDFIRE 2011) or other assessments. Partnerships among federal, state, and nongovernmental organizations (NGOs) will be critical for the continued and perhaps increased use of fire for ecological restoration. Today, annual burning represents a relatively small percentage of federal ownership (less than 10 percent) but is an important program area for federal agencies and is a major source of funding. Public acceptance and/or support for prescribed fire are important to any program's growth. Strong partnerships between research, state and federal agencies, and conservation groups will undoubtedly help burners to accomplish program goals.

The future growth in burning programs is likely to be less in this decade than in the past two decades. More stringent prescribed fire parameters, smoke issues, liability risks from escapes, and potential regulation of emissions could cloud the future for any significant program growth. Nevertheless, more pertinent research showing the continuing need for prescribed fire along with focused technology transfer and key partnerships could help alleviate many of the potential roadblocks.

There is a compelling logic to nature. Form really does follow function. Technology transfer from scientist to resource manager is partially dependent on the individual researcher's ability to publish. There are opportunities to leverage discoveries and findings among researchers delving into similar projects. There must be ways for managers to provide feedback to researchers to ensure there is a focus to what is being studied, i.e., that some of the questions scientists are studying have direct applicability to help program managers accomplish resource goals and objectives. There are opportunities to use natural events as a means to examine, explain, or demonstrate ecological processes to the public. There are new and/or emerging technologies that can greatly enhance technology transfer. The public has shown an amazing interest for several decades to better understand how things work in an ecological sense. Better public knowledge of ecosystem function promotes the potential "buy-in" to projects and programs that otherwise might be so controversial as to be impossible to implement. Consequently, a key message and challenge to researchers is to help managers better describe to the public why burning is important and how prescribed fire can be used to restore, enhance, or maintain fire-adapted ecosystems. Engaging the public by seeking input on both programmatic and site-specific projects can have a very positive effect. Using forums like the Fire Learning Network, Prescribed Fire Councils, and symposia like these can foster partnerships and meld diverse groups to a common goal.

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The content of this paper reflects the views of the author(s), who are responsible for the facts and accuracy of the information presented herein.

FROM THE FIELD: FIRE MANAGEMENT IN OAK FOREST ECOSYSTEMS

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RESTORING FIRE SUPPRESSED TEXAS OAK WOODLANDS TO HISTORIC CONDITIONS USING PRESCRIBED FIRE

Jeff C. Sparks, Michael C. Stambaugh, and Eric L. Keith

Wildland Fire Program Manager (JCS), Texas Parks and Wildlife Dept., 12016 FM 848, Tyler, TX 75707; Research Assistant Professor (MCS), University of Missouri-Columbia; Project Manager (ELK), Raven Environmental. JCS is corresponding author: to contact call 903-566-0535 ext. 237 or email at Jeff.Sparks@tpwd.state.tx.us.

Abstract.—Comparable to many oak ecosystems across the eastern United States, oak woodlands in Texas display characteristics of changing composition and structure due to altered fire regimes. Information describing historic fire regimes suggests woodlands underwent relatively frequent and repeated burning prior to major Euro-American influence in the early 19th century. Oak woodland management is a central goal of the Texas Parks and Wildlife Department natural community management; however many questions and challenges exist related to habitat loss and fragmentation, human populations, and prescribed fire implementation. In this paper we: 1) review information describing the historic fire regimes and community structures of Texas oak woodlands; and 2) detail fire prescriptions and challenges related to restoring long unburned sites.

INTRODUCTION

A goal of many land management agencies is to restore existing vegetation communities to resemble the vegetation composition and structure prior to industrialization or Euro-American settlement. The timing of major landscape and fire regime changes varied spatially but generally occurred in Texas during the mid-19th century or earlier. Restoring oak woodlands that have undergone decades of fire exclusion and become invaded by a dense understory and midstory can require intensive management and high cost inputs.

Using prescribed fire with standard published prescription parameters in these invaded oak woodlands often results in intermediate fire spread and relatively low intensity fires, thus not meeting objectives of reducing the stature of the invading understory and increasing the coverage of herbaceous species. It is often recommended that a combination of higher cost management tools such as mechanical and chemical treatments be used initially followed by prescribed fire as a maintenance tool. These more intensive management efforts greatly increase the cost for restoring these invaded woodlands, thus limiting the area that can potentially be restored. Based on 13 years of experience, we have begun to develop prescriptions for effectively restoring these vegetation communities to historic conditions by using fire as the primary management tool.

CONDITIONS OF HISTORIC TEXAS OAK WOODS AND PRAIRIES

Composition and Structure

Aside from anthropogenic features (roads, towns, agricultural lands), the major difference between the pre-Euro-American settlement vegetation and the current vegetation in Texas Oak Woods and Prairies (Fig. 1) is the increase in woody species biomass and resulting reduction in herbaceous species biomass (Diggs and others 2006). In the absence of fire (natural

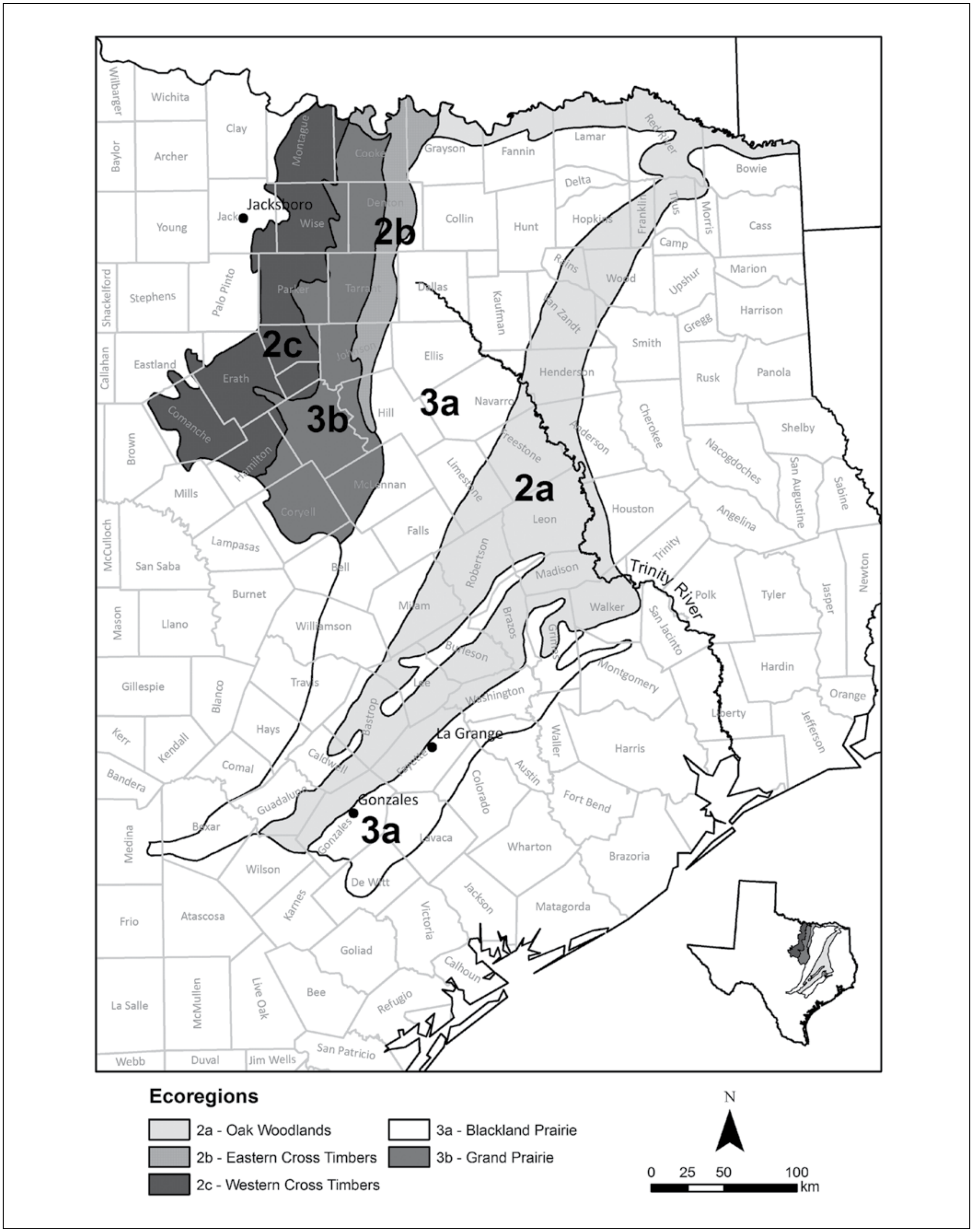

Figure 1.—Map of Texas ecoregions within the Oak Woods and Prairies region. Place names relate to the descriptions given for historic descriptions of vegetation and fire history.

and controlled), woody species have encroached into post oak (*Quercus stellata*) dominated communities that were historically open and park-like (Diggs and others 2006). The terms savanna and woodland do not accurately portray the region's current forest community structures, but according to early settler and explorer reports, they were accurate descriptors of pre-Euro-American settlement conditions (Diggs and others 2006).

Olmsted (1857) described an area most likely near the Houston County–Leon County line:

> "We came today upon the first prairie of any extent, and shortly after crossed the Trinity River. After having been shut in during so many days by dreary winter forests, we were quite exhilarated at coming out upon an open country and a distant view. During the whole day's ride the soil improved, and the country grew more attractive. Small prairies alternated agreeably with post-oak woods. The post-oak…forms a very prominent feature in Texas scenery and impressions. It is a somewhat small broad-leaved oak of symmetrical shape, and appears wherever the soil is light and sandy, in a very regular open forest growth. It stands in islands in the large prairies or frequently borders an open prairie through a large tract."

More densely forested areas would have occurred in association with natural fire breaks or in areas with lessened fire severities such as in rough terrain or low productivity sites. Roemer (1849) provides evidence for this vegetation patterning in his descriptions of an area between Gonzales and La Grange as "a sandy, hilly country, covered almost entirely with post oak forests."

Captain Domingo Ramón in 1716 described forests in what is probably present-day Burleson County as so dense that "there were not enough hatchets and knives to open a passage" (Foster 1995).

Olmstead (1857) described multiple community types and structures including the presence of fires in this description of Leon County:

> "During the first part of the day we went over small, level, wet prairies, irregularly skirted by heavy timber, with occasional isolated clumps and scattered bushes. Most of the prairies have been burned over. Both yesterday and to-day we have been surrounded by the glare of fires at night. After a few miles began post-oak, which changed to blackjack, and for the remainder of the day the country was as forbidding as a moor."

Approximately 100 km to the south (Colorado County), Roemer (1849) described the structure of post oak forests:

> "These forests …have a remarkable resemblance in winter to the cultivated German oak forests…. In other forests of North America many varieties of trees are usually found, but in the post-oak forests all are excluded with the exception of a few walnuts. Underbrush is also lacking. The soil upon which the post-oaks grow is usually of average fertility, but also often sterile and unproductive…. [There] is a wide zone where deposits of gravel and sand are found, and where farming cannot be carried on successfully. Here the land is covered with post-oaks."

In summary, the Texas Oak Woods and Prairies region was likely a complex mosaic varying from prairies and open savannas to forests with closed canopies (Diggs and others 2006; Keith 2008, 2009d, 2010a, 2010b; Keith and Leavitt 2008).

Fire Regimes

Since Euro-American settlement, fire regimes in Texas and throughout North America have been highly altered. Although fire is regarded as a principal historic disturbance to most of Texas' ecoregions (McNab and Avers 1994), relatively little work has been done to describe how fire regimes of Texas Oak Woods and Prairies varied spatially and temporally (Courtwright 2007, Diamond and others 1995). Here we describe fire regimes relevant to Texas Oak Woods and Prairies (Fig. 1).

Oak Woodlands

Very little information is available describing the historic fire regime in the Oak Woodlands (Fig. 1). Many of the relict oak woodlands have experienced infill of woody vegetation suggesting that the disturbance regime (fire and/or grazing) has been altered. A recent fire scar history study developed in a relict sand post oak (*Quercus margaretta*) woodland in Van Zandt County provides one example of what the fire regime may have been (Stambaugh and others 2011). Here, the mean fire return interval for the 325 years from 1681-2005 was 6.9 years. Prior to 1850 (approx. timing of major Euro-American influence), fire intervals ranged from 2 to 16 years, with a mean fire interval of 5.9 years. Based on the presence and recruitment of trees, there was no evidence of stand replacing fire events, however two relatively severe fires occurred prior to 1850. During the Civil War era, fire frequency decreased and coincided with establishment of an oak cohort. Nearly all fires recorded at the site occurred when oaks were dormant.

Eastern and Western Cross Timbers

Fire is considered an important disturbance agent to maintaining the Cross Timbers ecosystem (Engle and others 1996) which extends from southeastern Kansas to central Texas (Francaviglia 2000). Dyksterhuis (1948) related that early settlers of the Jacksboro vicinity recalled no shrubby undergrowth in the western Cross Timbers, but instead remember a grassy understory which commonly burned during dry periods. They also stated that when the first white settlers arrived in the western Cross Timbers fringe, the Indians were known to regularly burn off the grass and had done so for years. Dyksterhuis (1948) described the uncertainty as to the pre-Euro-American settlement fire regime of the Cross Timbers, but several accounts by early travelers mention the incursion of adjacent prairie fires into the Cross Timbers as the mechanism for grassy understories and branch formations of the trees. A survey conducted in the Trinity River watershed by the U.S. Department of Agriculture in 1942 used fire scars to show that periodic fires were still occurring through the mid-20th century (Dyksterhuis 1948).

Recently, several fire scar history studies in more northerly Cross Timbers have contributed to characterizing the fire regime (Allen and Palmer 2011, Clark and others 2007, DeSantis and others 2010, Stambaugh and others 2009). Common among these studies are pre-Euro-American settlement period mean fire intervals of 3 to 6 years, a preponderance of dormant season fire events, and evidence for anthropogenic burning. These studies have had little to bear on the range of fire severity, however all sites had trees >200 years in age, suggesting no stand replacing events had occurred during at least the previous two centuries. Based on observations of fire scar heights in Cross Timbers post oaks, many of these sites likely experienced low- to moderate-severity fires. However, fires have the potential to be very high severity (e.g., stand-replacing) and widespread (>100,000 acres), particularly during drought conditions. In April 2011, wildfires burning through the western Cross Timbers in Palo Pinto County were stand replacing crown fires.

Blackland Prairie and Grand Prairies

As the southernmost extension of the “true prairie” (i.e., tallgrass prairie), the Blackland Prairie probably had a similar fire regime to other tallgrass prairie regions. The Blackland Prairie experiences higher temperatures and a longer growing season than the rest of the tallgrass prairie ecosystem, and its fire regime may have reflected these environmental differences (Smeins 1972). Smeins (2004) cites two reports from the 1840s (Gregg 1844, Kendall 1845) that assert that frequent fires and large grazing animals were the primary influences on vegetation in the Grand Prairie and Cross Timbers regions in pre-Euro-American settlement times. Considering the Blackland and Grand Prairie’s positions between the oak-dominated Cross Timbers and Oak Woodlands, it is likely that their fire regimes are strongly related. Like the Cross Timbers and Oak Woodlands, it is likely that these large prairie subregions burned at least every 3 to 4 years. With a dominance of grass surface fuels, the potential burning window of the prairies was likely larger, leading to an increased fire frequency compared to areas dominated by litter and woody fuels.

Current Structure

Post oak savanna habitats that historically burned regularly and have continued to burn to present day are very open and dominated by native grasses and forbs in the understory (Keith 2008, 2010b; Keith and Leavitt 2008). Examples of this habitat type are now extremely rare. Pre-Euro-American settlement upland habitat types can be characterized most prominently by the Post Oak (*Quercus stellata*)-Blackjack Oak (*Quercus marilandica*) Series and the Post Oak (including sand post oak [*Quercus margaretta*])-Black Hickory (*Carya texana*) Series (Diamond and others 1987). In addition, the Loblolly Pine (*Pinus taeda*)-Oak Series, Shortleaf Pine (*Pinus echinata*)-Oak Series, and Bluejack Oak (*Quercus incana*)-Pine Series can be found in uplands on the eastern edge of the Oak Woods and Prairies Ecoregion and in the Bastrop Lost Pines Subregion (Griffith and others 2007). Bluejack oak woodlands occur on xeric sandy soils in this region. In the western portion of the region, habitats in the Plateau Live Oak (*Quercus fusiformis*) Series, Plateau Live Oak-Midgrass Series, and Ashe Juniper (*Juniperus ashei*)-Oak Series also occur as part of the ecotone with the Edwards Plateau Ecoregion. The Coastal Live Oak (*Quercus virginiana*)-Post Oak Series occurs in the southern portion of the ecoregion. A complete list of common plant associations found throughout the region is found in Table 1 (Nature Serve 2011). Plant associations describe the most narrowly defined habitat types.

Table 1.—Common plant associations including series and usual fuel model for habitats in the Post Oak Savanna and Cross Timbers and Prairies Ecoregions. See Table 2 for descriptions of fuel models.

Series	Association	Fuel Model
Post Oak-Blackjack Oak Series	Eastern Redcedar Forest	TL3
Coastal Live Oak-Post Oak Series	Live Oak-Post Oak-Cedar Elm-Water Oak Forest	TL6, TU2, TU3
Loblolly Pine-Oak Series	Loblolly Pine-Post Oak-Blackjack Oak Forest (Vernia or Jedd Rocky Upland)	TL6, TL8
Loblolly Pine-Oak Series	Loblolly Pine-Post Oak-Eastern Redcedar-Blackjack Oak-Black Hickory Forest	TL8
Loblolly Pine-Oak Series	Loblolly Pine-Sand Post Oak-Post Oak-Blackjack Oak Forest	TU2
Loblolly Pine-Oak Series	Loblolly Pine-Yaupon Ravine Forest	TL9
Plateau Live Oak-Midgrass Series	Plateau Live Oak / Little Bluestem Woodland	GR4
Ashe Juniper-Oak Series	Post Oak-Ashe Juniper-Purpletop-Virginia Wild Rye Woodland	TL6, TU3
Post Oak-Black Hickory Series	Post Oak-Black Hickory-Yaupon-Woodoats-(Little Bluestem)-Woodland	TL6, TU2, TU4
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak / Little Bluestem Woodland	GR2, GR6, GS3, GS4, TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak Cross Timbers Woodland	TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak Woodland	TU3

(Table 1 continued on next page)

Table 1 (continued).—Common plant associations including series and usual fuel model for habitats in the Post Oak Savanna and Cross Timbers and Prairies Ecoregions. See Table 2 for descriptions of fuel models.

Series	Association	Fuel Model
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-American Ash-Winged Elm-Woodoats Forest	TU2, TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Black Hickory-Farkleberry Forest	TL6, TL9, TU2, TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Cedar Elm Forest	TL6
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Cedar Elm-Little Bluestem-Coralberry Woodland	TU2, TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Eastern Redcedar Forest	TU2
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Eastern Redcedar-Little Bluestem Woodland	GS3, TU2, TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Red Oak-Yaupon-Woodoats Forest	TL6
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Rusty Blackhaw-Roughleaf Dogwood-Purpletop Forest	TL6, TU2, TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Winged Elm-Roughleaf Dogwood-Silver Bluestem-Little Bluestem Woodland	GS4, TU3
Post Oak-Blackjack Oak Series	Post Oak-Blackjack Oak-Yaupon Forest	TU2
Post Oak-Blackjack Oak Series	Post Oak-Cedar Elm Forest	TL6, TU2
Post Oak-Blackjack Oak Series	Post Oak-Cedar Elm-Blackjack Oak-Little Bluestem Woodland	GS3, GS4, TU3
Post Oak-Blackjack Oak Series	Post Oak-Cedar Elm-Blackjack Oak-Purpletop-Virginia Wild Rye Woodland	TU2, TU3
Coastal Live Oak-Post Oak Series	Post Oak-Cedar Elm-Cherokee Sedge-Purple Top Forest	TU2, TU3
Post Oak-Black Hickory Series	Post Oak-Cedar Elm-Coralberry-Caric Sedge Forest	TL6
Post Oak-Blackjack Oak Series	Post Oak-Gum Bumelia-Saw Greenbriar-Purpletop Woodland	GS3, TU3
Post Oak-Blackjack Oak Series	Post Oak-Little Bluestem-Longspike Tridens-Bushy Bluestem-Rush Spp Woodland	GS3
Post Oak-Blackjack Oak Series	Post Oak-Lovegrass-Silver Bluestem Woodland	GR3
Post Oak-Blackjack Oak Series	Post Oak-Mesquite Woodland	GR4
Post Oak-Blackjack Oak Series	Post Oak-Plains Love Grass-Longspike Tridens Claypan Savannah	GR3
Post Oak-Blackjack Oak Series	Post Oak-Plateau Live Oak-Cedar Elm-Caric Sedge-Virginia Wildrye Forest	TU3

(Table 1 continued on next page)

Table 1 (continued).—Common plant associations including series and usual fuel model for habitats in the Post Oak Savanna and Cross Timbers and Prairies Ecoregions. See Table 2 for descriptions of fuel models.

Series	Association	Fuel Model
Post Oak-Black Hickory Series	Post Oak-Sand Post Oak-Black Hickory-Blackjack Oak-Southern Red Oak-Woodoats Forest	TL6
Post Oak-Black Hickory Series	Post Oak-Sand Post Oak-Black Hickory-Blackjack Oak-White Ash-Woodoats Forest	TL6, TU2
Post Oak-Black Hickory Series	Post Oak-Southern Red Oak-Black Hickory-White Ash-Shortleaf Pine Forest	TU2
Post Oak-Blackjack Oak Series	Post Oak-Southern Red Oak-Blackjack Oak-Shortleaf Pine Forest	TL6, TU3
Post Oak-Blackjack Oak Series	Post Oak-Southern Red Oak-Water Oak-Nutmeg Hickory-American Ash Forest	TU2
Post Oak-Blackjack Oak Series	Post Oak-Texas Ash-Plateau Live Oak-Ashe Juniper Slope Forest	TL6, TU2, TU3
Post Oak-Blackjack Oak Series	Post Oak-Water Oak-Cedar Elm-Sugarberry Forest	TU2, SH9
Post Oak-Blackjack Oak Series	Post Oak-Water Oak-Sand Post Oak-Winged Elm Forest	TL2
Post Oak-Blackjack Oak Series	Post Oak-White Ash-Blackjack Oak-Woodoats Forest	TL6
Post Oak-Blackjack Oak Series	Post Oak-Winged Elm Forest	TL6
Post Oak-Blackjack Oak Series	Post Oak-Winged Elm-Water Oak-Blackjack Oak Forest	TU2, TU3
Post Oak-Blackjack Oak Series	Post Oak-Winged Elm-White Ash-Beautyberry-Woodoats-Nutrush Forest	TL6, TU2
Post Oak-Black Hickory Series	Sand Post Oak-Black Hickory-Loblolly Pine Forest	TU2, TU3
Post Oak-Black Hickory Series	Sand Post Oak-Black Hickory-Red Oak-Little Bluestem Woodland	TU3
Post Oak-Black Hickory Series	Sand Post Oak-Black Hickory-Red Oak-Yaupon-Woodoats Forest	TL6
Post Oak-Black Hickory Series	Sand Post Oak-Black Hickory-Southern Red Oak Forest	TU2
Coastal Live Oak-Post Oak Series	Sand Post Oak-Live Oak-Black Hickory-Yaupon Forest	TU2
Loblolly Pine-Oak Series	Sand Post Oak-Loblolly Pine-Black Hickory-Eastern Redcedar-Yaupon Forest	TL6, TL9, TU2
Shortleaf Pine-Oak Series	Shortleaf Pine-Southern Red Oak-Cherrybark Oak-Loblolly Pine-Post Oak-White Oak-White Ash Forest	TL6, TU2, TU3
Post Oak-Black Hickory Series	Southern Red Oak-Sand Post Oak-Post Oak-Eastern Redcedar-Water Oak Forest	TL6

Most of the natural vegetation that has not been cleared for agriculture or urbanization is generally characterized by a mature overstory of post oak and a dense understory dominated by woody species. The most common woody species in these dense understories within its range is yaupon (*Ilex vomitoria*) (Keith 2009a, 2009b, 2009c, 2009d). In many areas, yaupon often creates a near monoculture in the understory, shading and virtually eliminating all native herbaceous vegetation. In the native yaupon range, post oak woodlands and forests are noticeably dense and difficult to restore to habitats with open herbaceous understories. Common community types or plant associations of this type (Nature Serve 2011) include the Loblolly Pine-Yaupon Ravine Forest, Post Oak-Black Hickory-Yaupon-Woodoats-(Little Bluestem)-Woodland, Post Oak-Blackjack Oak-Red Oak-Yaupon-Woodoats Forest, Post Oak-Blackjack Oak-Yaupon Forest, and Sand Post Oak-Black Hickory-Red Oak-Yaupon-Woodoats Forest (Table 1).

Outside the native yaupon range, woodlands are noticeably more open and easier to restore to habitats with herbaceous understories. Woody species that also encroach into upland habitats within and outside the native yaupon range include winged elm (*Ulmus alata*), American beautyberry (*Callicarpa americana*), eastern redcedar (*Juniperus virginiana*), saw greenbriar (*Smilax bona-nox*), Small's greenbriar (*Smilax smallii*), roughleaf dogwood (*Cornus drummondii*) (on clay soils), upland privet (*Forestiera ligustrina*), Alabama supplejack (*Berchemia scandens*), coralberry (*Symphoricarpos orbiculatus*), Virginia creeper (*Parthenocissus quinquefolia*), and poison ivy (*Toxicodendron radicans*). The latter four species are often as effective as yaupon in out-competing native herbaceous vegetation outside the dominant range of yaupon (Keith 2006, 2010b; Keith and Leavitt 2008). Common plant associations include Eastern Redcedar Forest, Post Oak-Blackjack Oak-Eastern Redcedar Forest, Post Oak-Winged Elm Forest, and Post Oak-Winged Elm-White Ash-Beautyberry-Woodoats-Nutrush Forest (Nature Serve 2011) (Table 1).

High quality savannas and woodlands consist of areas that have been frequently burned, and these areas are often dominated by post oak and a diverse herbaceous ground cover layer (Keith 2008). The dominant herbaceous species is most often little bluestem (*Schizachyrium scoparium*). Other common species include purpletop (*Tridens flavus*), Virginia wildrye (*Elymus virginicus*), silver bluestem (*Bothriochloa laguroides*), meadow dropseed (*Sporobolus compositus*), black needlegrass (*Piptochaetium avenaceum*), rosettegrasses (*Dichanthelium* spp.), Texas ironweed (*Vernonia texana*), hairy sunflower (*Helianthus hirsutus*), and Georgia rockrose (*Helianthemum georgianum*). Areas that receive less frequent fire have developed into habitats best described as close-canopied forests (Keith 2010c). These forests are also dominated by post oak (and sand post oak) in the overstory, vary from sparse to dense in the midstory, and generally possess a sparse herbaceous understory. The most common herbaceous vegetation in these forests is woodoats (*Chasmanthium sessiliflorum*), nutrush (*Scleria oligantha*), rosettegrasses (*Dichanthelium* spp.), caric sedges (*Carex* spp.), and black snakeroot (*Sanicula canadensis*). Common plant associations include the Post Oak-Blackjack Oak/Little Bluestem Woodland in the Post Oak Savanna Ecoregion and the Post Oak-Blackjack Oak Cross Timbers Woodland in the East and West Cross Timbers Ecoregions (Nature Serve 2011) (Table 1).

PRESCRIBED FIRE MANAGEMENT

There are few published prescriptions for effective oak woodlands and savanna burns in Texas. Available prescription parameters are more applicable to meeting objectives in grass-dominated fuels or open restored woodlands where there are sufficient herbaceous fuels to carry fire. As fire is excluded from an area, the ingrowth of understory and midstory vegetation shelters the surface fuels, preventing grasses and other herbaceous species from growing. The primary carriers of fire become leaf litter or a combination of leaf litter and shrubs versus a grass leaf-litter combination. In these cases, fuel models change from a timber-understory (TU) to a timber litter (TL) fuel type (Scott and Burgan 2005). Fuel models in Texas fire suppressed oak woodlands, forests, and savannas can most commonly be categorized as either moderate load broadleaf litter (TL6) or moderate load, humid climate timber-shrub (TU2) (Tables 1 and 2) (Scott and Burgan 2005).

First entry burns in long unburned oak forest or savanna (Fuel Model TL6) can be difficult because the surface fuels are fully sheltered and have limited flammability. Under typical moderate prescriptions (RH >30, Winds <10 mph), we have observed only marginal, patchy burning with poor results for controlling woody invasive species in the understory and midstory. For restoration efforts to be effective, it is important to get a first entry burn intense enough to topkill the invading understory and midstory species.

Research in other forested communities has determined a fire line intensity of 500 kW/m is needed to control (topkill) woody understory stems ≥1 m tall of the same genus (Sparks and others 1999). Using BEHAVE to estimate fire behavior of specific burns where management objectives were successfully achieved, we determined the fireline intensity of these burns was near or above the 500 kW/m threshold

Table 2.—Fuel model descriptions relative to Texas oak communities. Prescription conditions are for initial burns with desired effects including topkill of understory woody vegetation.

Fuel Model	Description	Prescription
TL1	Forests with closed canopy juniper overstory and/or midstory. The understory is completely covered with juniper leaf litter and little or no woody or herbaceous fuels. Fire is carried predominantly by the juniper leaf litter.	No surface fire prescription Requires high input management such as mechanical or herbicide
TL6	Upland forests with closed canopy hardwood (predominantly post oak and southern red oak) overstory and a relatively sparse midstory. The understory is sparse and covered with hardwood leaf litter. Very few herbaceous plants are present. Fire is carried predominantly by the hardwood leaf litter.	Temperature: 40-90 °F Relative humidity: 12-30% Wind speed (mid flame): 6-18 mph 10-hour fuel moisure: ≤9%
TL8	Closed canopy forests dominated by pine. The midstory is sparse or composed of dense pine, and the understory is sparse and completely covered in a heavy layer of pine needles. Fire is carried predominantly by the pine needle litter.	Temperature: 40-75 °F Relative humidity: 20-50% Wind speed (mid flame): 4-12 mph 10-hour fuel moisure: 8-12%
TL9	Forests with a very close canopy overstory of hardwood or pine. The midstory is dense with woody species. This fuel type also describes midstory conditions with a dense pine needle drape primarily on yaupon. The understory is sparse with very dense hardwood litter or pine needles. Fire is carried by a combination of leaf litter and/or shrubs with heavy needle drape.	Temperature: 40-75 °F Relative humidity: 28-65% Wind speed (mid flame): 3-8 mph 10-hour fuel moisure: 9-15%
TU2	Closed canopy to moderately open canopy pine or hardwood forest overstories. The midstory is usually very dense with hardwood trees and/or shrubs. The understory is dense and dominated by woody shrubs and saplings. Grass and other herbaceous plants are sparse. Fire is carried by a combination of leaf litter and understory shrubs.	Temperature: 40-80 °F Relative humidity: 15-35% Wind speed (mid flame): 4-12 mph 10-hour fuel moisure: <10%
TU3	Moderately open to very open canopy forests or woodland with pine or hardwood overstories. The midstory is sparse with few or no stems that don't affect fire behavior. The understory is dominated by herbaceous species. Fire is carried by a combination of leaf litter and understory grass. This fuel model is often used to describe upland woodlands that are in a condition similar to those that existed prior to European settlement.	Temperature: 40-80 °F Relative humidity: 20-60% Wind speed (mid flame): 3-10 mph 10-hour fuel moisure: <12%

(Sparks, unpublished data). It is important to identify weather parameters needed to achieve a fireline intensity of 500 kW/m in the most common fuel models. In these invaded oak woodlands, it appears that this type of fireline intensity is only achievable under very extreme conditions (RH ≤25 percent; Winds >10 mph).

Developing a Prescription for Long Unburned Oak Woodlands

We have spent the past 13 years attempting to use fire as the primary management tool for oak woodland/savanna restoration. In addition, based on this experience we have attempted to develop prescriptions for restoring these vegetation communities with fire as the primary management tool. Through trial and error we have developed a prescription that is effective at pushing an intense fire through degraded, long unburned oak woodlands and topkilling the invasive mid- and understory woody species represented by fuel model TL6 (Tables 1 and 2).

Early on we attempted fires with high winds (15-25 mph) and moderate humidity (RH 25-40 percent) with the thought that wind would push fire through the oak leaf litter. We discovered that this worked some days and not others. We then conducted burns at lowered relative humidity (20-30 percent) and observed that high winds and low relative humidity often were effective, but not always. We then began to closely monitor fuel dryness, specifically 10-hour fuel moisture contents and time since a precipitation event. After several burns, we discovered 10-hour fuel moistures to be a critical component of the prescription. Even with moderate winds (6-12 mph), if 10-hour fuel moistures are less than 9 percent and relative humidity <30 percent, fires were generally intense enough to meet management objectives. However, the larger the unit, the higher the winds needed to be to ensure burn out within a given operational period.

Timing

We have determined that exact timing of the burn is not as critical as is achieving the other prescription parameters needed to produce a fireline intensity of 500 kW/m. However, we have determined that we are more likely to meet prescription variables in late winter and early spring, when cold fronts drive down relative humidity, increase wind speeds and the longer days and warmer temperatures produce better drying conditions. Fall and early winter months can have days within prescription parameters, but leaf fall is often not until late November or early December, and days are short, reducing drying time between rain events. Longer days in early spring permit relative humidity to remain low and within prescription for several hours whereas in the winter it reaches its low in early afternoon hours after which it immediately begins to recover. In a typical year (during the last 10 years or so), there are only a handful of days meeting all of these prescription parameters. During drier years, typically during a La Niña mode, the number of potential days increases dramatically. For example, during winter and spring of 2010, observed fuel moistures rarely were within prescribed parameters, but in 2011 observed fuel moistures were within the defined prescription parameters nearly weekly and often for multiple days at a time. Because of rapidly changing conditions it is important to be ready and not pass up a potential burn day.

Monitoring 10-Hour Fuel Moisture

We used 10-hour fuel moisture sticks comprised of four connected ponderosa pine dowels and the Forester Fuel Moisture Scale (Model 11552) to measure 10-hour fuels (Davies and McMichael 2005). Through comparisons of our observed readings and those from Remote Area Weather Stations (RAWS) we have determined fuel moisture stick measurements to be less variable within a day, especially as fuel moistures are in the 6-15 percent range. Due to this variability, we recommend land managers utilize on site 10-hour fuel moisture to guide their prescribed fire planning instead of data derived from RAWS stations. With respect to planning, it is recommended to begin looking for an acceptable burn window when the on-site fuel moisture reaches 12 percent for more than 2 consecutive days.

Fire Control

The above prescription has proven to be very effective; however it can produce extreme fire intensities and carries greater risk and costs than typical cool-season maintenance burns. This prescription should only be attempted by experienced fire crews with substantial preparation such as using well established fire breaks (bare mineral soil) and having a contingency plan and on-site contingency resources.

Burning under these prescription parameters will increase the likelihood of igniting snags and logs, thus increasing the time for mop-up and residual smoke. If maintaining snags and logs is a critical management objective, proper mitigation procedures should be established prior to ignition to protect them. The intensity of fires under these prescription variables will most likely generate new snags within the unit, and if the first entry fire is successful, subsequent fires can be conducted under less severe conditions, reducing risk of losing future snags and logs. Under these conditions embers generated by the fire can go unnoticed for hours or even days, increasing the chances for spotting well after an area has been burned. “Punky” downed logs or snags are more susceptible to glowing embers than surrounding oak leaf dominated fuels. Embers may smolder for up to 4 hours in these “punky” fuels before generating enough heat to start a fire and catch the surrounding area on fire. It is recommended that a crew remain on site to patrol for an extended period of time following one of these intense restoration burns.

Fuel Model Transition

Fuel models TL6 and TU2 (Table 2) best describe fire suppressed habitats with little to no herbaceous material in the understory that require severe conditions to produce a fire intense enough to meet management objectives during the first burn cycle or two. As the understory and midstory become suppressed by the initial fire treatments and light reaches the forest floor, grasses and forbs flourish, and the fuel model may transition into a TU3. The TU3 fuel model can be burned under more moderate conditions and still meet management objectives of woody control.

Fire Effects on Vegetation

The inherent abilities of plants to respond to fire depend partially on the fire regime to which the plant community has adapted (NWCG 2001). Based on rapid regrowth and ability of understory plants to resprout, most of the plant species common in the Oak Woods and Prairies have adapted to frequent and often intense fires. In these long unburned habitats, the amount of dead woody fuel, depth of litter and duff layers, and amount of dead material within or around a living plant may be greater than would have typically occurred historically when fires occurred relatively frequently. In this situation, the effects of fire on vegetation are likely very different than it would have been under historic conditions because of the higher temperatures and longer durations of fires.

Fire-related plant tissue mortality is dependent upon both the temperature reached and the duration of time the tissue is exposed to that temperature. The lowest temperature at which plant cells die is between about 50 to 55 °C (122 to 131 °F) (NWCG 2001). Killing all belowground reproductive structures usually occurs only where there is a long duration surface heat source, such as beneath a large pile of woody debris that sustains almost complete combustion (NWCG 2001). The severity of burn relates to the depth of the litter layer beneath the vegetation and its moisture content when the fire occurs (NWCG 2001). These high severity conditions of initial burns typically results in consumption of all litter and organic matter beneath shrubs and killing of all buds and roots in or near the organic layer. This kind of fire favors shrubs with buds and roots buried deep enough that they escape lethal temperatures. Fires which occur where there are deep accumulations of litter beneath shrubs and isolated trees or significant amounts of dead lower branches that burn off and smolder beneath a shrub crown are more likely to lethally heat roots and reproductive structures than a fire that occurs where there is sparse litter and few dead branches.

Tree mortality is often the result of injury to several different parts of the plant, such as crown damage coupled with a high percentage of cambial mortality. Mortality may not occur for several years following a fire. Death can occur from secondary infection by disease, fungus, or insects because the resistance of plants to these agents is often lowered by injury, and wound sites provide an entry point for pathogens (NWCG 2001). A tree weakened by drought, either before a fire or after wounding, is also more likely to die (Dickinson and Johnson 2001). A strong negative relationship exists between burn severity and postfire sprouting (based on personal observations).

High severity fires occur when large dead fuels and organic layers are dry (<12 percent 10-hour fuel moisture). These fires typically consume all litter, twigs, and small branches, most or the entire duff layer, and some large diameter dead, down woody fuels, particularly rotten material. Significant soil heating occurs, especially near fuel concentrations. Post oak often possesses relatively thick bark that will protect it from moderately intense fires that could damage the cambium. However, overstory post oak trees can be killed in high severity fires, particularly in areas where burning of dense fuel concentrations at the base of individual trees leads to localized intensities that kill the cambium and/or canopy buds.

Common woody species such as yaupon, American beautyberry, and saw greenbriar have thick, deep root caudices or tubers that aggressively resprout from the base of the plants following fire, including intense fires that occur in long unburned habitats. Other woody encroaching species such as winged elm and eastern redcedar rapidly colonize bare soil created by fire. Repeated and frequent fires (<2 years) are often necessary to kill the entire plants. Resprouting woody species after two or three fires continue to resprout but are often less vigorous and produce fewer stems. Reducing the total cover of these woody encroachers and creating open canopies under dead overstory trees then allows native perennial early successional herbaceous species such as broomsedge (*Andropogon virginicus*) and purpletop to begin to colonize. Herbaceous species adapted to these fire-suppressed habitats such as woodoats and nutrush also readily resprout from root bases and are seldom completely killed by these intense fires. Woodoats often becomes very dense in these severely burned areas immediately following fire. Early successional herbaceous species such as rosettegrasses, fireweed (*Erechtites hieracifolia*), and American pokeberry (*Phytolacca americana*) also colonize the bare soil created by these intense fires and often temporarily become the dominant understory plants. These annual species rapidly disappear in successive years following these high severity fires. As native herbaceous species gradually become more common and the woody encroachers are reduced, fire climax species such as little bluestem can slowly begin to colonize and eventually become common in the understory.

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FIRE MANAGEMENT AND WOODY INVASIVE PLANTS IN OAK ECOSYSTEMS

Joanne Rebbeck

Research Plant Physiologist, U.S. Forest Service, Northern Research Station, 359 Main Road, Delaware, OH 43015. To contact, call 740-368-0054 or email at jrebbeck@fs.fed.us.

Abstract.—The use of prescribed fire to sustain oak forests has increased rapidly in the last decade as the threat of poor regeneration and increased dominance of shade tolerant or fire sensitive tree species grows. While prescribed fire can favor oak regeneration, it may also increase the invasion and expansion of nonnative invasive plant species (NNIS). Little is known about the effects of fire on invasives in eastern U.S. oak forests. The majority of knowledge regarding the response of NNIS is anecdotal as managers have reported the expansion of invasives such as tree-of-heaven (*Ailanthus altissima*) and princess tree (*Paulownia tomentosa*) following prescribed fires. How can prescribed fire be used so that it does not facilitate the expansion of NNIS? Managers need to be proactive and integrate NNIS control strategies into prescribed fire and timber management programs at a landscape level. This paper presents a review of the current state of knowledge regarding woody NNIS and the use of prescribed fire in the Eastern United States. It includes a summary of the common traits of NNIS, potential responses of woody NNIS to fire, proactive approaches to managing for NNIS, and a list of recommended resources related to NNIS and forest management practices. In addition, an overview of the author's current work on the effects of prescribed fire on *Ailanthus* in southeastern Ohio is presented.

INTRODUCTION

Active fire suppression and the conversion of former agricultural lands over the last 80 plus years have produced dramatic changes in forest structure and composition within the eastern United States (Dibble and others 2008). Fire intolerant species such as red maple (*Acer rubrum* L.) often dominate the sapling and midstory layer creating dense shade conditions. In contrast, oaks (*Quercus* spp.) are disturbance-dependent species (Abrams 1996). Forest managers often utilize some combination of harvesting, fire, and herbicide treatments which alter light levels to favor oak establishment (Brose and others 2008). Unfortunately, many nonnative invasive plants also have the potential to thrive in these same disturbed habitats or open conditions. Oaks have many disturbance-type adaptations that confer protection from fire, such as root-centered growth, thick bark, vigorous sprouting ability, and well-protected buds (Dey and Fan 2009). However, they do not compete well with faster growing native and nonnative plant species, especially on higher quality sites (Johnson and others 2009). It is likely that competition between oaks and nonnative invaders can also occur on dry, low fertility sites since some woody nonnative plants (e.g., *Ailanthus*, *Paulownia*) can thrive on poor sites. Nonnative plant species have become established at varying densities and distributions within forest edges and interiors. There are some NNIS species, such as the shade tolerant oriental bittersweet (*Celastrus orbiculatus*), which becomes established in undisturbed forest canopies. Once established, it and many other NNIS can persist for extended periods in a suppressed condition until a disturbance occurs that increases light and other resources, allowing rapid growth and expansion.

Reintroducing fire to oak ecosystems may increase susceptibility to nonnative plant invasion which may impede the restorative effects of burning. Recent work suggests fire alone does not increase light levels enough to favor the growth of regenerating oak (Hutchinson and others 2005). However, prescribed fire following a shelterwood harvest has been a successful prescription to promote oak recruitment and retention (Brose and Van Lear 1998). This treatment creates a higher level of disturbance than fire alone, and thus increases concerns of NNIS expansion. When invasive plants are present or nearby, forest managers have reported increases in their abundance following prescribed burns plus timber harvesting.

Management recommendations to control NNIS within eastern forests have improved in the last decade (Evans and others 2006, Webster and others 2006). Until recently there has been limited research addressing how NNIS respond to fire in the Eastern United States. Huebner (2006) evaluated 17 common invaders for their response to fire and their potential to alter fire regimes. She reported that 12 species were potentially controllable with repeated growing season burns while 5, including *Ailanthus*, had the potential to increase in abundance in response to fire. Kuppinger and others (2010) found that *Paulownia* quickly established across varying habitats immediately after fire but over time failed to persist except in resource-limited habitats. The use of fire to control NNIS in the East has been tested with a few species such as Japanese stiltgrass (*Microstegium vimineum*) and Japanese barberry (*Berberis thunbergii*) with some success (Flory and Lewis 2009, Ward and others 2009). However, using prescribed fire to control NNIS should proceed with caution given the sparse information currently available. In this paper, I focus on how biological and ecological traits of woody invasive species are intertwined in predicting the impact of fire and other management activities on their abundance and expansion. The use of prescribed fire to control NNIS will not be addressed. I give an overview of an ongoing study of the effects of fire on *Ailanthus*, as well as an efficient and cost-effective invasive plant mapping tool. A proactive NNIS management action list is also provided.

COMMON TRAITS OF INVASIVE SPECIES

Whether the management goal is to promote or to control a given species, it is important to understand its biology. Knowledge of an invasive plant's life history is critical to predicting how it might respond to fire and other management treatments. Consideration must be given to its life form. Is it an annual, biennial, woody perennial tree, shrub, or vine? Is it dioecious, producing male and female plants such as *Ailanthus*, common buckthorn (*Rhamnus cathartica*), and autumn olive (*Elaeagnus umbellate*)? Monitoring the local seasonal patterns of a species' growth to include time of leaf break and leaf drop, flowering, and onset of dormancy is invaluable in developing the most effective fire, mechanical, or chemical control treatments. If treatments occur during periods when energy reserves are at their lowest levels, they will be most effective in reducing sprouting compared with periods of higher reserves.

Disturbance-dependent species are often shade intolerant and drought tolerant. They often have root-centered growth with large carbohydrate stores which can be readily utilized when growth-limiting shade is removed. Some NNIS such as *Ailanthus* produce extensive root systems that share carbohydrate reserves that support multiple shoots growing across large areas in varying light environments. Conversely, shade intolerant species display shoot-centered growth, quickly occupying space and gaining competitive height over slower-growing species.

Shade tolerant NNIS such as Amur honeysuckle (*Lonicera maackii*) typically are the first to leaf out in the spring and last to drop leaves in fall. The Oriental bittersweet vine can grow and persist in undisturbed forests until a canopy disturbance releases it to grow rapidly (Webster and others 2006). Linking the timing of fire and other methods of control to low energy reserves is essential to limit invasives. The challenge lies in trying to minimize harmful effects on native vegetation while at the same time controlling the NNIS.

There are a number of traits that are often associated with invasive nonnative plants. These may include but are not limited to: (1) ability to sprout, (2) prolific seed production, (3) seed dispersal by wind or birds, (4) rapid growth rate, (5) early age to seed production, (6) ability to form pure stands, and (7) presence in early successional plant communities.

Other important attributes of NNIS to consider include: (1) how long do seeds remain viable; (2) is the plant allelopathic (i.e., does it produce chemicals that inhibit native seed germination and seedling establishment); (3) is the species shade tolerant; and (4) can it persist suppressed in the understory for extended periods of time? These are all crucial in understanding how individual NNIS may respond to fire and other disturbances. Unfortunately, basic biological and ecological information is incomplete or lacking for a number of the common NNIS found in eastern U.S. oak forests, as demonstrated by the summary provided in Table 1.

POTENTIAL RESPONSES TO FIRE

It is expected that many woody NNIS will respond to prescribed fire in a similar manner to that of native vegetation. Common responses and postfire conditions that favor early successional invasives can include:

- **Stem topkilling with resprouting.** Depending on fire severity and timing of the fire, stems may be topkilled followed by the sprouting of dormant buds. *Ailanthus* is a prolific sprouter with an extensive shallow root system with many potential bud sources (Kowarik and Saumel 2007). Like oak, it appears to respond positively to fire with vigorous sprouting. But unlike oak, it can also expand by vigorous root suckering similar to aspen and sassafras (Burns and Honkala 1990).
- **Increased understory light and available soil nutrients.** Increases in resource availability often favor the growth and expansion of NNIS.
- **Consumption of litter layer and exposure of mineral soil.** As with oak, these conditions often favor seed germination and seedling establishment, especially those that require light for germination such as *Paulownia*.
- **Soil disturbance.** Site preparation such as fire breaks can create exposed bare soil.
- **Spread by equipment.** Movement of dozers, trucks, and other vehicles from one site to another can facilitate the spread of invasive plants across a landscape. Both seeds and small viable vegetative fragments can be transported.

In addition, topographic effects on fire severity and intensity can create a variety of microsites within a burned landscape. If thinning treatments are combined with a prescribed fire or if a fire is severe enough to cause overstory mortality, patches of invasive plants can occur in the openings). This can promote NNIS movement towards forest interiors. Depending on site features and fire severity, postfire competition among new propagules (e.g., wind-blown or bird-dispersed seeds) or sprouts can be very high. Consideration must be given to the ability of a NNIS to successfully occupy a site after a burn as well as its ability to reproduce in that post-disturbance community.

CURRENT STATE OF KNOWLEDGE

A comprehensive synthesis of available scientific and technical information regarding the interactions between fire and nonnative invasive plants is available through the U.S. Department of Agriculture Forest Service's Fire Effects Information System (FEIS) (USDA FS 2012). Detailed information for each of 69 invasive species includes: distribution and occurrence,

Table 1.—Ecological and biological traits of common problem nonnative woody invasives in eastern U.S. oak forests*

Common and Latin Name	Shade Tolerance	Breeding System	Annual Rate of Seed Production	Seed Dispersal	Age to Produce Seed	Seed Bank	Growth Rate	Sprouting Ability	Competitive Ability	Successional Status
Trees										
Ailanthus (tree-of-heaven),[†] *A. altimissa*	moderate	dioecious	very high (>200,000 /female)	wind	~5 yrs	1-2 yrs?	rapid (3-6 ft/yr)	vigorous	high; forms dense thickets	early, but can persist
Norway maple, *Acer platanoides*	very tolerant	dioecious	high	wind	?	1 yr?	moderate?	limited	suppression via dense shade	early, but persists
Paulownia (princess/ empress tree), *Paulownia tomentosa*	intolerant	perfect flowers	very high (>20 million/ tree)	wind	3-5 yrs	2-3 yrs	rapid seedling & sprout growth	very high	transient, high as seedling; less with age	early, can persist
Shrubs										
Autumn olive, *Elaeagnus umbellata*	moderate	primarily dioecious	very high (66,000 /mature shrub)	birds	3-5 yrs	?	?	vigorous	displaces natives, can form dense thickets	early
Bush honeysuckles, (Amur) *Lonicera* spp.	moderate to tolerant	perfect flowers, not-well studied	variable, up to 1 million/ plant	birds	5-8 yrs	~2-3 yrs	Rapid in high light	yes	leaves first to expand & last to drop; rapid seedling height growth	early, but can persist as large dense mature shrubs
Common buckthorn, *Rhamnus cathartica*	tolerant	typically dioecious	heavy	gravity and birds	5-6 yrs	several yrs	medium to fast	yes	forms dense thickets	varies: early and mid-late
Japanese barberry, *Berberis thunbergii*	tolerant	perfect flowers	varies with genotype	birds	?	~1 yr	varies with light	yes	form dense thickets	early, but persists
Multiflora rose, *Rosa multiflora*	tolerant	?	500,000/yr	gravity & birds	?	10-20 yrs	?	yes	form dense impenetrable clumps	early
Privet (Chinese, European, & Japanese), *Ligustrum* spp.	moderate to tolerant	perfect flowers	hundreds/ mature shrub	birds	?	none	not reported	yes	form dense thickets	early, but can persist

(Table 1 continued on next page)

Table 1 (continued).—Ecological and biological traits of common problem nonnative woody invasives in eastern U.S. oak forests*

Common and Latin Name	Shade Tolerance	Breeding System	Annual Rate of Seed Production	Seed Dispersal	Age to Produce Seed	Seed Bank	Growth Rate	Sprouting Ability	Competitive Ability	Successional Status
Vines										
Japanese honeysuckle, *Lonicera japonica*	moderate to tolerant	requires cross-fertilization	prolific	birds	3-5 yrs	?	rapid once established	yes	aggressive colonizer, creates dense mats & thickets	early, but persists
Kudzu, *Pueraria montana* var. *lobata*	shade intolerant	?	infrequent	infrequent seeder	3 yrs	?	rapid (65-98 ft/yr)	high	aggressive asexual colonizer	early, but persists
Oriental bittersweet, *Celastrus orbiculatus*	tolerant & intolerant	both dioecious & perfect	abundant	birds	?	1 yr	rapid post-disturbance	high	overtops and girdles, can form thickets	early, but persists

*Primary reference source is the Fire Effects Information System Database (USDA FS 2012)
†Demonstrated to be allelopathic
? Represents lacking or unknown information

botanical and ecological characteristics, fire effects, and management considerations. In 2011, the FEIS staff summarized the knowledge gaps of eastern U.S. invasive plants (Gucker and others 2011). They concluded that current information in the eastern United States is very limited and largely anecdotal. In addition, the transfer of research findings to managers needs improvement so they can make better informed decisions. They also recommend more experiments and observations of how NNIS respond to fire, including: (1) several years of data in a particular ecosystem, (2) various burning conditions, (3) season of burn, (4) varying fire severities, and (5) varying intervals between burns. It is noteworthy that these same research questions are also critical in developing better fire prescription guidelines for promoting oak restoration.

In 2008, the federal interagency Joint Fire Science Program (JFSP) made a request for research proposals that specifically addressed the interaction of invasive plants and fire in the eastern United States. Two main questions were addressed. First, what are the effects of nonnative invasive plant species on fire behavior and fire regimes? Secondly, what are the effects of fire and fire management on the distribution of NNIS? The three funded studies are:

1. Fire and the invasive Japanese stiltgrass (*Microstegium vimineum*) in eastern deciduous forests. Principal investigators: S. Luke Flory and Keith Clay, Indiana University.
2. To burn or not to burn Oriental bittersweet (*Celastrus orbiculatus*): a fire manager's conundrum. Principal investigators: Noel B. Pavlovic and Stacey Leicht-Young, U.S. Geological Survey.
3. Prescribing fire in managed oak forest landscapes: interactions with the invasive tree *Ailanthus altissima*. Principal investigators: Joanne Rebbeck and Todd Hutchinson, U.S. Forest Service.

Research questions for each project are provided in Table 2. Since all are active at this writing, annual project reports and updated deliverables are available at the JFSP website http://www.firescience.gov/JFSP_Search_Advanced.cfm. Future technical publications are planned with management recommendations. An overview of the *Ailanthus* project in Ohio follows.

Table 2.—Summary of research questions of Joint Fire Science Program funded projects investigating the interaction of fire and nonnative invasive plants in the eastern United States

Plant species	Life form	Study location	Research questions
Japanese stiltgrass	Annual grass	Big Oaks National Wildlife Preserve, SE Indiana	How do stiltgrass invasions affect fire behavior? How do prescribed fires and timing of fires affect invasions? Do invasions alter the effects of fire on native species and habitats? Can fire be used as a tool to manage invasions?
Oriental bittersweet	Perennial vine	Indiana Dunes National Lakeshore, Lake Michigan	How does fire impact seed viability? How does fire influence the susceptibility of different habitats to invasion? What are the effects of fire on established plants? Can bittersweet presence and abundance be predicted in a fire landscape?
Ailanthus	Tree	Tar Hollow State Forest, SE Ohio	What is the influence of past harvesting and fire history on the distribution and abundance of *Ailanthus* in a forested landscape? What are the direct effects of fire on the demography of Ailanthus populations?

FIRE AND *AILANTHUS* IN SOUTHEASTERN OHIO: A CASE STUDY

Background

Ailanthus is a common NNIS in the eastern United States and can invade and expand dramatically when forests are disturbed. It is extremely fast-growing, reaching heights of 80-100 feet. It is dioecious and is a prolific seeder with up to 350,000 seeds produced per tree in a single growing season (Kowarik and Saumel 2007). Seeds develop in the summer mature in early fall and can persist until March. Wind-dispersed seeds can travel in excess of 330 feet (Landenberger and others 2007). The longevity of seeds in soil is not fully known. It appears that seeds are short-lived, typically 1-2 years, but germination rates are very high (80-100 percent) in disturbed stands. In addition, *Ailanthus* is capable of aggressive clonal spread, often creating dense thickets that can outcompete native trees. While considered shade intolerant, clonal sprouts attached to a parent tree can persist in a shaded forest understory for up to 20 years (Kowarik 1995). Vigorous sprouts can develop 50-90 feet from a parent tree (Illick and Brouse 1926). Although the long-term effects of *Ailanthus* on native tree regeneration are not known, it likely has a negative impact because of its highly competitive traits and production of the allelopathic compound ailanthone (Hiesey 1996).

Very little is known about the direct and immediate effects of fire on *Ailanthus*. Saplings are easily topkilled by fire, but resprouting is prolific (Lewis 2007). Managers have reported observing increases in *Ailanthus* via seed germination immediately following fires. However, in landscapes with very small populations of *Ailanthus*, it does not invade burned sites (e.g., Hutchinson and others 2005). It remains unknown whether a postburn *Ailanthus* expansion will inevitably occur if the propagule pressure is high. It may be that fire, by reducing litter and increasing light, creates improved conditions for *Ailanthus* establishment as other disturbances have been shown to do (Kota and others 2007).

At Tar Hollow State Forest in southeastern Ohio, thinning and burning treatments were conducted as part of the national Fire and Fire Surrogate Study. The explosive expansion of *Ailanthus* from seed followed the combined thin plus burn treatment (Hutchinson and others 2004, Rebbeck and others 2005). *Ailanthus* trees (≥4 in d.b.h.) were present but not abundant in this unit prior to treatment, and no seedlings were observed. Both treatments occurred in the same dormant season, and 2 years later, *Ailanthus* seedlings were present in 96 percent of plots at a mean density of >5000 per acre. However, it appears that the combination of propagule pressure and disturbance may have been critical, because in the adjacent thin only and burn only units, there were few if any adult *Ailanthus* prior to treatment (low propagule pressure) and there was not a large increase in *Ailanthus* after those treatments.

The Ohio Department of Natural Resources, Division of Forestry has an active prescribed fire program to sustain oak forests (Bowden 2009). Between 2001 and 2008, a total of 14 prescribed fires have been applied to 2,100 acres within a 9,600 acre portion of the Tar Hollow State Forest. The goal of our research project is to gain a better understanding of how the distribution and abundance of *Ailanthus* is related to recent fires, past timber harvesting, seed sources, and other landscape and stand characteristics. The study area is located within the Southern Unglaciated Allegheny Plateau. The topography is highly dissected, consisting of sharp ridges, steep slopes, and narrow valleys.

Research Approach

Geo-referenced digital aerial sketch mapping technology (DASM) (Schrader-Patton 2003) in a low-flying helicopter was employed to identify seed-producing female trees and patches of *Ailanthus* in winter 2008 when persistent seeds were highly visible (Fig. 1). During a 2 hour flight, 98 seed-bearing females and 42 patches, ranging in size from 0.44 to 33 acres, were identified within a 9600 acre area (Rebbeck and others 2010). Aerially-identified females

Figure 1.—Aerial view of female *Ailanthus* trees with seeds during a 2011 winter helicopter survey on the Marietta District of the Wayne National Forest. Photo is courtesy of Tom Shuman, ODNR Division of Forestry.

were ground-truthed with 95.7 percent accuracy using hand-held Global Positioning System (GPS) units. The method appears to be an effective and efficient tool to map the distribution of *Ailanthus* seed sources in a landscape. Additional aerial mapping of more than 164,000 acres is underway in cooperation with the Wayne National Forest and Ohio Division of Forestry. In the course of 6 days, 78,000 acres were surveyed on the Wayne National Forest at an estimated cost of $0.40 per acre.

A systematic grid of research plots was installed to quantify the abundance, size, and age distribution of *Ailanthus*, as well as other stand attributes (e.g., fire, harvesting, and stand structure). Past timber harvest records were digitized and incorporated into a Geographic Information System (GIS) database. Current data analysis includes GIS and statistical techniques such as classification and regression trees (CART) and Random Forests (Prasad and others 2006) to determine the relationships of *Ailanthus* presence and abundance to distance and direction from seed-producing trees, fire, timber harvest, and other landscape attributes. Relationships are being tested to determine the extent to which burning and/or harvesting favors the spread of the *Ailanthus* and to determine possible mechanisms of its expansion in this landscape.

To gain a better understanding of how *Ailanthus* responds to fire, replicated research plots were installed in 2009 to study the direct effects of prescribed fire and herbicide treatments on *Ailanthus* demography and spread. Herbicide stem injections of *Ailanthus* (hack-n-squirt with imazapyr) were completed in autumn 2009, and prescribed burns were completed in April 2010. Treatment effectiveness and postburn survival, topkill, sprouting, and establishment of *Ailanthus* are being monitored for three postburn growing seasons (Figs. 2 and 3). If *Ailanthus* successfully establishes and remains competitive, a second fire may be needed. A field tour and workshop as well as accompanying technical publications are planned to provide guidance on conducting prescribed fires in the presence of *Ailanthus* within managed oak forest landscapes.

Figure 2.—Basal sprouting of topkilled *Ailanthus* tree 10 weeks after early spring prescribed fire.

Figure 3.—Abundant *Ailanthus* regeneration on forest floor within gap created from an herbicide plus prescribed burn treatment.

RECOMMENDATIONS: PROACTIVE MEASURES FOR MANAGING INVASIVES

Benjamin Franklin's famous quote, "an ounce of prevention is worth a pound of cure" could be the centerpiece for adopting a proactive management approach of NNIS. It is much more efficient to eradicate a newly-introduced small NNIS population than control one that is large and well-established (Fig. 4). The U.S. Department of Agriculture (USDA) recommends utilizing early detection and rapid response management of invasive species and has a manager's tool kit at the National Invasive Species Information Center (http://www.invasivespeciesinfo.gov/toolkit/detection.shtml).

These are my recommended guidelines in the development of a successful management plan. However, keep in mind that some of these suggested guidelines may be extremely challenging to implement.

- Actively monitor for invasives.
- Develop "invasive plant" awareness in field staff.
- Know what invasives are present and learn their life histories.
- Stay current on new emerging invasive plants.
- It is easier to control a small population than large one.
- Eliminate invasives before they start reproducing.
- Focus treatment efforts on seed-bearing trees to minimize dispersal.
- Eliminate populations well in advance of a silviculture treatment such as fire or harvest (time required will depend on longevity of seed).
- Careful monitoring and follow-up is essential. Don't assume that the treatment was effective.
- Take a landscape approach to management. Observe adjoining stands and properties. Engage surrounding property owners to control invasives. Partner with other groups and work within a local cooperative weed management area.
- Use the appropriate herbicide with the appropriate method at the appropriate time of the year for more effective chemical control. Only stem inject woody perennials in fall when resources are being translocated to roots.
- Adopt clean sanitation practices. Keep logging or other cutting equipment clean to minimize spread. Use clean gravel and native seed mixes.
- Follow a triage treatment approach to focus on the least infested areas first. If may be more practical and efficient to only remove seed sources from extremely infested areas and ban any other silvicultural treatments.
- Utilize or develop a decision support system to prioritize treatment areas.

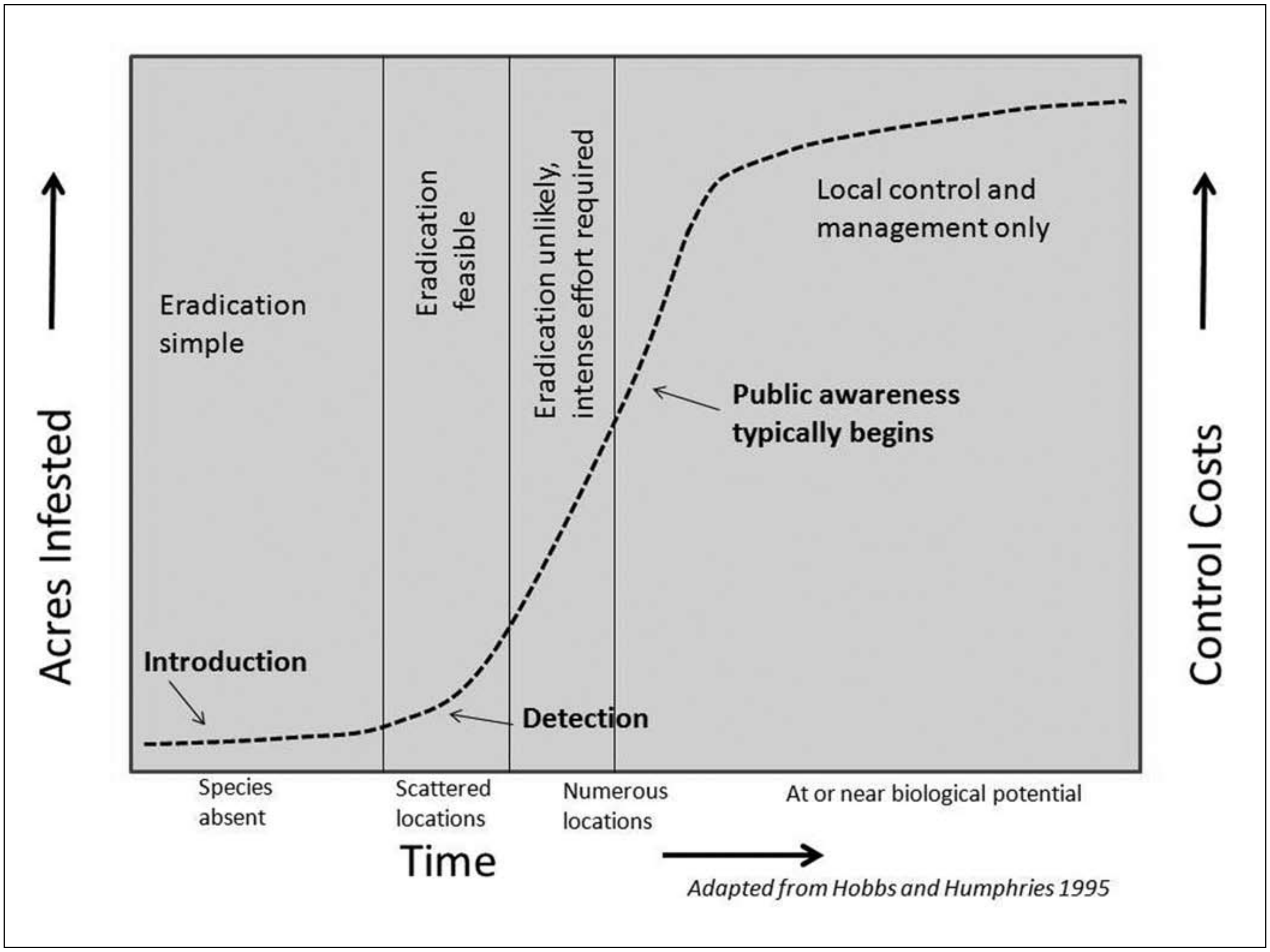

Figure 4.—Public awareness of invasive plants typically occurs after complete eradication is unlikely, increasing efforts and costs of control.

RECOMMENDED RESOURCES

1. **Mistaken Identity? Invasive Plants and their Native Look-alikes: an Identification Guide for the Mid-Atlantic.** A wonderful color photo guide with side by side comparisons of native and invasive plants that are difficult to distinguish.

Sarver, M.; Treher, A.; Wilson, L. [and others]. 2008. Dover, DE: Delaware Department of Agriculture. Mistaken identity? Invasive plants and their native look-alikes: an identification guide for the Mid-Atlantic. Available at http://weblogs.nal.usda.gov/invasivespecies/archives/2009/02/mistaken_identi.shtml.

2. **Fire Effects Information System (FEIS) Invasive Plants List.** This is a great resource, a one stop information portal for distribution and occurrence, biological and ecological characteristics, and management. It synthesizes available fire effects research on given species (includes peer reviewed and nonpeer reviewed literature). It was updated in 2011 to include 80 nonnative invasive plant species. Use caution when applying fire and management responses to other ecosystems and geographic regions.

U.S. Department of Agriculture, Forest Service. 2012. Fire Effects Information System, [Online]. Invasive plants list. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. Visit the following to access FEIS: www.fs.fed.us/database/feis/plants/weed/index.html.

3. **Wildland Fire in Ecosystems: Fire and Nonnative Invasive Plants.** An extensive review of information on NNIS and wildland fire. It covers a wide range of topics including management planning with an emphasis on the western United States.

Zouhar, K.; Smith, J.K.; Sutherland, S.; Brooks, M. L. 2008. Wildland fire in ecosystems: fire and nonnative invasive plants. Gen. Tech. Rep. RMRS-GTR-42-vol. 6. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 355 p. Available at http://www.treesearch.fs.fed.us/pubs/30622.

4. **Invasive Plant Responses to Silvicultural Practices in the South.** A guide to assist foresters and managers in the southern United States in developing management plans to reduce invasive plants. It is a good overview of integrating vegetation management guidelines and control techniques with silvicultural practices for other eastern U.S. regions.

Evans, C.W.; Moorhead, D.J.; Bargeron, C.T.; Douce, G.K. 2006. Invasive plant responses to silvicultural practices in the south. BW-2006-03. Tifton GA: The University of Georgia Bugwood Network. 52 p. View or download publication at http://www.invasive.org/silvicsforinvasives.pdf.

5. **A Field Guide for the Identification of Invasive Plants in Southern Forests.** An update of the 2003 edition. The guide displays distinguishing plant features throughout the year for accurate identification of 56 problematic invasive plants. Many species are found throughout the eastern region.

Miller, J.H.; Chambliss, E.B.; Loewenstein, N.J. 2010. A field guide for the identification of invasive plants in southern forests. Gen. Tech. Rep. SRS-119. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 126 p. View or download publication at http://www.treesearch.fs.fed.us/pubs/35292.

6. **A Management Guide for Invasive Plants in Southern Forests.** A companion to the Southern Forests Field ID guide described above. It provides great information on current management strategies and procedures for 56 invasive plants in a variety of habitats. Good overview on organizing, planning, and enacting invasive plant management and prevention programs.

Miller, J.H.; Manning, S.T.; Enloe, S.F. 2010. A management guide for invasive plants in southern forests. Gen. Tech. Rep. SRS-131. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 120 p. View or download publication at http://www.treesearch.fs.fed.us/pubs/36915.

7. **Invasive Plants Field and Reference Guide: An Ecological Perspective of Plant Invaders of Forests and Woodlands.** A handy ringed laminated guide suitable for field work. Provides key information for the accurate identification of invasive species, and gives a scientific synthesis of what is known about the behavior of these species in managed, disturbed, and pristine forested systems.

Huebner, C.; Olson, C.; Smith, H. 2005. Invasive plants field and reference guide: an ecological perspective of plant invaders of forests and woodlands. NA-TP-05-04. U.S. Forest Service, State and Private Forestry, Northeastern Area. View or download publication at http://www.treesearch.fs.fed.us/pubs/19822.

8. **A Guide to Nonnative Invasive Plants Inventoried in the North by Forest Inventory and Analysis.** The intent of this guide is to aid FIA field staff in identifying 44 invasive plant species in the 24-state Northern Research Station region (Maine south to Delaware west to Kansas and north to North Dakota). However, this guide can be used by anyone interested in learning about these invasive plants. It contains distribution maps, short descriptions, space for notes, and numerous pictures of each plant.

Olson, C. 2009. A guide to nonnative invasive plants inventoried in the North by Forest Inventory and Analysis. Gen. Tech. Rep. NRS-52. Newtown Square, PA; U.S. Department of Agriculture, Forest Service, Northern Research Station. 194 p. View or download publication at http://nrs.fs.fed.us/pubs/34183.

9. **Join a Regional Invasive Plants Listserve.** Subscribe to a regional invasive plants listserv to receive emails and updates to stay informed of emerging problem species and current control methods. Two such groups include the Midwest Invasive Plant Network at www.MIPN.org or the Mid-Atlantic Exotic Pest Plant Council website at www.ma-eppc.org.

SYNOPSIS

Land managers and researchers have much to learn about the interaction between fire and invasive plants in eastern oak forests woodlands and savannas. The use of fire alone should not be considered as a viable control option for invasives. Instead, an integrated management approach with early detection, removal, and monitoring should be utilized to minimize their deleterious impacts on oak ecosystems.

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ANIMAL POPULATIONS AND FIRE-MAINTAINED OAK ECOSYSTEMS

SONGBIRDS IN MANAGED AND NON-MANAGED SAVANNAS AND WOODLANDS IN THE CENTRAL HARDWOODS REGION

Frank R. Thompson III, Jennifer L. Reidy, Sarah W. Kendrick, and Jane A. Fitzgerald

Research Wildlife Biologist (FRT), U.S. Forest Service, Northern Research Station,
202 Natural Resources, University of Missouri, Columbia, MO 65211;
Research Specialist (JLR); Graduate Research Assistant (SWK); Joint Venture Coordinator (JAF).
FRT is corresponding author: to contact call 573-875-5341 ext. 224 or email frthompson@fs.fed.us.

Abstract.—We know little about the response of birds to savanna and woodland restoration in the Ozarks or how important such habitats are to birds of conservation concern. Bird species such as red-headed woodpecker, prairie warbler, field sparrow, and blue-winged warbler are species of regional concern, and declines of these species may be due to historical declines in savannas and woodlands. Our objective was to compare abundance of focal bird species between sites managed to restore savanna or woodland conditions and forested sites with no restoration management in the Ozark Highlands of Missouri and Arkansas during the breeding season. We consulted with local resource managers to identify sites they considered good examples of savanna or woodland restoration (managed sites) and also selected nearby stands on similar landforms that had no recent management (non-managed sites) and had succeeded to closed-canopy forest. We conducted 9 to 15 point counts along randomly located transects within these sites in 2007 and 2008. For species with >50 detections, we estimated density using distance sampling surveys, and for species with fewer detections we report the mean number of detections/point as an index of abundance. We conducted 260 surveys at managed sites and 244 at non-managed sites. Blue-winged warbler, eastern towhee, eastern wood-pewee, field sparrow, prairie warbler, and summer tanager were more abundant in managed sites whereas Acadian flycatcher, and worm-eating warbler were more abundant in non-managed sites. Abundance of blue-winged warbler, field sparrow, and prairie warbler decreased with canopy cover while Eastern towhee and summer tanager reached their greatest abundance in intermediate canopy cover. Eastern wood-pewee and prairie warbler were the most abundant breeding birds with 0.22 and 0.15 singing males/ha, respectively. Savannas and woodlands provide habitat for an interesting mix of grassland-shrub and canopy nesting birds that are of high conservation concern.

BACKGROUND AND OBJECTIVES

Savanna and woodland habitats historically covered a large portion of the Midwest and were the dominant habitats in the Ozark Plateau in the early 1800s (Nelson 1997). Before European settlement, oak savanna covered 11 to13 million hectares in the Midwest from Texas to Michigan, but only 2,607 hectares remained in1985 (Nuzzo 1986). Agricultural plowing, fire suppression, and forest succession eliminated most oak savannas (Peterson and Reich 2001). An estimated 24,700 to 49,400 hectares of restorable oak savanna maintain enough floristic diversity to justify preservation in the state of Missouri (Currier 1993).

There is not a clear distinction between savannas and woodlands. Savannas have been defined as 10-50 percent canopy cover, scattered trees and shrubs, and a ground cover of grasses, sedges, and forbs (McPherson 1997, Nelson 1985, Taft 1997), whereas woodlands have been defined as 30-80 percent canopy cover, open-grown trees, and ground cover of forbs, some woody plants, and grasses (Taft 1997). Fire historically maintained these open habitats by thinning the understory, reducing woody vegetation, and creating large openings in the canopy where sunlight reached the ground. This increased sunlight encourages floristic diversity with a dense ground flora consisting of grasses, sedges, and other composites that continue to grow throughout the summer as opposed to the herbaceous layer in a forest that peaks in the spring (Nelson 1985). Current efforts to restore savannas and woodlands use prescribed fire as the main management tool, although managers often use additional mechanical treatments consisting of selective tree removal to reduce tree density.

Savannas and woodlands are transitional habitats between the oak forests to the east and prairies to the west (McPherson 1997) and have been described as ecotones (Temple 1998). As ecotones, they have some biotic and environmental characteristics of adjacent systems (grassland and forest), as well as characteristics unique to the ecotone, enriching biodiversity of these habitats (Temple 1998). Current knowledge suggests these communities harbor few bird specialists (Grundel and Pavlovic 2007a). However, savannas and woodlands may represent important habitats for some species of birds, and bird species may have been eliminated or altered their habitat use when the availability of savannas and woodlands declined before researchers could gather baseline information (i.e., red-headed woodpecker [*Melanerpes erythrocephalus*]; Davis and others 2000). As such, their importance in the landscape remains understudied and poorly understood. Savanna and woodland restoration could potentially provide additional habitat for declining species that use grasslands or closed-canopy forests. Based on Breeding Bird Survey data, approximately 70 percent of 21 species associated with open woodlands and savannas are in long-term decline or are currently declining in eastern North America (Hunter and others 2001). In a study of breeding bird densities in central Minnesota, 9 of 20 species associated with savanna and woodland have been declining in abundance during the past 35 years (Davis and others 2000).

The use of prescribed fire to restore savanna and woodland affects vegetation structure and composition, which potentially influences habitat selection by birds (Brawn and others 2001). Avian species richness and densities appear to increase with restoration burns in the Midwest. Species richness and total density were greater on burned sites than unburned savanna restoration sites in central Minnesota (Davis and others 2000). Bird communities along a habitat gradient ranging from prairie to woodland had the greatest species richness in dry oak savanna with 5-65 percent canopy cover (Au and others 2008). Species of concern, such as blue-winged warblers (*Vermivora cyanoptera*), golden-winged warblers (*Vermivora chrysoptera*), and red-headed woodpeckers, were more abundant in savannas than prairies or woodlands (Au and others 2008). Frequent fires in savannas and woodlands in Indiana were positively correlated with species diversity and density of the most threatened species. Species with a preference for oak savanna habitat were at maximum density at an average fire frequency of once every 3 years (Davis and others 2000, Grundel and Pavlovic 2007b). Bird community structure was most strongly related to the use or absence of fire when managing savanna in the Minnesota study (Davis and others 2000). Savannas managed with cutting and no fire supported bird communities more similar to woodland than to other savanna sites managed with fire (Au and others 2008). While no specific cause was identified relating community structure to vegetation differences between the two habitats, the use of fire may shift a habitat enough to influence bird communities (Au and others 2008).

Our objective was to determine the response of focal bird species to savanna and woodland restoration in the Central Hardwood Region. We compared bird abundance between savannas and woodlands that were maintained or restored by active management (managed) versus potential savannas or woodlands with no recent management (non-managed) that had succeeded to closed woodland or forest. We hypothesized early successional species such as blue-winged warbler, brown thrasher (*Toxostoma rufum*), eastern towhee (*Pipilo erythrophthalmus*), field sparrow (*Spizella pusilla*), and prairie warbler (*Dendroica discolor*) would be most abundant in managed sites with the most open canopies or lowest basal areas. We hypothesized that forest species such as Acadian flycatcher (*Empidonax virescens*), Louisiana waterthrush (*Parkesia motacilla*), and worm-eating warblers (*Helmitheros vermivorum*) would be most abundant in non-managed sites with high canopy cover, and Eastern wood-pewee (*Contopus virens*), and summer tanager (*Piranga rubra*) would be most abundant on managed sites with intermediate canopy cover or basal area.

STUDY AREAS

We conducted our study in forests, woodlands, and savannas on public lands in Arkansas, Missouri, and Tennessee within the Central Hardwoods Bird Conservation Region (CHBCR) (Fig. 1). The CHBCR was comprised of >3 million ha of rolling hills covered primarily with hardwood forests (oaks [*Quercus* spp.] and hickories [*Carya* spp.]) interspersed with glades and woodlands and dissected by deep river valleys (Fitzgerald and others 2003). Other landforms within the region are steep-sided ridges and hills, karst terrain, gently rolling lowland plains, or bottomlands along major rivers with associated terraces and meander scars (McNab and Avers 1994). The Mississippi River floodplain bisected the CHBCR between Illinois and Missouri into two regions: the Ozark Highlands and Boston Mountains to the west, and the Interior Low Plateaus to the east (U.S. NABCI 2000).

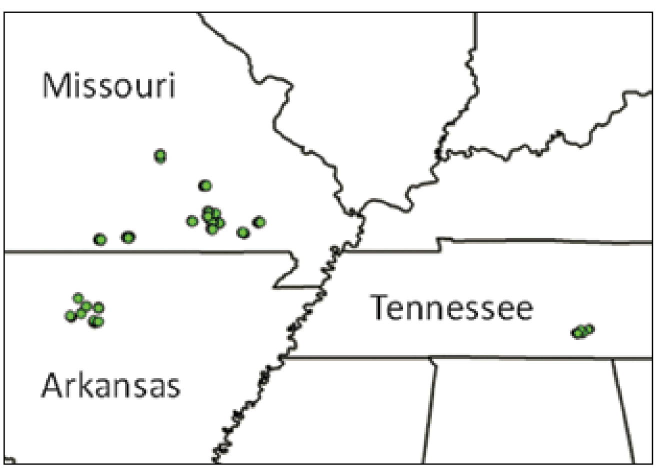

Figure 1.—Location of study sites where bird abundance was surveyed on managed savanna and woodlands and non-managed forest in the Central Hardwood Region in 2007-2008.

METHODS

Site Selection

We identified managed woodlands and savannas by contacting state and federal agencies. Stands selected were on ecological land types that were historically savanna or woodlands with a history of management that included repeated prescribed fires and had largely achieved the desired structure for a woodland or savanna. All managed stands had a history of prescribed burns and some had been thinned. Prescribed burns were conducted in late winter and early spring. We located non-managed stands within 10 km of each managed site; the non-managed stand consisted of forest on similar landforms and aspects as the managed stand but had no recent fire or tree harvest (>20 years). Stands ranged from 20 ha to >150 ha, and all sites were imbedded in a matrix of forest. We refer to the paired managed and non-managed stands as sites. The dominant tree species at these sites were black oak (*Q. velutina*), post oak (*Q. stellata*), red oak (*Q. rubra*), scarlet oak (*Q. coccinea*), white oak (*Q. alba*), hickory (*Carya* spp.), and shortleaf pine (*Pinus echinata*).

Bird Surveys

We conducted counts at points spaced at 250-m intervals on a grid starting from a random location. We located 9-15 points in a stand and did not place points within 50 m of the edge of the stand. Most points were substantially more than 50 m from a stand edge, and all stands were in a matrix of forest so edge effects were limited. We conducted one 10-minute survey at each point from 23 May 2007 through 30 June 2008. Counts were done from sunrise to 1000 hr during periods of no or low wind, no or light precipitation, and temperatures >10 °C. We counted the number of singing males for 10 focal bird species (acadian flycatcher, blue-winged warbler, brown thrasher, eastern towhee, eastern wood-pewee, field sparrow, Louisiana waterthrush, prairie warbler, summer tanager, and worm-eating warbler). We limited surveys to 10 focal species to ensure we were able to accurately collect the distance data needed for density estimation. For each singing male detected, we recorded the time of initial detection and the distance and direction to the bird from the observer. We measured distances using a Bushnell Yardage Pro laser range-finder (Bushnell, Overland Park, KS), but some distances were estimated when topography or vegetation precluded use of the range-finder. All observers received training on distance measurements prior to the surveys. At the end of each count, we measured canopy cover at the point by averaging two measurements with a spherical densiometer taken back-to-back at the point, and we measured basal area with a ten-factor prism.

We used two approaches to estimate abundance from surveys. We treated counts of detections of singing males during a 10 minute survey as a measure of relative abundance and fit generalized linear models evaluating treatment. Because not all individuals present during a count are detected by the observer, this approach assumes that the number of birds detected is related to the true abundance, and that variation in detectability does not contribute to bias in the results. For species with >50 detections, we used distance modeling to estimate detectability and true densities (Buckland and others 2001). More sophisticated analyses that address detectability require greater numbers of bird detections than we had for most focal species. These two approaches represent the tradeoffs between the more sophisticated modeling of treatments possible when considering relative abundance and addressing detectability to estimate densities.

Effects on Relative Abundance

For species with greater than 20 detections, we used a generalized linear model with a Poisson distribution to model the effects of treatment on bird relative abundance. The Poisson distribution is often used with count data and fit our data better than a negative binomial or normal distribution. While counts at individual points was the response variable in our models, we acknowledged the non-independence of points at sites by including a random effect for site in the model which treated sites as subjects and points as repeated measures. We hypothesized that treatment and canopy cover or basal area affected bird abundance. For our treatment effect we did not distinguish between savanna and woodland management because most stands had canopy cover in the range of woodlands. Rather, we simply considered stands managed or non-managed and included a continuous measure structure to capture additional variation. Because basal area and canopy cover are correlated, we determined which was more supported and included it in our models. The effect of canopy cover was supported more (n = 8) or equally (n = 1) compared to basal area, so we included canopy cover in models. We evaluated support for effects using an information theoretic approach to compare support for four a priori models based on Akaike's Information Criteria (AIC_c) adjusted for small sample sizes (Burnham and Anderson 2002). We included year as a fixed effect in all models to account for variability in abundance between the 2 years of the study. The null model consisted of an intercept and year effect. Additional models included the following fixed effects: Year + Treatment (managed or non-managed); Year + Canopy cover; and Year + Treatment + Canopy cover. We considered linear and quadratic forms of canopy cover and used the form most supported for each species. We interpreted relative support for models based on AIC_c to infer the importance of the factors considered and report model coefficients and predicted abundances as a function of covariates model-averaged across the set of candidate models to address model selection uncertainty (Burnham and Anderson 2002). We estimated predicted abundances for each covariate over its observed range while holding values of other covariates at their mean.

Density Modeling

For species with greater than 50 detections, we estimated density based on the distance to detected individuals at points, assuming detectability decreases with increasing distance between the observer and the detected individual. Analyses were carried out with Distance 6.0 (Thomas and others 2010) using three distance functions (a half-normal key function with cosine series expansion, a hazard-rate key function with simple polynomial series expansion, and a uniform function) and selected the most supported function based on Akaike's Information Criteria (Buckland and others 2001). We deleted observations at distances >100 m, which roughly corresponded to truncating the greatest 10 percent of recorded distances as recommended by Buckland and others (2001). We did not consider other factors affecting detectability such as observer, habitat, or date because the number of detections across each level of these factors was too sparse. We post-stratified estimates by treatment to estimate density and 95 percent confidence intervals for managed and non-managed sites.

RESULTS

We surveyed bird abundance at 244 points in non-managed stands and 260 points in managed stands located on 23 sites (Fig. 1). We surveyed 236 points in 2007 and 260 in 2008. We detected the following number of individuals for the focal species: eastern wood-pewees (257), summer tanagers (135), eastern towhees (106), prairie warblers (105), Acadian flycatchers (79), worm-eating warblers (61), field sparrows (44), blue-winged warblers (21), brown thrashers (7), and Louisiana waterthrushes (1). Because only one Louisiana waterthrush and seven brown thrashers were detected, we excluded them from further analyses. Median canopy cover was 82 percent (95 percent CI = 8, 96) in managed sites and 96 percent (95 percent CI = 80, 99) in non-managed sites (Fig. 2). Median tree basal area was 60 ft^2/acre (95 percent CI = 10, 150) in managed sites and 100 ft^2/acre (95 percent CI = 40, 170) in non-managed sites.

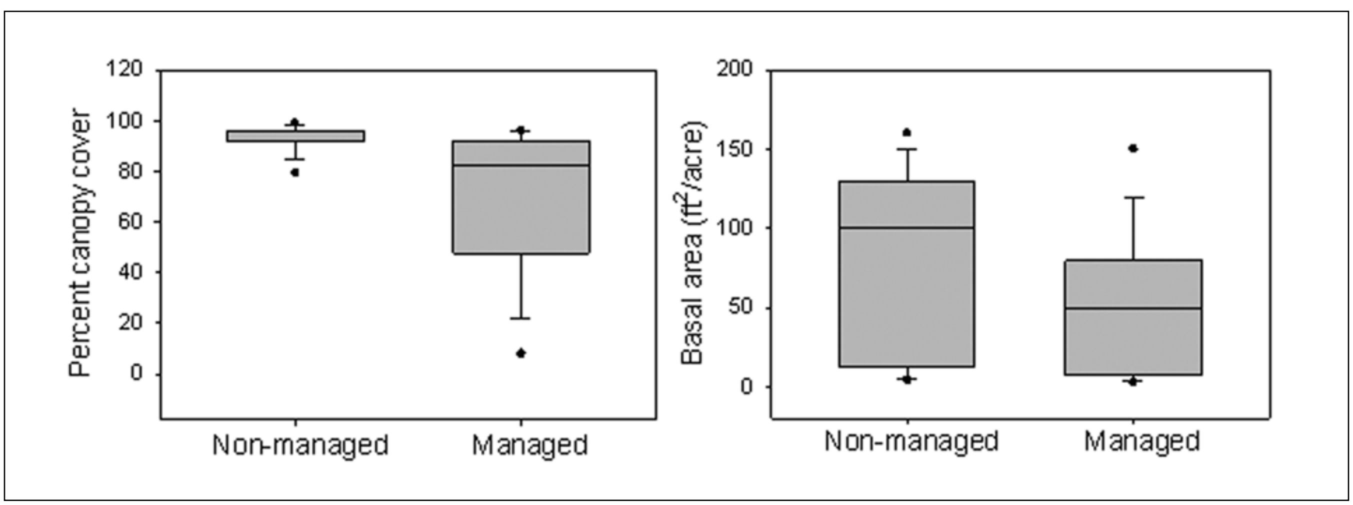

Figure 2.—Box plots indicating median, 25th and 75th, 10th and 90th, and 5th and 95th percentiles of canopy cover and basal area on managed savanna and woodlands and non-managed forests in the Central Hardwood Region in 2007-2008.

Effects on Relative Abundance

Treatment effects, canopy cover effects or combined effects of both were detected for all species analyzed (Table 1). Abundance of Acadian flycatchers and worm-eating warblers differed between managed and non-managed stands as expected for species breeding in mature forests. The treatment + canopy model received the most support for both species (Table 1). Abundances were greater in non-managed than managed sites (Table 2, Fig. 3) and increased linearly with canopy cover (Fig. 4), but canopy cover effects were not as strong as treatment effects and the confidence intervals overlapped zero (Table 2).

Blue-winged warbler, eastern towhee, field sparrow, and prairie warbler generally showed patterns in abundance consistent with our expectations for birds breeding in early-successional forest and were more abundant in managed stands or stands with more open canopies. Blue-winged warbler and field sparrow were most affected by canopy cover, eastern towhee by treatment, and prairie warbler by canopy cover and treatment (Table 1). These four species were all substantially more abundant in managed stands (Fig. 3). Abundance of blue-winged warblers and prairie warblers decreased linearly with canopy cover; a quadratic effect of canopy cover received some support for eastern towhee but after model-averaging there was no visible effect of canopy cover (Fig. 4).

Eastern wood-pewee and summer tanager nest in mature trees and are often considered forest species, but both were most abundant in managed stands (Fig. 3) Treatment had the strongest effect on eastern wood-pewee, while canopy cover had the strongest effect on summer tanager (Table 1). A quadratic effect of canopy cover was supported for summer tanagers and they reached greatest abundance at approximately 40 percent canopy cover (Fig. 4).

Table 1.—Support for four models predicting bird abundance in managed savanna and woodlands and non-managed forest in the Central Hardwood Region in 2007-2008 based on treatment (managed and non-managed) and canopy cover. Δ AIC_c is the difference in Akaike's information criteria adjusted for small sample size between the model and best model and ω_i is the weight of evidence for the model.

	Null		Canopy		Treatment		Treatment + canopy	
Species	**Δ AIC_c**	**ω_i**	**Δ AIC_c**	**ω_i**	**Δ AIC_c**	**ω_i**	**Δ AIC_c**	**ω_i**
Acadian flycatcher	17.5	0.00	6.9	0.02	0.9	0.39	0.0	0.59
Blue-winged warbler	3.7	0.07	0.0	0.47	1.8	0.19	1.1	0.27
Eastern towhee	100.4	0.00	64.1	0.00	0.0	0.70	1.7	0.30
Eastern wood-pewee	40.8	0.00	34.5	0.00	0.0	0.73	2.0	0.27
Field sparrow	44.2	0.00	0.0	0.51	20.5	0.00	0.05	0.49
Prairie warbler	74.3	0.00	40.7	0.00	4.2	0.11	0.0	0.89
Summer tanager	9.7	0.00	0.0	0.53	3.0	0.12	0.9	0.34
Worm-eating warbler	17.9	0.00	4.2	0.09	2.5	0.20	0.0	0.71

Table 2.—Model-averaged parameter estimates from generalized linear models with Poisson distributions used to predict bird abundance in managed savanna and woodlands and non-managed forests in the Central Hardwood Region in 2007-2008

		Year		Treatment		Canopy		Canopy^2	
	Intercept	**β**	**90% CI**	**β**	**90% CI**	**β**	**90% CI**	**β**	**90% CI**
Acadian flycatcher	-2.31	-0.70	-1.30, -0.09	-1.02	-1.61, -0.43	0.01	-0.01, 0.03		
Blue-winged warbler	-2.98	-0.86	-2.33, 0.60	0.38	-0.55, 1.31	-0.01	-0.03, 0.00		
Eastern towhee	-6.16	-1.24	-3.25, 0.76	4.67	2.82, 6.52	0.00	-0.009, 0.014	0.000	0.000, 0.000
Eastern wood-pewee	-1.37	-0.02	-0.32, 0.28	0.95	0.70, 1.20	0.00	-0.001, 0.001		
Field sparrow	-2.18	-0.02	-1.97, 1.94	0.48	-0.64, 1.60	-0.03	-0.043, -0.022		
Prairie warbler	-3.25	-0.06	-0.76, 0.65	2.64	1.81, 3.47	-0.01	-0.016, -0.001		
Summer tanager	-1.60	0.32	-0.05, 0.69	0.15	-0.21, 0.52	0.02	-0.006, 0.052	-0.000	-0.001, 0.000
Worm-eating warbler	-3.76	0.30	-0.22, 0.77	-0.81	-1.50, -0.13	0.02	-0.005, 0.045		

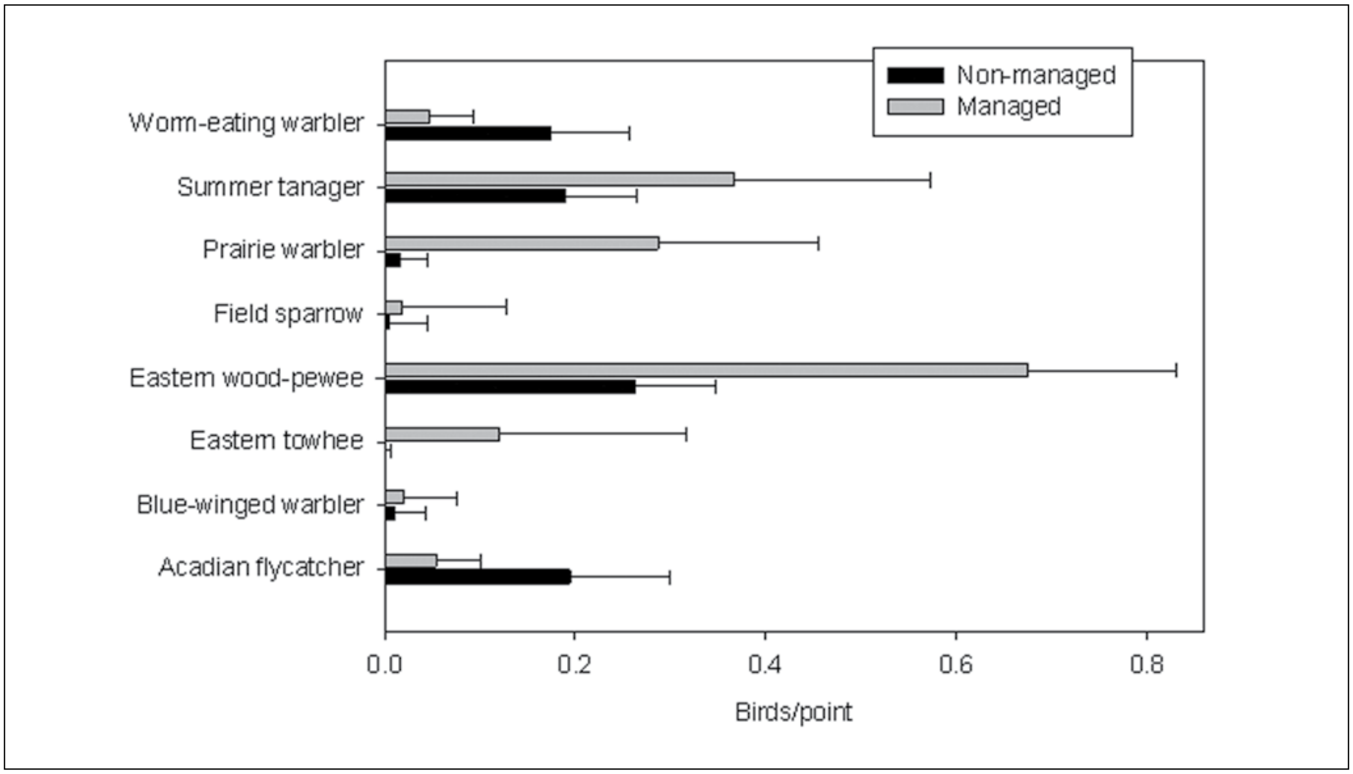

Figure 3.—Model-averaged predictions of relative abundance of birds (birds/count) predicted by generalized linear models on managed savanna and woodlands and non-managed forests in the Central Hardwood Region in 2007-2008.

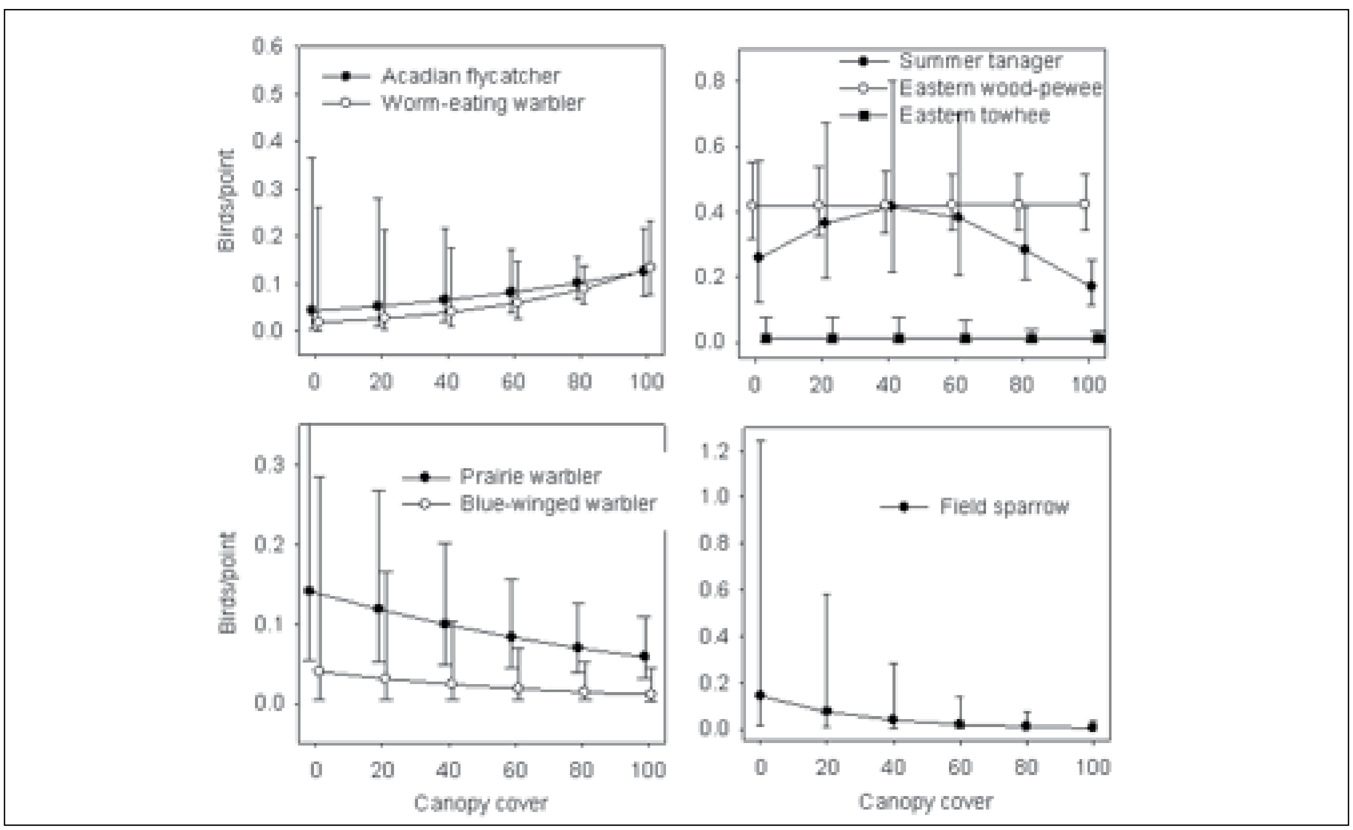

Figure 4.—Model-averaged predictions of relative abundance of birds (birds/count) predicted by generalized linear models as a function of canopy cover on managed savanna and woodlands and non-managed forests in the Central Hardwood Region in 2007-2008.

Densities

We were able to fit Distance models and estimate densities for six species at managed and non-managed sites (Table 3). Densities on managed and non-managed sites for each species followed a similar pattern to that predicted by the relative abundance models (Fig. 2). Eastern wood-pewees and prairie warblers were the most abundant species on managed sites. Acadian flycatchers and worm-eating warblers were the most abundant species on non-managed sites.

DISCUSSION

We found substantial differences in abundance of our focal species between managed and non-managed sites. Relationships to management and canopy cover generally followed our a priori expectations and reflect habitat needs of species from both ends of a successional gradient as well as some that seemed to prefer the intermediate conditions created by savanna-woodland management. Early successional species (e.g., blue-winged warbler, eastern towhee, field sparrow, prairie warbler) had greater abundance in managed sites or sites with the most open canopies, and abundance declined linearly with canopy cover. Savannas are typically defined by a dominant herbaceous ground cover, which would be achieved with frequent fire (short intervals). Allowing a longer interval between fires would presumably increase the woody growth and should support higher abundance of shrub-nesting species. The degree to which savannas and woodlands provide habitat for grassland, early-successional, and forest birds likely depends on micro-scale habitat features such as amount of ground cover that is herbaceous or woody and the structure of the canopy, as well as landscape features such as proximity to other suitable habitat patches, size of savanna or woodland stands, and composition of surrounding landscape.

Acadian flycatchers and worm-eating warblers were both less common on managed sites. Both species are typically associated with mid- to late-successional forests with a well-developed understory. Acadian flycatchers nest in understory trees such as flowering dogwood (*Cornus florida*). Worm-eating warblers nest on the ground and are often associated with deep leaf litter and high densities of shade-tolerant understory shrubs and trees. An objective of savanna-woodland restoration is often to remove the understory with the use of fire, so it is not surprising that these species were less common on managed sites. Conversion of forests to woodlands or savannas would likely negatively affect other forest-dependent species by removing the subcanopy layers and leaf litter ground layer.

Two species, eastern wood-pewee and summer tanager, reached their highest abundance in the intermediate canopy cover or tree densities that were found in managed savannas and woodlands. Both species are aerial insectivores, so the open canopy provides ideal conditions for foraging. Additionally, both seem to prefer to nest far out on horizontal

Table 3.—Estimated densities (singing males/ha) of birds in managed savanna and woodlands and non-managed forests in the Central Hardwood Region in 2007-2008. Density and probability of detection (p_d) were estimated by distance modeling, p-values > 0.05 (GOF) indicate no lack of fit.

			Non-managed		Managed	
	GOF	pd	Density	95% CI	Density	95% CI
Acadian flycatcher	0.87	0.41	0.165	0.115 - 0.237	0.034	0.024 - 0.049
Eastern towhee	0.49	0.68	0.002	0.001 - 0.003	0.113	0.076 - 0.169
Eastern wood pewee	0.49	0.69	0.081	0.062 - 0.107	0.220	0.167 - 0.289
Prairie warbler	0.95	0.48	0.006	0.004 - 0.008	0.144	0.106 - 0.195
Summer tanager	0.23	0.65	0.054	0.038 - 0.078	0.091	0.063 - 0.130
Worm-eating warbler	0.31	0.39	0.108	0.067 - 0.174	0.023	0.014 - 0.037

branches when possible, so open canopies also provide ideal structure for nesting. This contrasts with the other focal species which were associated more with either the most open or dense stands. Brawn (2006) similarly found eastern wood-pewees, summer tanagers, and eastern towhees were more abundant in restored savanna than forest.

Previously reported densities for blue-winged warbler, field sparrow, and prairie warbler are 0.3-0.4, 0.3-0.5, and 0.5-0.9 birds/ha, respectively, in glades and 3 to 5-year-old regenerating forest in the Missouri Ozarks (Fink and others 2006, Thompson and others 1992). Mean density of prairie warblers in managed sites in this study (0.11 birds/ha) was less than that reported in these earlier studies in the Ozarks, but this was likely because, on average, savanna and woodland stands had more canopy cover and less shrub cover than optimum for prairie warblers. Densities of Acadian flycatchers in non-managed sites (0.17 birds/ha) was not very different from densities previously reported in the Ozarks (0.2 birds/ha) (Thompson and others 1992).

Our results provide valuable information for conservation planners associated with the Central Hardwoods Joint Venture (CHJV), a public-private bird conservation partnership focused on the Central Hardwoods Bird Conservation Region. The CHJV staff and partners are charged with identifying habitat conditions and acreages needed to support desired population levels of priority species associated with both grass-shrubland and forest ecosystems as identified by Partners in Flight (Rich and others 2004). Little information was previously available that allowed planners to quantify priority species abundances in savanna and woodland ecosystems or to assess tradeoffs among grass-shrubland and forest-affiliated species if and when overstocked and degraded savannas and woodlands were restored. We showed that grass-shrubland bird populations could be increased with more widespread restoration of these ecosystems and that a loss of some number of forest-breeding birds would need to be offset by conservation efforts in appropriate forest landtypes.

While tree basal area and canopy cover were lower on managed compared to non-managed stands (Fig. 2), the average values for managed stands were in the upper end of that considered typical for woodlands. We believe this is because restoration of the sites we studied is still an ongoing process, and few sites had fully achieved the desired future condition. The high canopy cover and basal area may also be indicative of a need to more actively manage stands by thinning or that site conditions are more suitable for woodlands than savannas. Because managed savanna and woodland stands had greater canopy cover than typically described for these communities, these stands might have provided a slightly biased perspective of the bird communities expected in savannas and woodlands.

Future research should include a larger number of bird species to identify additional birds that reach high abundance in intermediate canopy covers or at managed sites, and also a larger suite of habitat characteristics, such as shrub and herbaceous variables. We did not include cavity-nesting species in this study, but it is likely that they reach higher abundances in these habitats given the increased snag density associated with more frequent fires. Additionally, to fully understand the value of savannas and woodlands as breeding bird habitat, we need to evaluate additional demographic parameters such as productivity. Better knowledge of the effects of timing, frequency of fire, and tree stocking levels is needed to develop best management practices for sustaining savanna and woodland communities.

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A REVIEW OF FIRE EFFECTS ON BATS AND BAT HABITAT IN THE EASTERN OAK REGION

Roger W. Perry

Research Wildlife Biologist, U.S. Forest Service, Southern Research Station, P.O. Box 1270, Hot Springs, AR 71902. To contact call 501-623-1180, ext. 108 or email at rperry03@fs.fed.us.

Abstract.—Fire is increasingly being used in oak forests to promote oak regeneration, improve wildlife habitat, and reduce hazardous fuel loads. Although recent research has begun to shed light on the relationships among fire, bats, and bat habitat, these interactions are not yet fully understood. Fire may affect bats directly through heat and smoke during the burning process or indirectly through modifications in habitat. Studies suggest fire generally has beneficial effects on bat habitat by creating snags, reducing understory and midstory clutter, creating more open forests, and possibly increasing abundance of flying insects. Direct effects of fire on bats during the burning process are still largely unknown. These potential direct effects likely differ for each species or roosting guild of bats, and may also vary by season and reproductive condition.

BACKGROUND

Fires ignited by lightning and Native Americans historically maintained a mosaic of forests, grasslands, savannas, and open woodlands throughout the eastern United States, including the eastern oak (*Quercus* spp.) region (Abrams 1992, Lorimer 2001). Prior to European settlement, fire frequencies ranged from every 3 to 19 years within this region (Guyette and others 2006). European settlement, logging, and clearing for agriculture altered these landscapes during the 18th and 19th centuries. During the 20th century, fire suppression caused many forests that were previously open and park-like to succeed to dense closed-canopy forests where fire-adapted plant species were replaced by shade-tolerant and fire-sensitive vegetation (Lorimer 2001, Nowacki and Abrams 2008, Van Lear and Harlow 2002). It is assumed that bats adapted to fire across these landscapes over thousands of years of frequent fire. However, changes in bat populations resulting from fire suppression-induced changes in habitat over the past century are difficult to differentiate from other anthropogenic effects such as disturbance of cave roosts, pesticides, habitat destruction, fragmentation, urbanization, and indiscriminant killing of bats that also occurred during portions of this period.

Fire may affect bats directly through heat, smoke, and carbon monoxide, or indirectly through modifications in habitat and changes in their food base (Dickinson and others 2009). Although use of fire by land managers has increased substantially in recent years, researchers have just begun to study the effects of fire on bats and bat habitat. Recent reviews have summarized the state of knowledge on the relationship between bats and fire. Carter and others (2002) provided a general review on effects of fire on bats in the southeastern United States and Mid-Atlantic States, and Keyser and Ford (2006) reviewed effects of fire on mammals (including bats) in eastern oak forests. A comprehensive review of direct and indirect effects of fire on bats, with an emphasis on Indiana bats (see Table 1 for scientific names of species), was presented by Dickinson and others (2009). My

Table 1.—Summer and winter roosting locations for bats found in the eastern oak region

Species	Summer Roosting	Winter Roosting
Eastern red bat (*Lasiurus borealis*)	Foliage, mostly hardwood canopies	Foliage, trees, low shrubs, leaf litter
Seminole bat (*L. seminolus*)[1]	Foliage, mostly pine canopies	Foliage, trees, low shrubs, leaf litter
Hoary bat (*L. cinereus*)	Foliage, pine and hardwood canopies	Unknown
Tri-colored bat (*Perimyotis subflavus*)	Foliage, mostly dead leaves in hardwoods, structures[2], caves, mines	Caves, mines, structures
Big brown bat (*Eptesicus fuscus*)	Tree cavities, under bark, structures, caves, mines	Caves, mines, structures
Rafinesque big-eared bat (*Corynorhinus rafinesquii*)	Tree cavities, under bark, structures, caves, mines	Caves, mines, structures, tree cavities
Southeastern bat (*Myotis austroriparius*)	Tree cavities, structures, caves, mines	Caves, mines, structures, tree cavities
Evening bat (*Nycticeius humeralis*)	Tree cavities, under bark, structures	Tree cavities, under bark, ground-level[3]
Silver-haired bat (*Lasionycteris noctivagans*)	Tree cavities, under bark	Tree cavities, under bark, caves, mines, structures, ground-level
Northern long-eared bat (*M. septentrionalis*)	Tree cavities, under bark, structures	Caves, mines
Little brown bat (*M. lucifugus*)	Tree cavities, under bark, structures	Caves, mines
Indiana bat (*M. sodalis*)	Tree cavities, under bark, structures	Caves, mines
Gray bat (*M. grisescens*)	Caves, mines	Caves, mines
Ozark big-eared bat (*C. townsendii ingens*)	Caves, mines	Caves, mines
Virginia big-eared bat (*C. t. virginianus*)	Caves, mines	Caves, mines
Eastern small-footed bat (*M. leibii*)	Caves, mines, under bark, structures, rock crevices	Caves, mines
Brazilian free-tailed bat (*Tadarida brasiliensis*)[4]	Caves, mines, structures	Caves, mines, structures

[1] Associated primarily with southern pine forests, rare in most of eastern oak region.
[2] Structures include buildings, bridges, culverts, water tanks, wells, and other man-made structures.
[3] Ground-level includes small mammal burrows, rock crevices, under tree roots, and other cavities or crevices on the ground.
[4] Resides primarily south of the eastern oak region.

objectives here are to expand upon those reviews and update information provided by recent studies. In the life history of bats, roosts and food are the two most important resources known to affect bat distribution and abundance (Kunz and Lumsden 2003). Therefore, I focus primarily on how fire in forests potentially affects these two resources. This discussion focuses on understory burning typically used for management purposes (not stand-replacing wildfire) and the effects of fire intensities typically found during prescribed burns.

Similar to birds, many species of bats undertake short (<500 km) or long-range (>1000 km) migrations between summer and winter use areas. Consequently, range maps often depict a species occurring throughout a wide area, but a species may occur only in portions of their range during each season (e.g., Cryan and Veilleux 2007). Furthermore, the ecology of bats often differs substantially among seasons. Thus, fire affects different species in different ways during each season, and burns conducted during winter will likely affect various species in different ways from those conducted between spring and fall.

ROOSTING AND FIRE

Bats may spend over half their lives in roosts, and roosts provide protection from predators and the weather (Kunz and Lumsden 2003). Therefore, adequate roost sites are critical to the survival of bats. Bats that occur in the eastern oak region use different types of roosts, and the type of roost used often differs between summer and winter (Table 1). The 17 bat species that occur or potentially occur in the region can be classified loosely into three guilds: foliage-roosting species; species that roost in cavities, crevices, or under bark of trees during all or part of the year (cavity- and bark-roosting species hereafter); and cave obligates. Many bats use different types of roosts between summer and winter. For example, red bats and tri-colored bats in forests roost mostly in foliage during summer; however, during winter, red bats continue to roost in forests whereas tri-colored bats retreat to caves or mines for hibernation. Therefore, differences in the natural history of each species determine how fire affects that species, and direct effects of fire on individual bats is likely dependent on season and intensity of fire, sex and age of bat, and reproductive condition of bat. Most tree-roosting bats switch roosts every 2-4 days (e.g., Lewis 1995); thus, an abundant supply of potential roost locations is needed to provide suitable roosting habitat within a forest stand.

Growing Season Fires and Foliage-Roosting Bats

During late spring, summer, and early fall (summer hereafter), four species that occur in the region roost in foliage (leaves or needles of trees and shrubs) during the day, including red bats, hoary bats, Seminole bats, and tri-colored bats (Table 1). Lasiurine bats (red, hoary, and Seminole bats) typically roost 15-19 m above the ground in the canopies of overstory trees (Hein and others 2008; Hutchinson and Lacki 2000; Menzel and others 1998; Perry and Thill 2007a, 2007b; Perry and others 2007a). Seminole bats roost primarily in the needles of pines, red bats roost mostly in deciduous foliage such as oaks, and hoary bats use both pines and hardwoods (Hein and others 2008; Hutchinson and Lacki 2000; Menzel and others 1998; Perry and Thill 2007a, 2007b; Perry and others 2007a; Willis and Brigham 2005). Tri-colored bats roost in a variety of locations during summer, including caves, mines, and buildings (Fujita and Kunz 1984). However, in forests lacking these substrates, they roost in foliage of trees (often oaks), and mostly in dead leaves or suspended clusters of dead pine-needles (Perry and Thill 2007d, Veilleux and others 2003).

Direct effects of growing-season burns on foliage-roosting bats are largely unknown. Because they typically roost relatively high (15-19 m) in the canopies of overstory trees during the warm season, it is unlikely that burning during the growing season leads to direct mortality. Further, carbon monoxide levels at roosting heights are unlikely a concern when flame lengths (fire intensity) are less than 1.6 m, which is typically the most intense fires observed during most prescribed burns (Dickinson and others 2010). Dickinson and others (2010) created models predicting potential burns to ears, wings, or other non-furred parts of bats based on roost heights, fire intensity (flame length), and wind speed. Winds may reduce the temperature and gas concentrations at roosting heights by distorting fire plumes. They suggest that bats roosting at heights above 12 m (wind speeds of 2 m/sec [4.5 mph]) to 22 m (wind speeds of 0 m/sec) would not be injured during intense controlled burns (flame lengths approximately 1.6 m high). Furthermore, they suggest models used to predict foliage scorch during prescribed burns (Reinhardt 2003) may potentially be useful as surrogates for predicting injury to bats.

When temperatures associated with fires are below lethal thresholds, heat and smoke could potentially disturb foliage-roosting bats regardless of roost height, causing them to relocate to other trees. However, foliage is not a limited substrate in most forests, and a substantial supply of alternative roosts is likely available if bats are disturbed (Carter and others 2002). Tri-colored bats, which tend to have great site fidelity to particular dead leaf clusters in hardwood forests (Perry and Thill 2007d, Veilleux and others 2003), may have more difficulty locating alternative roosts that provide adequate cover from predators after disturbance. Furthermore, because male tri-colored bats occasionally roost in suspended dead leaves close to the ground (<5 m high), they may be more susceptible to direct effects of fire than many other bats during summer (Perry and Thill 2007d).

Temperate, insectivorous bats undergo periods of torpor during which their metabolism is reduced and body temperatures may be similar to ambient air temperature. Bats control energy expenditure by regulating the frequency, depth, and duration of torpor (Speakman and Thomas 2003). Torpor lasting multiple days or weeks is considered hibernation. Because torpor can slow fetal development and reduce milk production (Racey 1973, Wilde and others 1999), females during the gestation and lactation periods of summer may spend less time in torpor than nonreproductive females or males (Cryan and Wolf 2003, Hamilton and Barclay 1994, Kurta and Fujita 1988, Solick and Barclay 2006). Because females are less likely to be in torpor during the reproductive season, they are likely able to escape fire more readily than males or non-reproductive females. However, non-volant young may not have the ability to escape heat and smoke from intense fires during the reproductive season. Nevertheless, females of many species (including most bats in the region) often carry non-volent young when roost switching, especially when disturbed (Davis 1970), and likely have the ability to escape oncoming heat if given sufficient warning.

Low-intensity burns during the reproductive season would likely have few negative effects on bats roosting in relatively tall (19-25 m) overstory trees. Furthermore, active bats maintain body temperatures of approximately 32-38 °C (90-100 °F) and may remain active above temperatures of 30 °C (86 °F) (Barclay and others 1996, Herreid and Schmidt-Neilsen 1966). Thus, bats may not be in torpor or may only be in shallow torpor during periods of relatively high ambient temperatures, which would allow for quick escape during fires. Dickinson and others (2010) suggested bats may arouse and escape fire in less than 10 minutes when temperatures are 25 °C (77 °F).

Dormant-Season Fires and Foliage-Roosting Bats

In southern portions of the eastern oak region, red bats and Seminole bats may be active during winter nights when temperatures are above freezing. During winter, red bats roost on the ground (under leaf litter) or 1-6 m above the ground in the lower branches of eastern red cedars or in persistent dead leaves of oaks or shrubs (Mormann and Robbins 2007). Red bats typically roost in trees when temperatures exceed 10 °C during winter (Mormann 2005). Seminole bats roost in the canopies of overstory pines and hardwoods, in understory vegetation, pine needle clusters suspended above the ground, or in pine litter on the forest floor during winter (Hein and others 2008). Evidence suggests hoary bats occur in the region during winter (Cryan 2003), but little is known about their winter roosting habits. Furthermore, little is known about the winter roosting habits of tri-colored bats outside hibernacula (caves, mines, and man-made structures), and it is not known if they use tree roosts or are active during winter in southern portions of the region.

Potential effects of dormant-season burning on species that use forests during winter are likely affected by ambient air temperatures. During colder periods of winter, lasiurine bats and some cavity- and bark-roosting species may retreat to roosts on or near the ground where temperatures are more stable (Boyles and others 2005, Flinn 2009, Hein and others 2008, Mormann and Robbins 2007, Perry and others 2010, Saugey and others 1998). For example, Mormann and Robbins (2007) found red bats switched from tree roosts to leaf litter roosts when temperatures approached freezing, and Flinn (2009) found most roosts of red bats in leaf litter when the maximum daytime temperature was <14 °C (57 °F). Hein and others (2008) found that Seminole bats roosted extensively on or near the forest floor when minimum nightly temperatures were below 4 °C (39 °F). Thus, on days when the previous night's temperature is below approximately 5 °C (41 °F), these bats are expected to be located in roosts more vulnerable to fire. Studies suggest that during winter, red bats roost in litter mostly on south-facing slopes where radiant heating may provide greater temperatures than other areas across a landscape (Flinn 2009, Mormann and Robbins 2007), whereas one study suggested red bats may roost more on north slopes where lower temperatures allow deeper torpor (Saugey and others 1989).

Many bats may take from 30 to 60 minutes to arouse from torpor when ambient temperature is 5 °C (Thomas and others 1990), and red bats roosting in leaf litter may take up to 30 minutes to arouse (Layne 2009). Furthermore, lower ambient air temperatures correspond with longer arousal times. However, when temperatures fall to near or below freezing (<5 °C or <41°F), red bats increase their metabolisms or arouse to prevent freezing (Dunbar and Tomasi 2006). Thus, a bat may not have sufficient time to arouse and flee from approaching flames on cold days (>5 °C), but may be in a lesser state of torpor when temperatures are below freezing. Data on lightning strikes from the Interior Highlands suggest natural fire ignitions peak in July-September, with a smaller peak in March-April (USDA FS 1999). Historical accounts also suggest Native Americans ignited fires mostly in September and November (USDA FS 1999). This led Carter and others (2002) to suggest that bats using forests burned during winter may not be fully adapted to winter burning.

Hein and others (2008) recommended caution when conducting prescribed burns on days when temperatures the previous nights are <4 °C. However, Layne (2009) suggested temperature from sunrise to the onset of the fire had a higher correlation value with red bat recovery times than night-before temperatures. Furthermore, Layne (2009) recommended conducting winter fires on days when temperatures are greater than 10 °C and starting the fire on north-facing slopes to give red bats a chance to passively rewarm and react to approaching fire. Nevertheless, if or how these bats escape injury during winter fire remains unclear, and it is unknown what temperatures these bats experience under the leaf litter.

Reports of red bats exiting leaf litter or located on the ground attempting to flee during dormant-season burns suggests they may arouse during winter fires (Moorman and others 1999, Saugey and others 1989). Furthermore, dormant-season burns are typically ignited in late morning or early afternoon when humidity is lower and temperatures are higher. For example, unpublished data on ignition times and temperatures of days when dormant-season burns were conducted (November-early March) in the Interior Highlands of Arkansas indicated average temperature 2 hours before ignition was 7.7 °C, and average temperature at time of ignition was 12.3 °C

(Table 2). Further, 36 percent of burns were conducted when temperatures 2 hours before ignition measured below 5 °C. Only 13 percent of burns were initially ignited when temperatures were below 5 °C, and many of these burns took hours to complete. Thus, in southern portions of the eastern oak region, dormant-season burns often take place at times when ambient temperatures allow for quicker arousal.

Scesny (2006) reported red bats aroused (22 minutes at 5 °C) when exposed to smoke and the sound of fire, suggesting fire provides cues that cause these bats to arouse. However, other studies suggest individual bats of other species may not arouse when exposed to nontactile stimuli such as sound and light (Speakman and others 1991), and bats in torpor may not be able to perceive sound when temperatures fall below 12 °C (Harrison 1965). Arousal from fire sounds and smoke may be a species-specific response of red bats adapted to roosting close to the ground in fire-prone areas, but additional study is warranted. Nevertheless, possible arousal from fire stimuli, along with the warmer temperatures just prior to many burns may enable red bats to arouse and escape oncoming flames during many dormant-season burns. Head fires typically move in the direction of the wind and may deliver smoke over the area to be burned for substantial periods prior to the arrival of fire. When head fires and slower-moving back fires are used simultaneously, smoke from the head fire may cover the area prior to arrival of the back fire. Thus, substantial smoke may inundate the area prior to arrival of fire, providing cues for bats to arouse. Further study is needed on the interactions of ambient temperature, torpor, arousal times, fire stimuli, and escape behaviors by bats.

Table 2.—Ambient temperatures (°C) for days when 372 controlled burns were conducted during November-March in the Interior Highlands of Arkansas, 2007-2010. Ignition time was estimated at 11 am for all days based on consensus from fire management officers.

Parameter	Min	Max	Average
Low temp previous night (6 am)	-10.6	19.4	2.0±0.34
Temp 2 hours before ignition (9 am)	-8.3	21.7	7.7±0.33
Temp at time of ignition (11 am)	-6.1	25.0	12.3±0.34

Growing Season Fires and Cavity- and Bark-Roosting Species

In forested areas lacking buildings or manmade structures, eight species roost primarily in cavities or crevices of trees during summer (Table 1). In forests, this guild roosts under exfoliating bark, in hollow trees, and in small cavities of damaged or diseased trees (Ford and others 2006; Lacki and Schwierjohann 2001; Perry and Thill 2007c, 2008). Individual species may roost mostly in snags or live trees, and some species may use cavities more than loose bark. For example, evening bats tend to roost more in cavities than under bark (Boyles and Robbins 2006, Menzel and others 2001a, Perry and Thill 2008), whereas Indiana bats tend to roost mostly under exfoliating bark of live trees or snags (Foster and Kurta 1999, Menzel and others 2001b). However, bats may exhibit regional preferences for tree species and roost types based on the composition of available tree species and previous forest disturbances such as disease outbreaks, ice storms, fires, and tornados that create abundant snags or defects in particular tree species. Within this guild, reproductive females typically roost in colonies during summer, whereas adult males and nonreproductive females usually roost alone.

Female maternity colonies in this guild are typically found in relatively tall trees with abundant solar exposure during summer (Brigham and Barclay 1996) where warmer roost temperatures promote fetal and juvenile growth (Speakman and Thomas 2003). Among this guild, roosts in trees for both sexes combined average around 5-10 m above the ground (Lacki and others 2009a, Menzel and others 2002b, Perry and Thill 2008). However, males of some cavity- and bark-roosting species often roost in smaller snags or closer to the ground than females during summer (Broders and Forbes 2004, Kurta 2005, Lacki and Schwierjohann 2001, Perry and Thill 2007c). For example, Perry and Thill (2007c) found 21 percent of roosts of male northern long-eared bats were located in small (<10 cm diameter at breast height [d.b.h.]) midstory trees and snags during summer, whereas <2 percent of female roosts were in these trees. Consequently, for some cavity- and bark-roosting species, males may be more susceptible than females to direct effects of fires during summer because of their closer proximity to the ground and thinner insulation provided by small diameter trees. Males may also enter torpor more frequently than reproductive females (Speakman and Thomas 2003), which could make arousal and escape from fire more difficult for males during cooler periods of summer.

Many of the factors associated with potential injury to foliage-roosting bat species during summer (e.g., height of roosts and fire intensity) may also affect cavity- and bark-roosting species. However, the types of roosts used by cavity- and bark-roosting species may affect vulnerability to injury from fire. Bats roosting under pieces of bark, which are typically closed at the top but open at the bottom, may be more affected by rising heat and smoke, whereas bats in cavities are likely more protected (Guelta and Balbach 2005). Thus, bats roosting in well insulated cavities located relatively high in the trees are unlikely to be subjected to injury. It is unknown how smoke affects bats in these roosts or if different types of roosts reduce or enhance smoke exposure to roosting bats.

Little is known of the direct effects of fire on this guild, and few studies have examined escape behaviors, direct mortality, or potential reductions in survival associated with effects of fire. Dickinson and others (2009) monitored two northern long-eared bats (one male and one female) in roosts during a controlled summer burn. Both bats exited their roosts within 10 minutes of ignition near their roosts and flew in areas where the fire was not occurring. Among four bats they tracked before and after burning, all switched roosts during the fire, but no mortality was observed. Likewise, Rodrigue and others (2001) reported flushing of a *Myotis* bat from an ignited snag during an April controlled burn in West Virginia.

Dormant Season Fires and Cavity- and Bark-Roosting Species

Many cavity- and bark-roosting species hibernate for extended periods during winter in northern portions of the region, often in caves or abandoned mines. Thus, many of these species are not directly vulnerable to dormant-season burns, with the possible exception of smoke intrusions into hibernacula (see below). In southern areas, some species including big brown bats, southeastern bats, Rafinesque big-eared bats, and evening bats, may be active and forage during warmer days of winter or may roost in trees (Barbour and Davis 1969, Boyles and Robbins 2006, Humphrey and Gore 1992). Furthermore, silver-haired bats are long-range migrants that can be found throughout most of the region during winter (Cryan 2003) where they roost in trees in southern portions of the region (Perry and others 2010).

Similar to the foliage-roosting species, some cavity- and bark-roosting bats that remain active during winter may roost on or near the ground during colder periods (<5 °C) of winter. For example, evening bats may use small mammal burrows during colder winter days (Boyles and others 2005), and silver-haired bats may be under tree roots, in rock crevices, or in tree cavities at ground level during colder winter days (Perry and others 2010). Thus, these species may use roosts that are more protected from excessive heat associated with fires than the foliage-roosting species during winter. For bats roosting in the ground, soil temperatures may not exceed 44 °C (111 °F) during prescribed burns at soil depths below 5 cm (Raison and others 1986). However, soil temperatures may exceed 200 °C (392 °F) at a depth of 9.5 cm under heavy fuel loads such as slash piles (Roberts 1965). Furthermore, smoke and CO_2 levels in ground cavities may be benign during controlled burns (O'Brien and others 2006).

Fire Effects on Cavity and Snag Dynamics

Members of the cavity- and bark-roosting guild rely heavily on hollow trees, senescent trees, or snags, and availability of snags or trees with cavities may be directly affected by fire. Snag density and population dynamics are complex and dependent on multiple factors. Age structure of stands and tree species affect snag dynamics. Natural disturbances such as insects, disease, wind and ice storms, lightning, drought, and wildfire all affect creation and destruction of snags. Snag densities are also affected by management prescriptions such as partial harvest, thinning, herbicides, and burning. Because of this complexity, managers often use models to predict snag dynamics in forest stands (McComb and Ohmann 1996, Morrison and Raphael 1993), and snag dynamics is included in the Fires and Fuels Extension of the Forest Vegetation Simulator model (FFE-FVS) (Dixon 2002).

Fire can affect the availability of roosting substrate for cavity- and bark-roosting bats by creating or consuming snags. Although stand-replacing or intense wildfires may create large areas of snags, effects of multiple, low-intensity prescribed burning on snag dynamics may be difficult to predict, especially for forests consisting mostly of fire-adapted species such as oaks. Low-intensity controlled burns with small fuel loads usually kill few or no overstory trees, but typically top-kill small (<5 cm d.b.h.) trees in the understory (personal observation). Low-intensity, ground-level fire may injure larger hardwood trees, creating avenues for pathogens such as fungi to enter and eventually form hollow cavities in otherwise healthy trees (Smith and Sutherland 2006). Fire may scar the base of trees, promoting the growth of basal cavities or hollowing of the bole in hardwoods (Nelson and others 1933, Van Lear and Harlow 2002). Consequently, repeated burning could potentially create forest stands with abundant hollow trees. Trees located near down logs, snags, or slash may be more susceptible to damage or death, and aggregations of these fuels can create clusters of damaged trees or snags (Brose and Van Lear 1999, Smith and Sutherland 2006). However, snags created by fire may not stand as long as snags created by other disturbances because these snags may be weakened at the base by fire (Morrison and Raphael 1993).

In stands with no recent history of fire, prescribed burns may initially create abundant snags by killing small trees and species that are not fire tolerant. Species with thin bark such as beech (*Fagus grandifolia*) and red maple (*Acer rubrum*), may suffer substantial damage or death, whereas oaks and hickories (*Carya* spp.) with thicker bark may suffer little or no damage (Brose and Van Lear 1999, Hare 1965). Furthermore, smaller-diameter trees are at greater risk from mortality due to fire (Hare 1965, McCarthy and Sims 1935). Although burning often creates substantial numbers of small (<15 cm d.b.h.) snags (Horton and Mannan 1988, Morrison and Raphael 1993, Stephens and Moghaddas 2005), effects on larger trees depends greatly on fire intensity, species of trees that are present, fuel loads, and past fire history. In oak savannas, frequent burning may eventually eliminate tree species that are not fire tolerant such as red maple, black cherry (*Prunus serotina*), and serviceberry (*Amelanchier* sp.) (Peterson and Reich 2001). However, long-term fire suppression can allow many relatively fire-intolerant species to grow into size-classes that are resistant to fire (Harmon 1984). Regardless, bats often take advantage of fire-killed snags. For example, Boyles and Aubrey (2006) found that initial burning of forests after years of suppression created abundant snags, resulting in extensive use of these burned areas by evening bats for roosting. Similarly, Johnson and others (2010) found that after burning, male Indiana bats roosted primarily in fire-killed maples.

Fire may consume some standing snags. For example, in forests of ponderosa pine (*Pinus ponderosa*), loss of larger snags (>20 cm d.b.h.) was 43 percent following introduction of fire and 21 percent following second fires, but net loss (including new snags created by fire) was only 12 and 3 percent, respectively (Bagne and others 2008). Furthermore, Holden and others (2006) found fewer large (>47.5 cm d.b.h.) snags in ponderosa forests burned twice compared to those burned only once, but found no difference between areas burned two or three times. Thus, initial burns may create abundant snags and second burns may consume some of these, but further burns may not eliminate substantial numbers of large snags. However, comparable studies from eastern oak forests are not available.

Season of burning and topography also affect potential damage or death of overstory trees in hardwood stands. Winter burns tend to cause the least overstory damage because of cooler ambient temperatures and the dormant state of trees (Brose and Van Lear 1999). Spring burns may cause the greatest damage to overstory trees because of higher ambient temperatures, sunlight on boles, and fully hydrated vascular tissues that may reach lethal temperatures when burned (Brose and Van Lear 1999). Summer burns tend to be less damaging than spring burns, likely because of bole shading and lower intensity of fires (Brose and Van Lear 1999). Dry, upland sites on ridge tops and steep slopes tend to burn more intensely, and trees in these locations may be more susceptible to damage during fires.

Fire and Forest Structure for Roosting

Aside from creating snags, periodic prescribed burning may reduce the number of woody shrubs, understory trees, and midstory trees (10-25 cm d.b.h.) in the short term (Blake and Schuette 2000, Hutchinson and others 2005). Longer-term applications of prescribed fire may reduce stand density (Hutchinson and others 2005, Peterson and Reich 2001) and complexity (clutter). Repeated low-intensity fire reduces clutter in the midstory and understory and creates more open forests, which may provide more favorable roosting (and foraging) conditions for many bat species, especially females during the reproductive season. Studies often find roost trees (mostly female) further from other overstory trees (Betts 1998, Brigham and others 1997b) and less canopy cover at roost sites compared to random locations (Kalcounis-Rüppell and others 2005). Canopy gaps created by fire may provide favorable roosting sites with greater solar exposure during summer for maternity colonies of some cavity- and bark-roosting species (Johnson and others 2009). Furthermore, maternity roosts may be located in areas with few midstory trees or relatively lower tree densities, which may provide both greater solar exposure and more open areas immediately around and below roosts that would otherwise impede inexperienced juvenile flyers (Perry and Thill 2007c). Thus, burned areas may have lower tree densities, less structural clutter, more open canopy, and greater numbers of snags, which may provide favorable roosting areas for many species.

Studies often find bats favor burned areas for roosting. For example, Perry and others (2007b) found five of six species, including red bats, Seminole bats, northern long-eared bats, big brown bats, and evening bats roosted disproportionally in stands that were thinned and burned 1-4 years prior but that still retained large overstory trees. Boyles and Aubrey (2006) found evening bats used burned forest exclusively for roosting. Furthermore, Johnson and others (2009) found that for northern long-eared bats, roost-switching frequency, duration at roosts, and distance between successive roosts were similar between burned and unburned forests.

Caves and Mines

Bats of 13 species may use caves during all or part of the year. Three bats (gray bat, Ozark big-eared bat, and Virginia big-eared bat) are year-round cave obligates (Table 1). Little is known of the effects of fire on adjacent cave and mine habitats used by bats, but these effects may be especially important in karst areas of the eastern oak region. Fire could alter vegetation surrounding entrances, which could potentially modify airflow (Carter and others 2002, Richter and others 1993). Smoke and noxious gases could enter caves, depending on air-flow characteristics of individual caves or mines and weather conditions such as temperature (Carter and others 2002, Tuttle and Stevenson 1977). Fire may not cause levels of gases high enough to be toxic to bats in caves or mines, but gases could potentially cause arousals during hibernation (Dickinson and others 2009). Caviness (2003) noted smoke intrusion into hibernacula during winter burning in Missouri, but no arousal of hibernating bats was observed. No quantitative studies have examined smoke intrusions into caves and mines or smoke effects on hibernating cave bats, but this is an area that needs to be studied.

FORAGING AND FIRE

Fire and Forest Structure for Foraging

Various factors can affect bat use, activity, and foraging within forests stands. Although studies often find the greatest levels of bat activity in forested riparian areas (Carter 2006, Ford and others 2005, Grindal and others 1999, Zimmerman and Glanz 2000), bat activity and foraging may be greatly influenced by forest clutter. Studies throughout North America suggest that most bats avoid highly cluttered areas and prefer to forage and travel in areas with less clutter (Brigham and others 1997a, Erickson and West 2003, Hayes and Loeb 2007, Humes and others 1999). Bats are often more active in early- and late-seral stages which are usually less cluttered than in intermediate forest stages (e.g., Burford and Lacki 1995a, Erickson and West 2003, Humes and others 1999, Loeb and O'Keefe 2006, Menzel and others 2005). Thinning may reduce clutter and lead to increased bat activity (Erickson and West 2003, Lacki and others 2007), although some studies suggest no response by bats to thinning (Tibbels and Kurta 2003).

Responses to clutter differ among bat species. Differences in bat size (mass), bat morphology, and the echolocation frequencies used among species are believed to make some species more adapted to foraging in cluttered habitats, whereas others are more adapted to foraging in open habitats (Aldridge and Rautenbach 1987, Norberg and Rayner 1987). Species such as big brown bats likely forage more in open forests (Ford and others 2005, Ford and others 2006), whereas northern long-eared bats and Indiana bats may readily utilize cluttered forests (Broders and others 2004, Ford and others 2005, Owen and others 2003, Schirmacher and others 2007), and red bats may use both cluttered and uncluttered habitats (Carter and others 2004, Menzel and others 2005). However, associations between individual species and levels of tolerable clutter are not concrete, and further study on foraging-habitat associations is needed. Total bat activity may be greater above the forest canopy than below, and some species, such as hoary bats and big brown bats, may be more active above the forest canopy (Menzel and others 2005). Reproductive condition may also affect bat foraging; less-maneuverable pregnant females may be less able to forage in cluttered habitats than nonpregnant bats (Aldridge and Brigham 1988).

In general, within forest stands, fire reduces clutter that theoretically would provide favorable foraging habitat for some species. Similar to thinning, fire may reduce tree densities and create more open forest canopies. However, thinning and midstory removal may reduce structural clutter substantially more than burning alone, and bats may respond to thinning more strongly than simply burning. For example, in the Piedmont of South Carolina, Loeb and Waldrop (2008) found overall bat activity greater in thinned stands than unthinned controls, whereas activity in thinned and burned stands was intermediate. Fire may also kill small groups of trees, creating small gaps in the forest canopy. Small openings, such as tree gaps and group openings, often have higher activity than the surrounding forest (Menzel and others 2002a, Tibbels and Kurta 2003). Quantitative studies have found bat activity response to burned forests is generally favorable or not discernable. For example, following spring burns, Lacki and others (2009b) found home-range sizes and core areas of northern long-eared bats during late spring to summer were unaffected by burn-induced changes in habitat, but bats foraged more in burned habitats than unburned areas. Alternatively, Loeb and Waldrop (2008) found no difference in bat activity levels in burned and unburned habitats.

Fire and Insect Abundance

All species of bats found in the eastern oak region are voracious insectivores. An individual can consume from 40 to 100 percent of its body mass in insects nightly (Kunz and others 1995). Most bats in the region consume insects in flight, although some species such as big brown bats and northern long-eared bats may glean insects off foliage (Faure and others 1993, Stamper and others 2008). Consequently, abundance of nocturnal flying (and to a lesser extent, foliar) insects may have a direct effect on fitness of individual bats, but short- and long-term effects of prescribed burning on abundance of nocturnal flying insects remain unclear.

Arthropod communities consist of numerous orders, families, and species that vary in natural history and include detritivores, herbivores, and predators. Therefore, responses of the overall insect community to fire are complex. Among individual insect taxa, fire and fire frequency may affect species groups differently, with some families increasing in burned areas and other families decreasing in richness or abundance depending on season and intensity of burns. Insect communities may differ between burned and unburned areas (Swengel 2001). Some flying insects (at least 40 species, mostly beetles) are attracted to fires, and the subgenus *Melanophila* use infrared sensors to locate fires where they lay eggs on burnt woody debris immediately after cooling (Hart 1998).

Although many studies have examined effects of burning on insects, these studies are often not directly applicable to the insects available to bats in eastern oak forests subjected to burning. Most previous studies in North America have focused on effects of burning on insect abundance and diversity in grasslands or agricultural lands. Previous studies often include taxa that are not readily available to bats or combine taxa such as ground-dwelling and flying species. Furthermore, sampling methods used to determine effects of burning may affect observed responses. For example, pitfalls and litter sampling techniques often sample ground-dwelling insects that are not part of the food base for bats. Also, many studies of volant insect abundances examined diurnal insect abundance, and it is unclear if patterns of abundance for nocturnal insects mirror those of diurnal insects. Relatively few studies have examined effects of burning in a way that would provide insight into the specific groups of insects that bats utilize as food (i.e., nocturnal flying insects).

Aside from the limitations of previous studies stated above, richness and abundance of herbivorous flying and foliar insects has long been associated with plant species richness (Hartley and others 2007, Knopps and others 1999, Murdoch and others 1972) because many insect species forage on the foliage, pollen, or nectar of specific plant species. Fires may produce more lush plant growth, and postburn vegetation may be attractive to recolonizing insects (Swengel 2001), although size and heterogeneity of burns likely affect the ability of flying insects to recolonize burned areas. In eastern oak forests, the herbaceous layer harbors the majority of plant species richness, and burning typically increases community diversity and abundance of herbaceous plants (Hutchinson 2006). Consequently, one would expect short-term reductions in insects due to mortality from fire and a temporary reduction in understory foliage, followed by subsequent increases in flying and foliar insects from enhanced abundance and diversity of herbaceous plants later.

Results of studies examining burning and abundance of flying insects often give conflicting results. In grasslands, studies that collected flying insects suggest that recently burned sites produce more flying insects than nonburned sites (Hansen 1986, Nagel 1973), although burning may reduce abundance of ground-dwelling insects (Buffington 1967, Bulan and Barrett 1971, Seastedt 1984, Warren and others 1987). Studies suggest that fires may cause a short-term decrease of 95 percent in soil macroarthropods immediately after fire (Paquin and Coderre 1997), which could ultimately affect volant species because many volant species have a larval or pupae stage that is resident in litter or soil. Nevertheless, studies in other regions suggest many flying insects are resilient to burning and recover quickly because of their mobility, whereas ground- and litter-dwelling insects have low resilience to fire (Lamotte 1975, Moretti and others 2006). In California chaparral, insect abundance peaked the first year after fire, likely because of an influx of generalist species that took advantage of the lush vegetation in the postburn area, but abundance declined the second and third years after burning (Force 1981).

In oak-dominated forests of the eastern United States, thinning or other reductions in basal area likely has more of an effect on herbaceous plant growth than burning alone. In eastern oak forests, combinations of thinning and burning along with mechanical understory shrub control may produce substantially more flying insects than forest stands that are burned but not thinned or in unburned stands (Campbell and others 2007). In oak savannas, abundance of flying insects may be low the year of fire but quickly rebounds in subsequent years after burning (Siemann and others 1997). However, Lacki and others (2009b) found a 34 percent increase in nocturnal insects used by bats in burned areas during the first year following spring burns in Kentucky. Thus, fire in eastern oak forests may or may not cause a short-term decrease in abundance of flying insects, but may ultimately increase overall abundance.

Fires may also have indirect effects on insect production. For example, in riparian areas, fires may increase nutrient delivery into streams and reduce canopy cover, which may increase water temperatures, all leading to increased productivity (Minshall and others 1997, Spencer and Hauer 1991). Increases in emerging insects may result from this increased productivity (Malison and Baxter 2010, Minshall 2003), providing more food resources for bats. Malison and Baxter (2010) found streams in high severity burned areas had substantially greater insect emergence than streams in low severity burns or unburned areas, and bat activity in severe burn areas was substantially greater.

Moths are one of the most important insect groups in the diets of many eastern bats, and some bats are moth specialists, including the big-eared bats (*Corynorhinus* spp.) (Burford and Lacki 1995b, Hurst and Lacki 1997, Leslie and Clark 2002). Although most larval caterpillars of moths feed on vegetation (many are agricultural pests), adults either use nectar sources such as herbaceous flowers or do not feed as adults. Consequently, abundant and diverse herbaceous vegetation likely produces more food sources for those adults that feed. Restored woodlands subjected to periodic burning may produce substantially more nectar sources than mature unmanaged forests (Rudolph and others 2006). In forests, caterpillars of most moth species feed on woody plants such as oaks (Summerville and Crist 2002). Furthermore, early seral clearcut stands may be dominated by moth species whose caterpillars feed on tree species such as *Prunus* spp. and herbaceous vegetation, whereas mature forests may be dominated by species whose caterpillars feed on oaks, hickories, acorns, fungi, and lichens (Summerville and Crist 2002). Therefore, abundance and diversity of woody plants may be more important to moths than abundance and diversity of herbaceous vegetation in the understory.

Studies of moths in the eastern oak region have compared abundance and diversity of moths among forest age classes and pasture monocultures (Burford and others 1999, Dodd and others 2008, Summerville and Crist 2002), but relatively few studies have examined effects of burning on moth abundance. Lacki and others (2009b) found a 22 percent increase in moth abundance the first year after burning in Kentucky, although the difference was not significant. In frequently burned pine woodlands of Arkansas, Thill and others (2004) found moth abundance was generally greater in forests managed using frequent fire compared to unburned controls, except for the first couple of months immediately following the burn.

FOREST DIVERSITY AND BATS

During landscape-level burns, differences in forest density and topography often result in a mosaic of burned and unburned areas that provide different levels of clutter and density of snags. For example, riparian areas (greenbelts, streamside management zones, or riparian zones) are often not subject to harvest or thinning and often burn less intensely than the surrounding forest due to greater shading and moister litter conditions. These buffers often provide greater densities of trees, more cluttered habitats, and more small trees and small snags than the surrounding forest landscape. Individual bat species may use these areas more or less than then their availability. For example, Perry and others (2007b) found that within thinned and burned stands, less than 2 percent of Seminole bat roosts were in unthinned greenbelts, whereas most (90 percent) roosts of tri-colored bats were in greenbelts. Furthermore, Loeb and Waldrop (2008) found activity of big brown bats greater in thinned stands than unthinned controls, whereas activity of tri-colored bats did not differ among treatments. Thus, heterogeneous habitats created by various levels of thinning and burning intensity may provide a range of roosting and foraging habitats for a varied bat community.

CONCLUSIONS

Fire is increasingly being used in oak forests to promote oak regeneration, improve wildlife habitat, and reduce hazardous fuel loads. Although recent research has begun to shed light on the relationships among fire, bats, and bat habitat, these interactions are not yet understood. These interactions offer substantial opportunities for expanded research. Studies suggest burning may have positive, negative, or no effect on various aspects of bat ecology, but effects may vary among bat species, time of the year, fire frequency, ambient temperatures, and intensity of burns. In general, burning appears to improve habitat by creating snags and opening up habitats for foraging and roosting. Fire may also help create heterogeneous landscapes that provide a variety of habitats for multiple species. Abundance of bat food (insects) may also respond positively to fire. Direct effects of fire on bats during the burning process are still uncertain but may vary considerably due to timing and intensity of burns and the species of bat considered. Nevertheless, bats likely adapted to this disturbance over the millennia in areas such as the eastern oak region where fire has played a central role in the formation and maintenance of these forests.

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RESPONSES OF TIMBER RATTLESNAKES TO FIRE: LESSONS FROM TWO PRESCRIBED BURNS

Steven J. Beaupre and Lara E. Douglas

Professor (SJB), University of Arkansas, Department of Biological Sciences, Fayetteville, AR, 72701; Natural Resource Specialist (LED), Bureau of Land Management. SJB is corresponding author: to contact, call 479-575-7561 or email at sbeaupre@uark.edu.

Abstract.—Timber rattlesnakes (*Crotalus horridus*) are excellent model organisms for understanding the effects of large scale habitat manipulations because of their low-energy lifestyle, rapid response to changes in resource environment, uniform diet (small mammals), and simple behaviors. We present two case studies that illustrate interactions between timber rattlesnakes and fire in a single large population. Case 1 describes the decimation and subsequent 11 year recovery of a timber rattlesnake subpopulation associated with a fire during a particularly vulnerable time of year. In Case 2, four control plots, three cut (thinned) plots, three burned plots, and three plots that were both cut and burned were studied. Our primary goals were to monitor responses of the food chain to the above four treatments and to assess timber rattlesnake responses as potential indicators for the relative success of manipulations. Although plant communities did not initially differ among treatment plots, manipulated sites experienced increases in early-successional annual vegetation after thinning and burning. Biannual live-trapping sessions indicated an increase in abundance of principal prey species after manipulations, although this increase was not uniform among treatments. Timber rattlesnakes that utilized manipulated sites exhibited enhanced growth and body condition relative to snakes that foraged solely in control areas. Snake physiological responses were more rapid and well-defined than measurable small mammal population responses suggesting that these top predators may potentially serve a role as indicator species for restoration ecology. Our case studies illustrate both direct and indirect effects, as well as dramatically divergent outcomes resulting from minor changes in the timing of fire application.

BACKGROUND

Although the practices of thinning and fire application have become important tools for forest habitat management, relatively little is known about the responses of large-bodied ground-dwelling reptiles to large-scale habitat manipulations (however see Reinert and others 2011b). Most studies of the effect of fire on reptiles in eastern oak-dominated ecosystems have found little to no effect on diversity, species composition, and abundance (Ford and others 1999, Greenberg and Waldrop 2008, Renken 2006). Frequently, in some treatments (typically thinned and burned plots) some reptiles show increases in abundance (Kilpatrick and others 2004, Matthews and others 2010), suggesting beneficial effects of fire. The few studies that have examined seasonal burn timing (Keyser and others 2004, Renken 2006) suggest that reptile communities may be resilient to both dormant and growing season burns (for a counter-example, see Griffiths and Christian 1996).

Most of the afore-mentioned studies use similar methods. Specifically, reptile populations are sampled using one or more methods, including drift fences with funnel traps or pitfall traps. Both funnel and pitfall traps are subject to sampling biases since not all reptiles can be equally caught using these methods (Crosswhite and others 1999, Enge 2001, Greenberg and others 1994, Jenkins and others 2003). In eastern oak-dominated ecosystems, pitfall traps tend to capture small mobile species such as fence lizards (*Sceloporus undulatus*), members of the five-lined skink group (*Plestiodon* spp.), ground skinks (*Scincella lateralis*), Anoles (*Anolis carolinensis*), small snakes such as garter snakes (*Thamnophis* spp.), and fossorial species such as worm snakes (*Carphophis* sp.) and ring-necked snakes (*Diadophis* spp.). Indeed, members of this short list tend to be the primary species upon which assessments of the effects of fire are made (but see Steen and others 2010). Funnel or box traps seem more effective at sampling larger snakes (Burgdorf and others 2005, Crosswhite and others 1999, Enge 2001, Greenberg and others 1994, Jenkins and others 2003, Sutton and others 2010). We note that many larger reptiles (e.g., box turtles, larger snakes) or those that are less mobile (e.g., ambush foragers such as the timber rattlesnake; Beaupre and Montgomery 2007) typically occur in low density and are relatively poorly sampled by funnel traps and pitfalls, and thus, their responses to fire may be under-represented in the literature. Whereas we acknowledge the importance of a variety of trapping methods for quantifying community dynamics, significant information regarding response to environmental perturbation can be obtained through radio telemetry in appropriate target species (Beaupre 2008, Durbian 2006, Reinert and others 2011b).

Long-lived vertebrates with slow life-histories (e.g., long life spans, low growth rates, late age at maturity, infrequent reproduction with small litter size) such as the timber rattlesnake (*Crotalus horridus*) that forage by ambush in the leaf litter (Brown 1993), may be especially susceptible to fire, depending on their evolutionary interaction with fire. Timber rattlesnakes are of special conservation concern throughout much of their geographic range as documented by their legally protected status throughout the northeast (Brown 1993). Thus, knowledge of their response to logging and fire may be of great value to habitat managers. It has been suggested that the behavior and physiology of top predators may be effective monitoring tools for assessing the outcomes of habitat manipulations (Beaupre 2002, Beaupre and Douglas 2009, DeAngelis and others 1991, Dunham and others 1989). With training, rattlesnakes may be easy to find (depending on species and locality), easy to measure and manipulate, and may provide a time- and cost-effective monitoring approach. Herein, we describe and discuss two case studies of the response of timber rattlesnakes to the application of fire in a habitat management context. Both cases illustrate potential benefits and costs associated with application of fire.

STUDY SITES

We conducted studies on the McIlroy Madison County Wildlife Management Area (MMCWMA) in Madison County, northwest Arkansas. The MMCWMA is a public holding with a primary function as a public hunting ground. The MMCWMA and adjacent lands including Bear Hollow Natural Area (BNHA) and Ozark Natural Science Center (ONSC) comprise over 6,000 ha managed for multiple uses. Topography can be rugged with permanent and intermittent streams, steep rock walls along ridgelines, periodically maintained food plots, and artificial ponds for game management. We have been continuously engaged in studies of the ecology and physiology of timber rattlesnakes at these sites since the fall of 1995 (e.g., Beaupre 2002, 2005, 2008; Beaupre and Montgomery 2007; Beaupre and Zaidan 2001; Browning and others 2005; Wills and Beaupre 2000).

In general, forests at the MMCWMA are typical of disturbed Ozark hardwood forests and could be described as closed-canopy, dense, even-aged stands of mixed hardwoods and pine resulting from long-term fire suppression (Spetich 2004). Aggressive tree harvesting in the late 19th century followed by roughly 100 years of fire suppression (Guyette and others 2006, Smith and others 2004, Stambaugh and Guyette 2006) dramatically altered species dynamics, causing an increase in recruitment of shade tolerant species (Burns and Honkala 1990) while oaks and hickories, which once dominated these forests, had low establishment rates (Spetich 2004). The system became an even-aged, closed-canopy forest with relatively little herbaceous understory growth (Spetich 2004). Canopy closure and decreased herbaceous growth has reduced ground-level productivity resulting in mast-crop (primarily acorns) dependence of wildlife (Fralish 2004, Spetich 2004).

The timber rattlesnake is a low-energy adapted ambush forager that feeds on small mammals, primarily including squirrels, chipmunks, and deer mice (Clark 2002, Montgomery 2005, Reinert and others 2011a, Wittenberg 2009). Previous studies have suggested that annual variation in acorn mast crop affects small mammal densities, which in turn rapidly affect feeding rates, growth dynamics, and body condition of timber rattlesnakes (Beaupre 2008). The population dynamics of timber rattlesnakes at MMCWMA can be described as “boom and bust” with periods of extended starvation punctuated by bursts of growth and reproduction associated with resource availability (Beaupre 2008). Their simple and narrow diets leave little ambiguity regarding the structure of their food chain. Studies of timber rattlesnakes have been facilitated by highly developed radio telemetry techniques (Reinert 1992). Their relatively small home ranges (as compared with predatory mammals or birds) and their tolerance to close approach by humans allows them to be repeatedly tracked on foot and relocated with high precision (Beaupre 2008, Reinert 1992). The extreme sensitivity and rapid physiological responses of timber rattlesnakes to variation in the food resource environment coupled with ease of measurement and estimation of body condition potentially make them excellent bioindicators for assessing changes in small mammal populations as a result of habitat management practices that affect ground level seed production. Similarly, timber rattlesnake populations are highly sensitive to the effects of short-term mortality events (Brown 1993, Sealy 2002) and disturbances that undermine their food chain (Beaupre 2008).

CASE 1: DECIMATION AND RECOVERY AT SITE K

Background

Site K is a timber rattlesnake den complex (defined as an association of several independent hibernacula located on the same rocky outcrop) centrally located in our study area. The complex consists of at least five unique cracks that are occupied by one or more hibernating snakes distributed north to south along a 450 m west-facing ridgeline. Timber rattlesnakes typically emerge from hibernation in early April and rapidly disperse from the immediate hibernation area to take shelter under transitional rocks, logs, or leaf litter. In the weeks that follow emergence the snakes usually remain concealed, using cover to insulate them from temperature extremes while physiologically acclimatizing to warmer temperatures outside the den crevice. Prior to emergence, while in the den, snakes are likely insulated from damage due to surface fires. During a posthibernation fire, however, snakes are directly exposed to heat, oxygen deprivation, and desiccation, as they are physically located in the flammable substrate.

Since the inception of our studies in 1995, sections of the MMCWMA have been periodically burned either as part of a formal management plan, accidentally, or as a result of arson. Fire management plans at MMCWMA have historically been directed at fuel reduction, but more recently have been designed for wildlife habitat improvement (see Case 2). In spring of 1999 (between April 6 and April 15), a management-related fire swept through Site K involving all known hibernacula and transitional habitats associated with snakes that used the site. The fire encompassed approximately 10 ha and was of sufficient intensity to completely consume ground cover (leaf litter) and structure (logs) in the immediate vicinity of Site K. The fire occurred after emergence and during the period when snakes had taken refuge in transitional habitat, but before snakes had moved to summer foraging ranges.

Methods

Site K has been visually surveyed during spring emergence (March 20-April 20, depending on weather) in every year since spring 1996. Although some variation in search effort among years is unavoidable, the site has been consistently surveyed (two to four visits per year during peak emergence) by the primary author. All captured snakes were transported to the laboratory at the University of Arkansas where they were individually marked (by PIT tag), sexed (by caudal probe), weighed (±0.01 g), and measured (snout-vent length [SVL], head length [HL], head width [HW], and tail length [TL]) in a squeeze box (Beaupre 2008, Quinn and Jones 1974). After processing, all snakes were released at their point of capture.

Results and Discussion

In 4 years of spring sampling (1996-1999) prior to the fire, 20 individual timber rattlesnakes were captured from Site K, processed, and released. The fire occurred between April 6 and April 15, 1999, immediately postemergence when most snakes had entered vulnerable transition sites. Two snakes had been captured at the site in 1999 on April 6 and were safe in the laboratory when the fire struck. These two snakes were released on Site K after the fire, but no additional snakes were captured in 1999 at the site after the fire. In the years immediately following the fire, no snakes were found in spring surveys (Fig. 1). Snakes were not captured again until 2002 (n=2) with relative abundance increasing in 2005-2011 (Fig. 1). In the 12 years since the fire, 16 individual timber rattlesnakes have been captured at the site. The fire had a pronounced effect on the body size distribution of snakes that resided at the site (Fig. 2). Prior to the fire, the SVL distribution was relatively evenly distributed with some snakes present in all size classes, including several very large adult males and a large number of reproductively mature females near 75 cm SVL (Fig. 2). Furthermore, active reproduction was known to be ongoing, as we observed pregnant females there in 1998. In the years after the fire, there was an apparent complete loss of snakes in the two largest size classes and a shift of the most frequent size class from 75 cm to the bin centered on 55 cm SVL (Fig. 2). This size class is typical of snakes roughly 3 to 5 years old that have not yet reached the age and size of first reproduction. Because timber rattlesnakes exhibit high den site fidelity (Agugliaro 2011, Browning and others 2005) and highly repeatable home ranges (Beaupre, per obs.), we interpret this result as a near total loss of snakes at the site due to the effects of fire. The site was then slowly repopulated by dispersing young (Cobb and others 2005) over the 12 postfire years from adjacent unaffected sites.

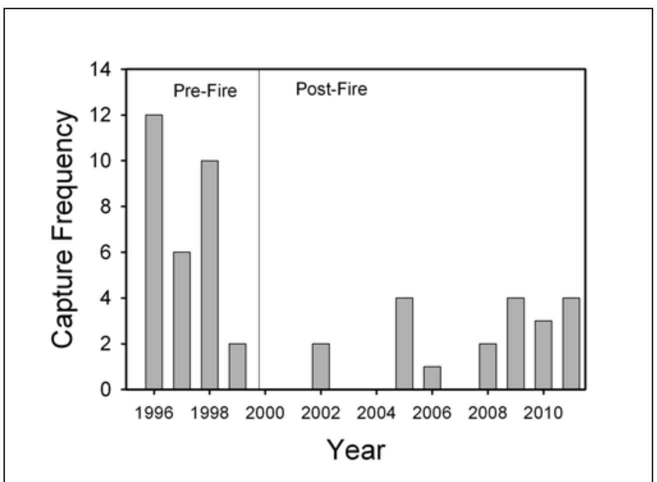

Figure 1.—Timber rattlesnake capture frequencies at Site K before (pre-spring 1999) and after (post-spring 1999) a destructive fire swept through the den area in the days just after spring emergence. Note: the two snakes captured in 1999 were captured prior to the fire.

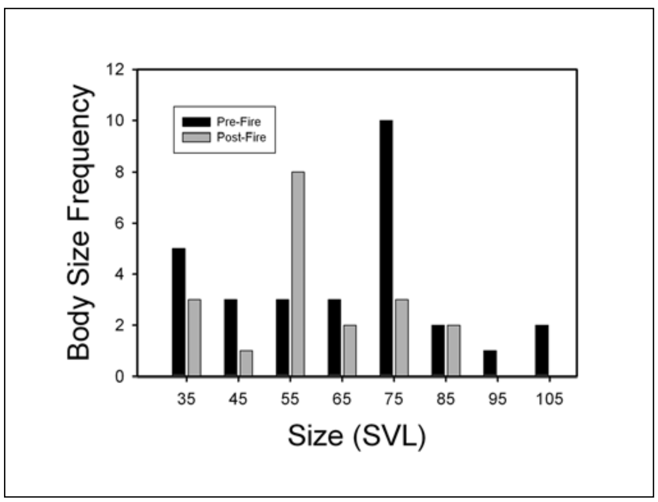

Figure 2.—Site K timber rattlesnake body size (snout-vent length: SVL in cm) distribution before fire (black bars) and after fire (gray bars). Each bin represents a 10 cm range centered on the tick label value.

Capture records before and after burns and from adjacent sites have revealed that 2 of the 16 snakes captured in later years (one male, one female) were present at Site K prior to the fire. We interpret these individuals to be persistent survivors of the fire. No other individuals present prefire at Site K were captured there or elsewhere in the study site after the fire. Notably, one of the two recaptured individuals (the male) was safely in the lab during the fire. A third individual of the 16 was a known migrant, captured and marked as a neonate on September 12, 1999 at an adjacent site (ca. 1150 m distant) and next appeared on Site K on April 9, 2005 as a small (79.6 cm SVL, 336.94 g) adult male. The remaining 13 snakes were marked as new captures on Site K. One of these new individuals was large enough to be considered an adult, the remaining 12 were sub-adults, all having SVL of less than 60 cm. These young snakes were unlikely to have been present at the site prior to 2009 and were probably either born at Site K or migrated from one of roughly 40 nearby areas (Browning and others 2005) in the years following the fire. Radio telemetry observations on neonate timber rattlesnakes in Tennessee support the contention that at least some neonates disperse to nonmaternal hibernacula (Cobb and others 2005). The persistent postfire paucity of adult (SVL = 65 cm) reproductive females suggests that most if not all of these young snakes have dispersed to Site K from adjacent den areas.

The preceding data led us to the following interpretation. A healthy population of rattlesnakes hibernating at Site K was decimated (only 2 of 20 known snakes survived) by a fire that struck after spring emergence when the snakes were particularly vulnerable. In the 12 years postfire, the site has been repopulated mostly by juvenile migrants from adjacent areas of higher timber rattlesnake density. We note that the site continues to recover and predict that in the coming years as sub-adults mature, reproduction will resume and a more typical size distribution will be reestablished. However, due to the long intervals between reproductive events and late female age at maturity in this species (Brown 1993), full recovery may take many more years. With respect to timber rattlesnakes, the severity of this fire was due primarily to the posthibernation timing of its application. It is also fortunate that the size of the fire was small, leaving surrounding hibernacula relatively undisturbed to serve as source populations for juvenile migrants.

CASE 2: CAN FIRE ENHANCE TIMBER RATTLESNAKE HABITAT?

Background

In 2004, we were approached by the Arkansas Game and Fish Commission (AGFC) and were informed that sections of our study site would be selectively thinned of timber and/or burned in an effort to enhance habitat quality for wildlife. Subsequent planning led to establishment of a detailed experimental design (described below). The AGFC sought to enhance habitat for game species, of which white-tailed deer (*Odocoileus virginianus*) and turkey (*Meleagris gallopavo*) were of primary interest. Thinning and burning were applied with the ultimate goal of opening habitat to approximate the historically present oak savanna or shortleaf pine forest system (Dey and others 2004, Guyette and others 2006).

Like white-tailed deer and turkey, small mammal populations (e.g., squirrels, chipmunks, and other rodents) are also dependent on acorn mast crop. These small mammals comprise over 90 percent of the diet of timber rattlesnakes. We hypothesized that small mammals would respond to increases in ground level food production much as deer and turkey would, and that rattlesnakes would in turn respond to increasing prey availability. Because of the ease of capture and measurement of timber rattlesnakes, we also hypothesized that they might be effective organisms to monitor changes in food productivity at the ground level. Thus, we measured changes in vegetation structure associated with forest manipulations, changes in small mammal populations, and changes in growth and body condition (both indicative of changes in resource availability) of timber rattlesnakes. Herein, we review the major findings of our study. A more detailed account can be found in Douglas (2010), and forthcoming publications.

Methods

In 2005, areas on ridge tops intended for timber harvest were identified by AGFC managers, and trees slated for removal were marked. Six plots ranging in size from 4 to 26 ha were chosen for timber harvest. Low-impact logging occurred during the summer of 2007 and was accomplished by two to three individuals on foot with chainsaws. Logs were skidded to existing forest roads and removed without major habitat disruption. Fire was applied on March 12, 2008, prior to rattlesnake spring emergence. The net result of these management activities was 13 experimental plots, including 3 plots that were only harvested (cut-only), 3 plots that were both harvested and burned (cut-burn), 3 plots that were burned but not harvested (burn only), and 4 plots (control) that were not manipulated. Our experimental design approximated a BACI (before, after, control impact) design (Williams and others 2001). However, manipulated plots were located on ridge tops only and were chosen based on accessibility and suitability for harvest and the ability to control application of fire. Thus, plots were not chosen at random, a normal assumption of BACI design, weakening inferences about the effects of manipulations by potential interactions related to the geographic proximity of certain site types (Bennett and Adams 2004). All experimental and control plots were located within known home ranges of several timber rattlesnakes, and plots were sized and located such that several snakes could occupy a plot simultaneously, but all snakes could easily avoid all manipulated plots.

Vegetation

We monitored plant diversity and community composition at all sites beginning in summer 2005 and continued throughout the study (Douglas 2010). Circular quadrat surveys (Cox 1980, Lindsey and others 1958, Patterson and James 2009) were employed at three spatial scales (1 m^2, 15 m^2, and 0.04 ha) three times per year to examine herbaceous plants, understory saplings and shrubs, and mature trees. Species were identified on site using field guides (Hunter 2004, Little 1980). Problematic identifications were confirmed with the help of ONSC field staff, Professor Doug James at the University of Arkansas, or University of Arkansas Herbarium staff. Data were analyzed by a variety of techniques, including MANOVA and principal components analysis.

Small Mammals

Small mammals at all 13 plots were sampled biannually (spring, late summer) from 2005 to 2009 using Sherman live traps (3” x 3.5” x 10” LFAHD Folding Trap and 3310A Non Folding Trap, H.B. Sherman Traps, Tallahassee, FL), with populations assumed to be open between trapping periods. Each trapping session lasted five nights (after five nights of prebaiting), a brief period through which the population was treated as closed. Fifty traps were set per site per night, with 4 of the 13 sites sampled per night. Trapped animals were marked with a unique ear tag (Self Piercing Fish Tag, National Band and Tag Company, Newport, KY) and released, and marks were used to identify recaptured individuals. Trap data were analyzed by Lincoln-Peterson estimator and by trap success rates which can be adjusted for trap disturbance and, in this case, provided qualitatively similar results to more formal population density estimators (Douglas 2010).

Rattlesnake Physiological Responses

Snakes were captured between 2006 and 2009 during routine den and gestation site surveys and incidentally during random walks through study areas. We surgically implanted radio transmitters (Holohil SI-2T, Holohil Systems, Ltd., Carp, Ontario) in the coelomic cavity of adult snakes (minimum 300 g) under anesthetic (Beaupre 2008, Reinert and Cundall 1982, Wills and Beaupre 2000). Depending on year, 20 to 30 snakes were tracked approximately three times weekly using portable receivers (Wildlife Materials TRX 1000s, Murphysboro, IL or Communication Specialists Inc. R1000 receiver, Orange, CA) and hand-held antennae (Yagi three-element directional antenna). Snake locations were recorded using a Garmin GPS III Plus (Olathe, KS). At least twice per active season, snakes were captured and brought to the lab for morphometric measurements (SVL, HL, HW, TL, and body mass) as described above (Case

1). All measurements were made by the first author to minimize observer bias. Morphometric measurements were used to estimate growth rates in SVL (cm/year) and body condition (residuals from a nonlinear mass-length regression; a measure of fatness relative to length) (Beaupre and Douglas 2009). Snakes that spent more than a week of the active season foraging in manipulated areas were classified as manipulated (man). Snakes that spent their entire active season away from manipulated areas were classified as controls (con). Thus, statistical comparisons (repeated measures analysis of variance) were possible among treatments both before and after the habitat manipulations (Douglas 2010).

Results and Discussion

Major shifts in vegetation occurred from 2005 to 2009 in association with both experimental manipulations and natural events that affected canopy density and dynamics. The most significant component of habitat change was associated with increases in sun grasses, early annuals, early perennials, shade perennials, sassafras, blackberry, and sedges. Most of these groups are associated with invasions of forest openings and are seed-producing. In addition to planned forest treatments, all sites including control sites were also affected by natural events between 2005 and 2009. In 2005, all sites clustered closely to the grand mean of pre-manipulation sites, suggesting that sites were quite similar prior to manipulations. However, each plot type exhibited a different response to manipulation with burn and cut-burn sites exhibiting increases in seed-producing sun-tolerant species, and control and cut sites exhibiting increases in tree recruitment (Douglas 2010). Application of logging and fire caused significant shifts in vegetation communities observed at our 13 sites. Paradoxically, control sites exhibited large shifts in vegetation, although in a slightly different multivariate direction than burned sites. We attribute this shift in control sites to a severe ice storm in February of 2009 that caused a significant canopy opening event, subsequently changing incident sunlight on the ground and affecting vegetation assemblages. Thus, control sites responded in a similar multivariate direction as thinned (cut-only) sites, but to a lesser degree. Burn-only and cut-burn sites shifted significantly toward greater density of ground level seed producing annuals and perennials. All manipulated sites showed increases in ground level seed producing plants, thus cutting and burning did achieve, to some measure, intended management objectives at the MMCWMA.

Small mammal trapping data were collected for a total of 25,750 trap-nights, adjusted to 21,596 when disturbance by larger mammals (e.g., raccoons, skunks, opossums) was accounted for. During this time, there were 1016 total captures of 463 individuals. Capture rates varied from 0 to 45 captures per 100 trap-nights, with a mean of 5.58 captures per 100 trap-nights. Eight potential snake prey species were captured, although the vast majority of all mammals captured (92.7 percent) were either deer mice (*Peromyscus maniculatus*) or white-footed mice (*Peromyscus leucopus*). After adjusting for high trap disturbance rates and using different approaches to cope with departures from normality in trapping data, we are confident that there were significant changes in small mammal capture success over time and that the responses were dependent on treatment (Douglas 2010). Our data suggest that small mammal trapping success increased in cut, cut-burn, and control sites, while staying relatively unchanged at burn-only sites. The degree of increase in trapping success was greatest at cut-burn sites. Together with vegetation data, these results are consistent with the hypothesis that manipulations improved food availability in manipulated plots, and a numerical response of small mammals resulted. Enigmatically, burn-only sites exhibited some shift in vegetation but did not result in increased small mammal abundance. One possible explanation is that continued presence of closed canopy and reduction of ground level complexity did not favor small mammals. Also, snakes foraged in burn-only sites, and the lack of structural complexity compared to cut and cut-burn sites may have enhanced their prey capture success by reducing obstructions to sensory reception and strike mechanics. There is also evidence (Douglas 2010) that burn-only sites are thermally extreme (hotter than other treatments), which may adversely affect small mammals. Surprisingly, control sites also exhibited small but significant increases in small mammal abundance, which we attribute to increases in ground level seed production after a canopy-opening ice storm. In our more complete analyses (Douglas 2010), Lincoln-Peterson abundance estimates were highly correlated with capture success estimates, and both methods yield similar conclusions regarding increases in small mammals that serve as prey species for timber rattlesnakes. Thus, we concluded that snake prey abundance increased at cut-only and cut-burn sites relative to control and burn-only sites.

Prior to habitat manipulations, the growth rates of snakes that used areas planned for manipulation were not statistically distinguishable from snakes using control areas (Douglas 2010). However, after application of thinning and fire, snakes that used manipulated sites exhibited statistically significant increases in growth rates (Douglas 2010). Body condition in snakes using control areas appeared to decrease from 2005 to 2009, whereas body condition for snakes using manipulated areas did not change and was maintained at healthy levels. Thus, control areas were poor sites for snake mass gain throughout the duration of the study, whereas the best plots were the manipulated sites. Furthermore, a regression of body condition on proportion of time spent in manipulated plots was significant suggesting increased foraging success in manipulated plots (Douglas 2010).

Growth rates and body conditions of timber rattlesnakes were improved for snakes that used manipulated areas after manipulations occurred. This was consistent with expectations from previous studies of the effects of elevated resource levels on rattlesnakes (Beaupre 2008, Taylor and others 2005). Also, use of manipulated habitat for a greater proportion of a snake's active season was associated with higher body condition index (Douglas 2010). Therefore, increased use of manipulated habitats appears to result in healthier snakes. In light of these data, we suggest that not only does habitat manipulation enhance habitat quality for timber rattlesnakes, but also that both body condition and growth rates of timber rattlesnakes may be viable indicators of the effectiveness of large-scale habitat manipulations. Timber rattlesnakes responded rapidly to changes in resource environment, and their physiological responses were more immediately and unambiguously measured than changes in small mammal abundance or capture success which is highly variable and requires very large allocation of resources and effort.

CONCLUSIONS

Our two case studies located in the same experimental system offer dramatically different lessons regarding the response of timber rattlesnakes to the application of fire in an Ozark oak ecosystem. We offer several caveats regarding our observations and the quality of our experimental data. Results obtained are clearly specific to our sites and our general approach. The effects of logging and fire application on reptiles in general are likely highly variable, and dependent on a number of relevant conditions. For example, as demonstrated by Case 1, the severity and timing of fire application will likely have a profound effect on outcomes for ground-dwelling reptiles. Application of fire immediately after emergence had a local but devastating effect on ground-dwelling, leaf-litter hiding timber rattlesnakes. For Case 2, prior knowledge of the pending manipulation allowed us to negotiate with AGFC to apply fire to the system prior to spring emergence of most large-bodied reptiles present in our system. Foreknowledge and planning resulted in dramatically different outcomes than those seen at Site K. Thus, timing is critical, and at least for the preservation of timber rattlesnakes, fire is probably best applied during their natural dormant season.

The vulnerability of timber rattlesnakes and other large-bodied snakes to fire during the growing season is unknown. Aside from post-spring emergence, snakes in particular may also be vulnerable during ecdysis, when eyesight and other senses are impaired as old skin lifts and separates from new tissue in preparation for shedding (Means and Campbell 1981, Russell and others 1999). Clearly, the ability of a species to endure fire will depend upon its evolutionary history and the frequency and intensity of fires in its native habitat. Nevertheless, some accounts in the literature suggest that rattlesnakes can survive low intensity growing season fires in nature. For example, only 2 of 68 marked eastern diamondback rattlesnakes (*Crotalus adamanteus*) succumbed to direct effects of prescribed fire in a Florida scrub ecosystem (Means and Campbell 1981). Likewise, low intensity fires in desert island systems can be survived by montane rattlesnake species (Smith and others 2001). However, in the French Alps, an autumn grassland burn (while snakes were still active) doubled the mortality rate of the highly endangered Orsini's viper (*Vipera ursinii*), and surprisingly, the snakes showed no improvement in body condition in the postfire environment (Lyet and others 2009). It is also clear that intense growing season fires can be catastrophic for rattlesnakes in some systems. The first author witnessed such an event in the Sonoran Desert in July 1995 when a thriving population of western diamondback rattlesnakes (*Crotalus atrox*) was locally extirpated, mostly by indirect effects, in the aftermath of the Rio Verde fire which consumed over 20,000 acres (Beaupre 1995). In any event, growing season fires should be expected to produce some mortality and possibly high mortality under some conditions. For healthy populations of common species, this mortality may be tolerable. For small relict populations of sensitive or threatened species, dormant season burns or alternatives to fire should be considered.

We also suggest that initial conditions of the manipulated forest will profoundly influence outcomes. Degraded forests are likely to benefit through increases in heterogeneity, and associated food production. Healthy climax forests would likely be adversely affected. Furthermore, maintenance of a healthy disturbance-dependent state requires not only the initial manipulation, but frequent return of fire to the system. However, optimal intervals of fire return for most vertebrates are unknown. In addition, climate-related weather events (e.g., ice storms, late freezes) have the capacity to impact experimental data, providing less than perfect confidence in all interpretations.

Finally, we caution the reader by reminding of some shortcomings in our Case 2 experimental design. In violation of the assumptions of a formal BACI design (Williams and others 2001), our sites were not chosen at random but were restricted to ridge tops and areas with sufficient logging access to saleable timber. This is less an admission of flawed experimental design and more of an acknowledgment of reality and the need to seize opportunities to forward our understanding, however imperfect they may be. Furthermore, all of our postmanipulation observations were made under transient successional dynamics. A critical question is whether differences in vegetation, small mammal abundance, and indicators of rattlesnake health will persist as the system settles into a long-term disturbance-dependent state.

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AN EMERGING FIRE PROGRAM

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GENESIS OF AN OAK-FIRE SCIENCE CONSORTIUM

Keith W. Grabner, Michael C. Stambaugh, Richard P. Guyette, Daniel C. Dey, and Gary D. Willson

Ecologist (KWG), USDI Geological Survey, Columbia Environmental Research Center, 4200 New Haven Rd, Columbia, MO 65201; Research Associate (MCS); Professor (RPG); Research Forester (DCD); Research Coordinator (GDW). KWG is corresponding author: to contact, call 573-441-2782 or email at kgrabner@usgs.gov.

Abstract.—With respect to fire management and practices, one of the most overlooked regions lies in the middle of the country. In this region there is a critical need for both recognition of fire's importance and sharing of fire information and expertise. Recently we proposed and were awarded funding by the Joint Fire Science Program to initiate the planning phase for a regional fire consortium. The purpose of the consortium will be to promote the dissemination of fire information across the interior United States and to identify fire information needs of oak-dominated communities such as woodlands, forests, savannas, and barrens. Geographically, the consortium region will cover: 1) the Interior Lowland Plateau Ecoregion in Illinois, Indiana, central Kentucky and Tennessee; 2) the Missouri, Arkansas, and Oklahoma Ozarks; 3) the Ouachita Mountains of Arkansas and Oklahoma; and 4) the Cross Timbers Region in Texas, Oklahoma, and Kansas. This region coincides with the southwestern half of the Central Hardwoods Forest Region. The tasks of this consortium will be to disseminate fire information, connect fire professionals, and efficiently address fire issues within our region. If supported, the success and the future direction of the consortium will be driven by end-users, their input, and involvement.

INTRODUCTION

With respect to fire management and fire practices, one of the most overlooked regions lies in the middle of the country (Pyne 2009). In this region there is a critical need for both recognition of fire's importance and sharing of fire information and expertise. Fire science delivery is severely limited with dissemination of research results relying primarily on scientific publications or presentations, if the research is disseminated at all. There is no formal fire science laboratory at any level (federal, state, or private) and fire information delivery is primarily through the Forest Service's Northern Research Station. Smaller groups such as the Fire Learning Network (The Nature Conservancy) and Prescribed Fire Councils exist, however there is limited correspondence, collaborative activities, or advertisement by these groups, particularly at a large scale and encompassing regional fire regimes. Many prescribed fire programs and demonstration plots exist in the region that would benefit fire discussions between practitioners and scientists. Long-term research programs that provide cutting-edge fire information exist in the region (e.g., OK-FIRE [Carlson and others 2010], pyric herbivory [Fuhlendorf and others 2009], fire history) that would benefit fire managers outside the region; however their exposure has been limited.

We are in the planning stages towards developing a fire information consortium that will promote the dissemination of fire information across the interior United States and that will focus on fire information needs of oak-dominated communities such as woodlands, forests, savannas, and barrens. A consortium is defined as a group of individuals or organizations formed to undertake an activity that is beyond the capabilities of the individual members. This consortium will connect fire professionals and will address fire issues within the region. Consortium activities will be determined by a needs assessment of end users, but are likely to include

- developing an information clearinghouse (e.g., website) to serve the consortium region;
- highlighting regional fire management and research;
- development/access to online fire bibliography;
- presentations via webinars regarding regional management or research activities;
- organization of fire management and fire science symposia at national, regional, and state conferences; and
- supporting and initiating field visits for fire professionals to meet and discuss their respective needs.

FIRE CONSORTIUM REGION

The geographic region of this consortium consists primarily of the oak-hickory and western mesophytic forest regions (Braun 1950). The region is bordered by the tallgrass prairie of the Great Plains to the north, the Appalachian Mountains to the east, the Mississippi alluvial plain and the Upper West Gulf Coastal plain to the south, and the southern mixed and shortgrass prairie to the west (Fig. 1). The region covers nearly 1 million hectares and touches 11 states. The unifying features of this region include: 1) historically (last 1000+ yr) oaks were a dominant tree species and 2) fire was an important disturbance affecting vegetation development and the corresponding habitat. Oaks remain an important component of this region despite fire disturbance being largely lost due to fire suppression, land use changes, agriculture, and urbanization. Within this region, agricultural commodities (e.g., livestock, crops, and wood products) are economically important and have had important influences on fire regimes.

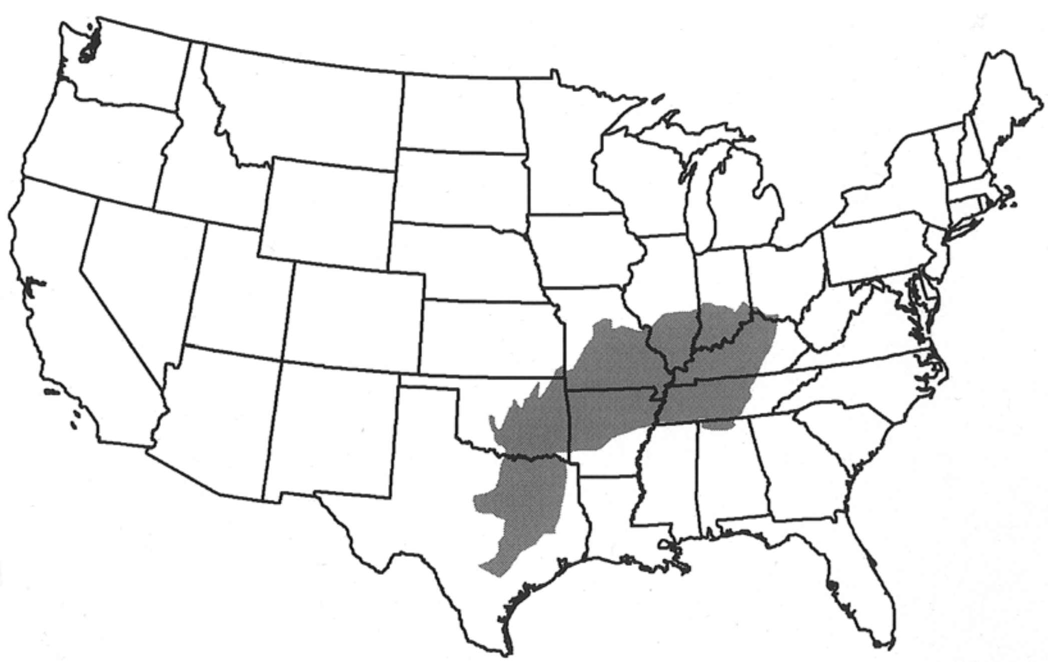

Figure 1.—The approximate region of the planned oak-fire consortium.

Largely due to historic repeated burning, many oak communities within this region were comprised of savannas and woodlands with open canopies capable of supporting diverse herbaceous plant assemblages. In addition to fire, grazing by native herbivores likely influenced this vegetation structure, however little is known to what extent and importance. Furthermore, prolonged drought, though potentially infrequent, was likely an important factor influencing vegetation and annual acres burned. Historically, fire frequency broadly varied along a latitudinal gradient that was a function of the region's continental climate (Guyette and others 2006). Fire frequencies likely averaged 10 years or less for these fire-maintained communities. Fire regime characteristics common to the region include: dominance of dormant season fire events, relatively low lightning ignition frequency, surface fires, low- to mixed-severity fires, and potential for broad extent fires in areas with low to moderate topographic relief.

Today, prescribed fire is commonly used in the region for ecosystem management, but its acceptance ranges widely (McCarty 2005). Roughly half a million acres or more currently burn annually from prescribed and wildland fires. Fire research has historically and is currently being conducted within the region by scientists at federal and state agencies and universities. Key fire organizations include the U.S. Forest Service, Northern Research Station Units (Delaware, OH, Columbia, MO, and the Eastern Area Modeling Consortium), Oklahoma State University, Texas A&M University, Texas Tech University, University of Arkansas, University of Illinois, University of Kentucky, and University of Missouri. Fire science is generally communicated to end users via federal agency proceedings, scientific journals, university extension, and professional societal meetings.

The region contains a diverse representation of federal public lands including: 5 national forests, 11 national park service units, 11 wildlife refuges, 1 Native American administered land, and numerous Department of Defense managed lands. Currently there is no formal group that organizes and disseminates fire science for agencies or organizations within the region. Many challenges of fire managers are specific to the region proposed. Key regional challenges related to fire include: controlling tree expansion into woodlands (juniper, oak, and pine), restoration of historic vegetation communities, oak regeneration, ecosystem management, invasive and exotic species control, managing small remnant grasslands, and assessing and managing fire risk in grassland and woodland ecosystems.

FIRE CONSORTIUM ORGANIZATION

Ultimately much of the planning and activities of the consortium will be determined from surveys and needs assessments of the end users. Although the consortium proposal was led by the authors we envision the consortium to be a user-driven consortium in that its goals and activities come from input of end-users. During proposal development fire professionals from throughout the region expressed their interest in the consortium formation. Consortia already developed in other regions of the United States have provided examples and guidance for how this consortium may be organized. We intend to provide a fire science delivery system for information that targets a wide range of end users and utilizes existing Web resources where available. We intend to design the consortium to be an inclusive organization that will provide information for fire science professionals to fire novices.

Based on the guidance of the Joint Fire Science Program, we expect the consortium to have a coordinator, governing board, and advisory board. Activities of the consortium will be directed by a governing board comprised of fire managers, practitioners, and scientists. The governing board will provide a long-term vision, guidance, and review of consortium effectiveness. These individuals will be geographically distributed across the consortium region, and represent a collaborative and multi-institutional team.

The advisory board will guide the activities of the consortium. The role of the advisory board will be to ensure the consortium is addressing its overarching mission, meeting the goals of the Joint Fire Science Program's Fire Science Delivery Network, and effectively administrating programs. The advisory board will evaluate the effectiveness of the consortium by interviewing end-users and evaluating the amount of participation in consortium supported conferences or workshops. The advisory board will consist of representatives from state and federal agencies, regional nongovernmental organizations (NGOs), private landowners, and private industry. Additionally the advisory board will include a mix of researchers, managers, and administrators. This mix will ensure that the consortium is addressing as wide range of fire issues pertinent to the region.

Intended Focus of the Consortium

The guiding principles of the consortium will mirror those of the Joint Fire Science Delivery Network:

1. Be inclusive; make sure all partners have the opportunity to be involved.
2. Serve as a neutral science partner.
3. Be customer driven, both in structure and function.
4. Operate collaboratively by fostering joint management and science communication.
5. Be innovative; pursue new and creative ways to disseminate knowledge.
6. Facilitate the flow of fire science information, dialogue of new science findings, and needs of resource managers and policymakers.

FUTURE WORK

The consortium will collate fire science information and develop a method for sharing this information in an interactive format. The primary focus of the consortium will be to assist in the dissemination of fire science information relevant to the needs of fire practitioners and researchers. The consortium will work to address the objectives of the Joint Fire Science Programs fire science delivery network.

Dissemination of information and building relationships. Information will be disseminated by the consortium through many outlets. A website will act as a portal for regional fire information, events, and regional fire professional contacts. Research highlights and publications will be made available through the website. The consortium will utilize traditional web-based information sharing tools but will remain receptive to emerging methods for information distribution (e.g., mobile messaging).

Listing and describing existing research and synthesis information/methods to assess the quality and applicability of research. One of the first goals of the consortium will be to identify fire science research that has occurred within the region and develop an online bibliography that will include peer reviewed literature, gray literature, and internal reports. While the bibliography is being developed, manuscripts will be reviewed and the information assessed for applicability across the region. In addition to developing a regional bibliography, a geodatabase of current fire research will be created, perhaps using existing resources (e.g., Conservation Registry).

Demonstrating research on the ground. The consortium will ensure that fire research and monitoring projects within the region are highlighted. Updates of ongoing fire research will be made available through the consortium using multiple methods. We will host field days where researchers and fire managers can meet and interact at fire research or management sites and receive tours of on-the-ground operations. Additionally, field projects will be highlighted through webinars. Webinars will permit information sharing with consortium members across the region without the time and expense of travel. Webinars will be archived and made available for future access.

Other methods for highlighting research occurring in the region will include consortium participation in regional natural resource conferences. Regional conferences include, but are not limited to, the Central Hardwoods Conference, the North American Prairie

Conference, the Midwest Savanna and Woodland Conference, and the Midwest Fish and Wildlife Conference. Although the consortium's focus will be regional, we will also be represented at national fire meetings.

Adaptive management. The consortium has the potential to act as a catalyst for the development of adaptive management projects across the region. By enabling communication between researchers and fire managers throughout the region, the consortium can promote cooperation and collaboration in the development of studies utilizing an adaptive management framework.

New partnerships. The consortium will promote establishing new relationships between fire researchers and practitioners. Consortium led meetings such as field days, regional conferences, and teleconferences will enable managers to provide input into fire research direction and needs.

Needs assessment for end-user communities. A needs assessment will be conducted to identify end-users and their fire information needs. Needs assessments will be conducted using an online survey, phone interviews, and face-to-face interactions. We envision end-users will include private landowners, private, state, federal, and Native American land managers, fire professionals, and scientists. In addition, we expect that over 50 colleges and universities and numerous NGOs within the region will become active in the consortium.

CONCLUSION

A final proposal for this planned consortium will be considered in October 2011. If funded, consortium operations should be underway quickly based on the planning-phase work. Existing consortia across the United States have been funded with 2-year budgets with possible continuation. Additional support for consortia is leveraged from partners who benefit from its existence.

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EXTENDED ABSTRACTS

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GROWTH RESPONSE OF MATURE OAKS FOLLOWING TSI AND PRESCRIBED BURNING TREATMENTS

Margaret Anderson, Eric J. Holzmueller, Charles M. Ruffner, and John W. Groninger

Research Assistant (MA), Assistant Professor (EJH), Professor (CMR and JWG). Department of Forestry, 1205 Lincoln Drive – MC 4411, Southern Illinois University, Carbondale, IL 62901. EJH is corresponding author: to contact, call 618-453-3708 or email at eholzmue@siu.edu.

ABSTRACT

Oak dominated forests were historically maintained by periodic, low intensity fires in southern Illinois, but over the past century, fires have generally been suppressed, resulting in the increased abundance of fire intolerant species and decreased oak abundance (Ruffner and Groninger 2006). Recent research indicates that mechanical thinning (timber stand improvement, TSI) and prescribed fire increase the regeneration of oak species and suppress the development of fire intolerant species (Carril 2009). However, it is unclear how the residual oak trees respond to these treatments. The objective of this project was to assess how two forest management practices, TSI and prescribed fire, affect the growth of residual trees in oak dominated forests.

Three sites were selected for this study. Each site was similar in terms of slope, aspect, elevation, and site index. Prior to treatment all sites had a two-aged structure and were dominated by oak and hickory species in the overstory and sugar maple, white ash, and American elm in the understory. Each site was divided and randomly assigned one of four treatments: 1) TSI, 2) prescribed burning, 3) TSI and prescribed burning, or 4) no treatment (control). TSI treatments removed all undesirable trees < 8 inches in 2002. Stumps were treated with herbicide (Garlon 4, Triclopyr - 16 percent a.i.) immediately following cutting. Prescribed burning treatments were applied in the spring of 2002 and 2006. In 2010, we sampled 10 dominant or codominant oak trees from each treatment at each site to determine differences in growth among treatments.

Results indicated that there was no difference in growth, either pretreatment or posttreatment, among the three sites ($P > 0.12$). Mean age of the sampled trees across all sites was 102, and posttreatment growth across all treatments was minimal, ranging from 0.07 to 0.08 inches/year. There did not appear to be any increased growth following the cutting and/or burning treatments when compared to pretreatment growth ($P = 0.63$), and these treatments did not differ when compared to the control ($P = 0.58$). Growth, however, did appear to be related to tree age ($r^2 = 0.44$, $P = 0.01$) indicating that, on average, younger trees grew slightly more than older trees.

Although TSI and/or prescribed fire did not increase residual oak growth when compared to the control, these treatments can still be beneficial because of the associated increased oak regeneration (Carril 2009). Examination of the growth rates of the cored trees over time indicated these stands were mature and well beyond peak annual growth. Our results suggest that working with younger stands and/or additional management (e.g., heavier cutting) may be necessary to increase residual tree growth.

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USING REMOTELY-SENSED IMAGERY TO MONITOR POST-FIRE FOREST DYNAMICS IN UPLAND OAK FORESTS ON THE CUMBERLAND PLATEAU, KENTUCKY

Mary Arthur, Christine McMichael, and Gretchen Sovkoplas

Professor (MA and GS), Department of Forestry, University of Kentucky, 214 T.P. Cooper Building, Lexington, KY 40546-0073; Associate Professor (CM). MA is corresponding author: to contact, call 859-257-2852 or email at marthur@uky.edu.

ABSTRACT

Throughout the central hardwood and southern Appalachian regions, forest managers are increasingly using prescribed fire to achieve a suite of management objectives. When such management occurs on federal lands, monitoring is mandated to determine the efficacy of prescribed fire to achieve stated objectives, yet limited funding for such efforts restricts the number, size, and spatial array of monitoring plots. Monitoring is typically done with a small number of monitoring plots across very large landscapes, and therefore, is limited in its ability to reveal the true landscape-scale impact of forest management. Based on its use in other forested ecosystems, remote sensing data have the potential to provide accurate and relatively inexpensive information in support of landscape-scale management objectives in central hardwood and southern Appalachian forests. Here we describe our preliminary efforts to develop this approach.

We used Landsat 5 Thematic Mapper satellite imagery to: (1) calculate the Normalized Difference Vegetation Index (NDVI) as a function of the red and near-infrared reflected energy, such that greener vegetation has a higher NDVI; (2) examine changes in the NDVI with burn treatment; and (3) analyze the relationships between NDVI and canopy cover measured using hemispherical photography (Alexander and others 2008). Buffered polygons were created from aerial photos to encompass each of 93 forest plots arrayed across three study sites, each with three treatments: fire-excluded, frequent burn (spring 2003, 2004, 2006, and 2008), and infrequent burn (spring 2003, 2009). The polygons were matched with plot-scale locations to capture a larger area of similar vegetation. We used Landsat 5 Thematic Mapper summer (typically June) images obtained for each year from 2002 to 2010 to calculate NDVI. Plots were located on differing landscape positions and were classified as sub-mesic, intermediate, or sub-xeric based on vegetation. Analysis of variance (ANOVA) was used to test the main effects of treatment (nested within site) and landscape position on NDVI in each year. Regression analysis was used to test the strength of the relationship between NDVI measured in one year relative to another year. Regression was also used to examine the relationship between NDVI and canopy cover.

As expected, prior to burning there was a close relationship between the NDVI in 2001 and 2002 ($R^2 = 0.66$, $p < 0.0001$). In summer 2003, after both sites were burned the previous spring, the NDVI was lower (less green) on burned compared to fire-excluded sites ($p = 0.0004$). Comparing NDVI in 2003 with 2002, the R^2 for the relationships varied from $R^2 = 0.53$ for control, $R^2 = 0.21$ for less frequent, and $R^2 = 0.17$ for frequent sites (after one burn had been conducted on both treatments in 2003). Differences in NDVI were greatest the first year after burning, despite repeated burning in subsequent years. By 2006, NDVI on less frequent burn sites was similar to that on fire-excluded

sites; both had higher NDVI compared to the frequent burn sites, which by 2006 had been burned three times, including the previous spring. A second burn in 2009 on the less frequent burn sites reduced NDVI below that of the frequent sites, but NDVI rebounded the following year.

We also examined the relationships between NDVI and canopy cover on a subset of sites (N = 33) from 2002 through 2007. In 2002, there was little variability in canopy cover or NDVI across these sites; although the relationship was significant, canopy cover explained little of the variability in NDVI ($R^2 = 0.11$; $p = 0.03$). In subsequent years, after burning the relationship between NDVI and canopy cover was strongly positive. For example, the first growing season after the first burn (2003), canopy cover explained 50 percent of the variability in NDVI ($p < 0.0001$). In 2007, after three burns in the frequent sites and one burn in the less frequent sites, there was still a strong positive relationship between canopy cover and NDVI ($R^2 = 0.38$; $p < 0.0001$), but the slope of the relationship was lower than in 2003.

The intent of the prescribed burning in our study was to remove midstory stems of fire-sensitive species with fairly small changes to the overstory. Nonetheless, immediately after the first fire, following which we found the greatest stem mortality (Arthur, unpublished data), there were highly significant differences in NDVI among treatments, and a strong relationship between NDVI and canopy cover. As overstory canopy cover was replaced by understory resprouting (Arthur, unpublished data; Blankenship and Arthur 2006; Chiang and others 2005), increasing leaf area led to increasing NDVI such that by 2010 there were small but significant differences between the control and frequent burn sites, with the less frequent sites having NDVI similar to the controls.

Future work on this project will examine additional stand characteristics (crown health, stem density, basal area) in relationship to NDVI and other spectral vegetation indices (SVIs), explore differences among landscape positions in burning effects on SVIs, and determine whether combinations of SVIs can be used to distinguish between canopy cover and understory sprouting response to burning. Prescribed fire used to create a range of habitats with more open canopies, such as oak woodlands and savannas, may lead to forest canopies that are more easily detected using this approach. Thus, we plan to expand our analysis to a landscape recently burned by wildfire which created a greater range of impacts to the forest canopy, and thus may provide a further test of this approach.

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FIRESTEMII – A 2-D HEAT TRANSFER MODEL FOR SIMULATION OF STEM DAMAGE IN PRESCRIBED FIRES

Efthalia K. Chatziefstratiou, Gil Bohrer, Anthony S. Bova, and Matthew B. Dickinson

Graduate Research Associate (EKC) and Assistant Professor (GB), Department of Civil and Environmental Engineering and Geodetic Science, 416 Bolz Hall, The Ohio State University, Columbus, OH, 43210; Physical Scientist (ASB); Ecologist (MBD). EKC is corresponding author: to contact, call 614-292-9582 or email at chatziefstratiou.1@osu.edu.

Abstract.— In this work, we present FireStemII, a software tool that allows fire managers to predict tree mortality and stem injury resulting from a prescribed forest fire. FireStemII is a revision of FireStem (Jones and others 2004, 2006). When prescribing a fire, it is important to have a tool that predicts tree mortality and the extent of stem injury given the fuel conditions, fire behavior, and height aboveground along the stem. FireStemII is a physically-based thermodynamic 2-D model of tree stem injury as a function of external heat forcing. It provides increased capability for predicting fire-induced mortality and damage before a fire occurs. By directly simulating tissue temperatures, moisture loss, and charring, it determines the depth and circumferential extent of damage caused by incident heat flux around a stem at a given height. These data are further integrated to provide a depth of necrosis around the stem and an index of vascular cambium viability.

INTRODUCTION

FireStemII simulates the influence of fire as a dynamic and spatially heterogeneous heat flux boundary condition around the circumference of a virtual slice of a stem. Several other numerical models for heat transfer in tree stems subjected to fire conditions have used temperature boundary conditions (Costa and others 1991) and were typically 1-D models (Rego and Rigolot 1990), not accounting for differential heating rates and injury responses around stems. FireStem was the first, to our knowledge, to include a heat flux boundary condition (Jones and others 2004). The realism and dynamics of FireStemII will allow it to be linked to a coupled fluid dynamics and combustion (fire) model in future work. Time series of heat flux will be simulated around stems at different heights aboveground and at different locations throughout a burn area (Fig. 1). For example, as the fire-line passes a tree, turbulence on the lee side of the stem leads to increased heating and injury at the back, while the sides and front of the stem are less affected (Gutsell and Johnson 1996).

Figure 1.—FireStemII as a 2-D model of tree stem injury takes into consideration the uneven heating in the circumferential direction in case of a fire.

Stem Heating Model

The model is based on a two-dimensional heat transfer formulation:

$$\frac{\partial \rho i}{\partial t} = \frac{1}{r}\frac{\partial}{\partial r}\left(rk\frac{\partial T}{\partial r}\right) + \frac{1}{r}\frac{\partial}{\partial \theta}\left(\frac{k}{r}\frac{\partial T}{\partial \theta}\right)$$

where r is radius (cm), k is conductivity (W/(m•K)), ρ is wood density (g/cm^3), i is specific internal energy (J), T is temperature (K), and θ is angle (radians).

Stem Simulation

Stems are simulated as circular slices out of infinitely long cylinders (Jones and others 2004). The required inputs to the model include geometric information (stem diameter, outer and inner bark thickness), and physical properties (thermal conductivity, density, specific heat, and moisture content in inner bark, outer bark, and wood). Stems are divided into radial wedges (we used 16 wedges), and each wedge was divided into nodes in the radial direction. The distance between nodes is flexible and was set here to 1 mm. The new feature of FireStemII in comparison to Jones and others (2004) is that the stem is modeled as two dimensional. In addition, the numerical solver was improved with a more robust Crank-Nicholson approach, and two new routines for desiccation and bark charring were used.

Physical Phenomena

Desiccation

The change in water mass at each nodal point is calculated at each time step. The relation for water loss was taken from Morvan and Dupuy (2001):

$$\frac{\partial w}{\partial t} = W_m \frac{k_w}{\sqrt{T}} \rho \cdot M \cdot \exp\left(-\frac{E_w}{RT}\right)$$

where W_m is a multiplier that allows the water loss rate to be adjusted, M is moisture (%), T is temperature (K), ρ is wood density (g/cm^3), the coefficient k_w (6.05E5 K$^{0.5}$/s) and the exponential factor E_w/R(5956 K) are taken from Morvan and Dupuy 2001.

Bark charring

Pyrolysis is modeled in a manner analogous to water loss with the exception that in each time step, charring can occur only one node away from a previously charred node. The rate equation is based on Ragland and Aerts (1991):

$$\frac{\partial p}{\partial t} = P_m \cdot A_p \cdot \rho \cdot (1 - c_f) \cdot \exp\left(-\frac{E_p}{RT}\right)$$

where P_m is the pyrolysis multiplier, c_f is the density fraction, the coefficient A_p (7E7 s^{-1}) and the exponential factor E_p/R(15610 K) are also taken from Ragland and Aerts (1991). The combustion of charred material (glowing combustion) is not modeled.

Tissue injury

The thermally caused mortality in population of cells is described by a rate equation, where the rate of decline in tissue viability is proportional to current viability (Dickinson and Johnson 2001; Dickinson and others 2004):

$$\frac{dV}{dt} = -\kappa V(t)$$

where V is viability value between 1 (survival) and 0 (cell death), t is time (s), and κ is a species-specific, temperature-dependent rate parameter. Cellular necrosis is assumed if V is reduced below 0.5.

EXPERIMENTAL METHODS

Controlled laboratory stem heating experiments were conducted on eight regional species: red maple (*Acer rubrum* L.), sugar maple (*Acer saccharum* Marsh.), mockernut hickory (*Carya tomentosa* [Poir.] Nutt.), yellow-poplar (*Liriodendron tulipifera* L.), blackgum (*Nyssa sylvatica* Marsh.), eastern white pine (*Pinus strobus* L.), chestnut oak (*Quercus prinus* L.), and northern red oak (*Quercus rubra* L.) in order to validate FireStemII. The stem sections were fitted with three thermocouples to monitor temperatures at bark surface, beneath the bark surface, and at the cambium.

Total mass loss with heating was also determined by measuring pre- and posttreatment stem mass. Finally, depth of necrosis into the sapwood following stem heating was determined by staining stem sections with tetrazolium chloride (TTC).

RESULTS AND DISCUSSION

The laboratory stem-heating experiments of 53 tree sections of tree species mentioned above were also simulated with FireStemII. The results from the physical experiments are compared with the results from FireStemII simulations of the same cases. (Figs. 2 and 3).

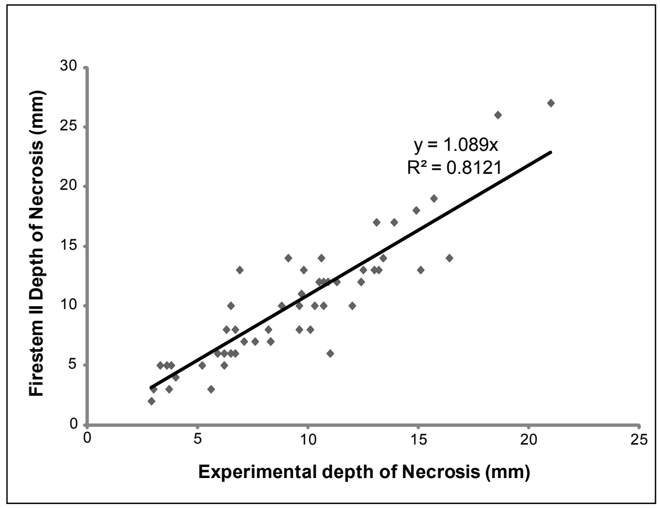

Figure 2.—Necrotic depth of 53 tree sections from eight different species. Heating experiments in the lab are compared with the necrotic depth estimated with FireStemII.

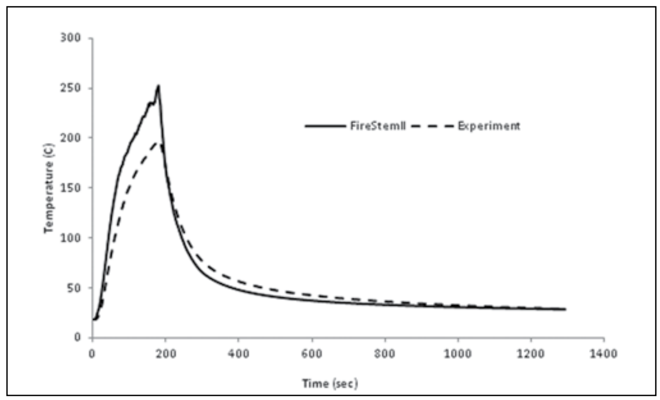

Figure 3.—Temperature variation through time just beneath bark surface in a chestnut oak tree section (stem diameter = 12.7 cm) in one lab experiment and from FireStemII simulation of the same case.

AKNOWLEDGEMENTS

The authors thank Joshua Levi Jones and Ravishankar Subramanian for collaboration and developments of earlier versions of the model. The study was funded by NASA-NESSF-Earth Sciences Fellowship #NNX09AO26 to Anthony Bova and Gil Bohrer, and through the U.S. Forest Service, Northern Research Station Agreement 09-CR-11242302-033. Bret Butler, Dan Jimenez, and Joe O'Brien provided useful feedback on an earlier draft of the manuscript.

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EFFECTS OF TWO PRESCRIBED FIRES ON RED MAPLE REGENERATION ACROSS FOUR LEVELS OF CANOPY COVER IN NORTHERN LOWER MICHIGAN

William W. DeBord, David S. Buckley, Terry L. Sharik, Jason P. Hartman, and Jason G. Henning

Graduate Research Assistant (WWD) and Associate Professor (DSB and JGH), Department of Forestry, Wildlife and Fisheries, 274 Ellington Plant Sciences Building, University of Tennessee, Knoxville, TN 37996-4563; Professor (TLS), Department of Wildland Resources, Utah State University; Unit Manager (JPH), Michigan Department of Natural Resources and Environment.
DSB is corresponding author: to contact, call 865-974-7978 or email at dbuckley@utk.edu.

INTRODUCTION

Mixed oak forests persist as a predominant forest type in eastern U.S. forests, but very little oak recruitment into the overstory has occurred since the implementation of fire suppression in the early 1900s (Abrams 1992). Advanced oak regeneration on high quality and some intermediate quality sites is being replaced in the midstory by more mesic, shade tolerant species (Abrams and Nowacki 1992). A large proportion of the total increase in the density of competitors can often be attributed to an influx of red maple (*Acer rubrum* L.) (Abrams 1998, 2005). Current thinking is that reintroduction of representative fire regimes into eastern oak systems may help promote oak recruitment and impede the development of red maple competition. It has been suggested that oak seedlings and saplings are better adapted to fire than those of red maple and other competitors due to greater allocation of resources to roots and several other adaptations (Brose and Van Lear 1998). As a result, it follows that multiple fires should place red maple and other competitors at an ever-increasing disadvantage relative to oak species.

As a component of a larger, long-term oak regeneration project established in 1990, the objectives of this study were to determine the additive effects of two spring prescribed fires on red maple regeneration across four levels of canopy cover, and test the hypothesis that red maple densities in medium (25 cm tall – 2.54 cm diameter at breast height [d.b.h.]) and large (>2.54 cm d.b.h.) size classes would be reduced to a greater degree following the second fire than following the first.

METHODS

Second-growth natural oak stands on state forest land in northern Lower Michigan were selected as study sites. The sites are located within the Grayling Outwash Plain of the Highplains District of the northern Lower Peninsula (Albert 1995). All stands occurred on sandy, mixed, frigid, Alfic Haplorthods developed in pitted outwash. Soils in all stands had comparable physical and chemical properties (Kim and others 1996). All sites were moderately productive with slopes ≤5 percent. Site index for northern red oak (*Quercus rubra* L.) was 17.8 m at 50 years (Carmean 1978). Based on counts of annual growth rings on stumps following overstory treatment implementation in 1991, the stands studied were 106-118 years old in 2009.

Each of three 1.74 ha replicate oak blocks was subdivided into four overstory plots measuring 66 × 66 m (0.44 ha). Each overstory plot was randomly assigned one of four canopy cover treatments: clearcut (CC), 25 percent residual canopy cover (25 percent), 75 percent residual canopy cover (75 percent), or uncut control (UC). All stands were cut between fall 1990 and early spring 1991. Four understory treatment subplots measuring 15 × 15 m (0.02 ha each) were arranged in a square at the center of each canopy treatment plot in 1991, and the understory treatments were maintained intermittently through 1996.

All stands were burned on May 15, 2002 and burned again on May 16, 2008. The goal in both years was to achieve strip-head fires with 0.9 m flame lengths in order to top kill all hardwood seedlings on all sites. Maximum fire temperatures were documented with temperature indicating paints (B.J. Wolfe Enterprises Inc., Agoura Hills, CA) painted on the bottom surface of horizontally oriented ceramic tiles. One ceramic tile was mounted on a steel stake 0.6 m above the soil surface in the center of each of the four understory plots within each overstory treatment plot.

Natural regeneration was measured in 2001, 2003, 2006, and 2009 at four sampling points within each understory plot near the end of the growing season in late July/early August. Genets of woody stems were tallied by species into three size classes: 1) Small (stems <25 cm height); 2) Medium (stems 25 cm height – 2.54 cm diameter); and 3) Large (stems 2.54 cm diameter – 10 cm diameter). Small size class stems were measured within 1 m^2 quadrats. Medium and large size class stems were measured within 2 and 4 m diameter circular plots, respectively. Smaller sampling plots were nested within larger plots, and all three plots were centered on each of the four sampling points within each of the understory plots.

Paired t-tests were used to evaluate differences between the differences in pre- and postburn seedling densities associated with each fire using SAS/STAT® software (version 9.2, SAS Institute Inc., Cary, NC). Hypothesis tests were performed with $\propto = .05$.

RESULTS AND DISCUSSION

Maximum fire temperatures ranged from 40 to 100 °C at 0.6 m above the soil surface. Temperatures were comparable in both years, and flame lengths were typically 0.3 m or less. In the case of medium and large red maple stems, in most of the overstory treatments there were differences between the differences in maple density before and after Fire 1 and the differences in density before and after Fire 2 (Table 1). Differences associated with the second fire were far smaller in most cases, however, than those associated with the first fire (Table 1). Consequently, the hypothesis that red maple stems in the two larger size classes would be reduced more by the second fire than by the first was not supported.

Many of the larger surviving red maple stems were stump sprouts that emerged from thinned midstory saplings following implementation of canopy treatments in 1991. These sprouts were able to develop over 11 growing seasons prior to the first prescribed fire. Densities may have been reduced to a greater degree following the first fire due to high mortality among stressed, ill-formed, or diseased stems, leaving only the most robust, well-established stems which then had six growing seasons to recover before the second fire. Reducing these robust stems may require continued repeated burning.

Prescribed fires implemented more frequently than in this study may help to reduce the density of large red maple stems that are able to survive and recover over longer time periods between fires. More frequent burns, however, will also have effects on oak stems. Overall, the two prescribed fires had a cumulative effect, reducing red maple densities by 46-74 percent in the medium size class and by 53-87 percent in the large size class, which may improve the future competitive position of oak seedlings and saplings.

ACKNOWLEDGMENTS

This research was funded by the University of Tennessee Department of Forestry, Wildlife and Fisheries, and the Michigan Department of Natural Resources (DNR). Special thanks to several Michigan DNR personnel and to Chris Miller, John Johnson, and Stephen Grayson for their help in the field.

Table 1.—Red maple densities by size class, year, and canopy treatment, differences between pre- and postfire densities associated with each prescribed fire, and results for paired t-tests conducted within canopy treatments to compare differences in red maple density associated with each fire.

Medium Size Class

Canopy Treatment[1]	2001 Density	2003 Density	Fire 1 Difference	2006 P-value[2]	2009 Density	Fire 2 Density	Difference
	Stems/ha	*Stems/ha*	*Stems/ha*		*Stems/ha*	*Stems/ha*	*Stems/ha*
CC	9416.68	4244.14	–5172.54	0.0605	4111.51	2453.64	–1657.87
25%	23475.37	13262.92	–10212.45	**0.0094**	14920.79	10941.91	–3978.88
75%	22480.65	12599.78	–9880.88	**0.0057**	17772.32	12135.57	–636.74
UC	16114.45	6432.52	–9681.93	**0.0063**	8289.33	4708.34	–3580.99

Large Size Class

Canopy Treatment	2001 Density	2003 Density	Fire 1 Difference	2006 P-value	2009 Density	Fire 2 Density	Difference
	Stems/ha	*Stems/ha*	*Stems/ha*		*Stems/ha*	*Stems/ha*	*Stems/ha*
CC	2287.85	1177.08	–1110.77	**0.0082**	1110.77	1077.61	–33.16
25%	1823.65	629.99	–1193.66	**0.0035**	613.41	464.20	–149.21
75%	646.57	132.63	–513.94	**0.0388**	198.94	82.89	–116.05
UC	116.05	16.58	–99.47	0.3400	66.31	16.58	–49.74

[1]CC = clearcut; 25 percent = 25 percent canopy cover; 75 percent = 75 percent canopy cover; and UC = uncut controls.
[2]Bold P-values indicate statistically significant differences between differences in red maple density before and after Fire 1 and differences in red maple density before and after Fire 2 suggested by paired t-tests with alpha = 0.05.

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TALL LARKSPUR BENEFITS FROM FIRE IN CHINKAPIN OAK WOODLANDS

Dan Drees

Fire Ecologist, National Park Service, Ozark National Scenic Riverways, Van Buren, MO 63965.
To contact, call 573-323-8234 ext. 26 or email at daniel_drees@nps.gov.

ABSTRACT

Tall larkspur (*Delphinium exaltatum*) is a globally rare wildflower (NatureServe 2010). In Missouri, tall larkspur is strongly associated with chinkapin oak (*Quercus muehlenbergii* Engelm.) dominated woodlands (85 percent of sites) (Mullarkey and others 2008). It is an important oak woodland indicator species that requires strong filtered sunlight to maintain viable populations. In the absence of fire, the woodlands that tall larkspur prefers are subject to succession towards forest. The subsequent increase in shade greatly reduces tall larkspur populations (Fig. 1).

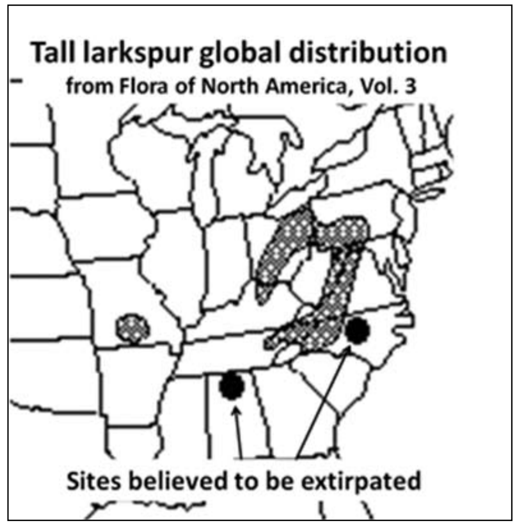

Figure 1.

According to www.natureserve.org, “A primary threat to tall larkspur is loss of habitat due to succession of vegetation in the absence of a natural fire regime. Encroachment of trees and shrubs (e.g., eastern redcedar [*Juniperus virginiana* L.]) into occupied habitat has likely resulted in the loss of many individuals and populations over time.” Historically, fire kept woody plants from becoming too dense in tall larkspur’s preferred habitat. This attractive wildflower may persist naturally without fire for decades on some sunny, steep slopes and the sunny banks of small high-gradient streams. This is especially true if windstorms, ice storms, or selective timber harvesting have thinned the tree canopy. However, only fire will prevent leaf litter accumulation from suppressing tall larkspur seedling germination over most of its preferred habitat.

Tall larkspur population inventories at Ozark National Scenic Riverways (ONSR) between 1984 and 2008 documented dramatic population declines at unburned sites. The largest population (650 plants) declined by 64 percent in total number and by 94 percent in number of reproductive plants. In 2009, the Ozark Highlands fire ecology crew of the National Park Service discovered a new population of 2,481 tall larkspur plants in a fire managed woodland at ONSR. It is the largest reported natural population of tall larkspur in a prescribed fire management unit anywhere.

In 2010, ONSR embarked on an ongoing project with permanent plots to study the fire ecology of tall larkspur in the Current River watershed where 15 of 17 populations in Missouri are located. We would like to collaborate with managers of natural tall larkspur populations throughout the plant’s range.

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LOCAL COLLARED LIZARDS NEEDED LANDSCAPE-SCALE FIRE

Dan Drees

Fire Ecologist, National Park Service, Ozark National Scenic Riverways, Van Buren, MO 63965.
To contact, call 573-323-8234 ext. 26 or email at daniel_drees@nps.gov.

ABSTRACT

Evolutionary biologist Dr. Alan Templeton used 15 years of field research on eastern collared lizards to advocate for landscaped-scale prescribed burn units beginning in 1994. Another 18 years of field research by Dr. Templeton and his colleagues demonstrated that landscape-scale burns are necessary for a variety of reasons. In addition to restoring glade habitat for collared lizards, the large burns were necessary to open surrounding woodlands as travel corridors (Templeton and others 2007). Burning the surrounding woodlands permitted additional glades to be colonized and facilitated genetic dispersal.

Dr. Templeton started in 1979 with a search of the Stegall Mountain vicinity (Current River watershed in Missouri) to determine if historic populations of the collared lizard still existed. No collared lizards were observed in the initial surveys. The areas numerous igneous and dolomite glades had become too overgrown in trees, especially eastern redcedar. The historic populations of collared lizards probably died out due to shading of their open glade habitat.

In 1982, cedar removal and small prescribed fires were initiated to reopen the best remaining glades on nearby Stegall Mountain. In 1984, collared lizards were captured from several healthy populations 40 to 80 miles away and released on Stegall Mountain. After a few years, Dr. Templeton realized that collared lizards were not moving to nearby glades that were separated by thin bands of dense woody vegetation, some only 50 meters wide.

Dr. Templeton, as part of a Biodiversity Task Force commissioned by the Missouri Department of Conservation, suggested burning several hundred acres of glades and woodlands at once. At the time, this was a radical concept in Missouri. In 1994, the first landscape-scale burn was completed on Stegall Mountain. The collared lizards immediately responded by colonizing several nearby glades.

One of Dr. Templeton's students found that grasshoppers were the primary food of local collared lizards (Östman and others 2007). Another student discovered that there was a 650 percent increase in grasshoppers in the prescribed burn units (Jon Hallemeier and A. Templeton, unpublished results). Grasshoppers and other ground dwelling insects are also the primary food for turkey hatchlings and many species of grassland birds.

Indeed, the National Wild Turkey Federation (NWTF) has recognized for a long time that glades and their surrounding woodlands should be properly managed with prescribed fire to create the optimal mix of nesting cover, brood rearing habitat, and acorn production. Locally, the NWTF has been an important ally in promoting the restoration of fire-dependent habitats that support a diversity of rare plants and animals in addition to important game species.

Dr. Templeton's dream was to see at least 2,000 contiguous acres in prescribed fire management. In 2009, the National Park Service, the Missouri Department of Conservation, and The Nature Conservancy joined forces to complete over 5,000 contiguous acres of prescribed burns in the Stegall Mountain area in just one season.

Collared lizards have now colonized all of the suitable glades within these adjoining burn units. Most importantly, the collared lizards are indicators that landscape-scale prescribed fire has restored, stabilized, or increased populations of many other fire-dependent conservative species.

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EFFECTS OF WILDFIRES AND LIMING OF PINE-OAK-HEATH COMMUNITIES IN THE LINVILLE GORGE WILDERNESS, WESTERN NORTH CAROLINA

Katherine J. Elliott, James M. Vose, Jennifer D. Knoepp, and William Jackson

Research Ecologist (KJE), Research Ecologist and Project Leader (JMV), and Research Soil Scientist (JDK), U.S. Forest Service, Southern Research Station, Coweeta Hydrologic Laboratory, Otto, NC 28763; Air Quality Specialist (WJ).
KJE is corresponding author: to contact, call 828-524-2128 or email at kelliott@fs.fed.us.

ABSTRACT

Linville Gorge Wilderness (LGW) is a Class I area in the southern Appalachian Mountains, western North Carolina. Over the last 150 years, LGW has been subject to several wildfires, varying in intensity and extent (Newell and Peet 1995). In November 2000, a wildfire burned 4000 ha in the wilderness; the fire ranged in severity across the northern portion of the wilderness from low severity in coves to high severity along ridges and bluffs (Reilly and others 2006). In May 2007, the Pinnacle wildfire burned ca. 2000 ha in the southern portion of the wilderness. A large portion of the Pinnacle fire overlapped the area previously burned in 2000, resulting in much of the central section of LGW being burned twice in less than 7 years. In addition, dolomitic lime was aerially applied at a rate of 1120 kg ha^{-1} on the most severely burned area. We hypothesized that liming the most severely burned area would accelerate the restoration of acidic, nutrient depleted soils by adding basic cations, balancing soil pH, reducing soil and soil solution Al, and subsequently increase ecosystem productivity. We sampled vegetation (composition, biomass, and foliar nutrients) and soil nutrients in five treatment areas in LGW over a 2-year period following the most recent wildfire. The treatment areas were: severely burned twice (2000 and 2007) plus dolomitic lime application (2xSBL); moderately burned twice plus lime (2xMBL); severely burned twice, no lime (2xSB); moderately burned once (2000), no lime (1xMB); and an unburned and unlimed reference area (REF).

All wildfire burned sites experienced overstory mortality (>300 stems ha^{-1}). There were no live overstory trees on 2xSBL, and 2xSB had a larger number of dead trees than all other sites. The large number of dead pines on all sites, including the reference, was due in large part to a southern pine beetle (*Dendroctonus frontalis* Zimm.) outbreak in 1999-2001. The three areas impacted by the recent wildfire had little to no live pine in the overstory. Overstory biomass was lower on the severely burned areas (2xSBL and 2xSB) than the other treatments, with no significant change between 2008 and 2009 (Fig. 1). By 2009, understory density was higher on 2xSBL than the other treatments (Fig. 2a), with higher numbers of tree species (Fig. 2b). We found no differences in shrub density among burned treatments, and only 2xSB had higher shrub density than REF (Fig. 2c). As expected, ericaceous (heath) species sprouted after fire, and their density increased between 2008 and 2009 for all recently burned treatments.

We found no differences in foliar nitrogen (N) concentration among treatments for either evergreen or deciduous species (Table 1). Evergreen foliar calcium (Ca) was greater on 2xMBL and REF than the severely burned sites (2xSBL and 2xSB); whereas deciduous foliar Ca and phosphorus (P) were greater on 2xMBL than 2xSBL, 1xMB and REF (Table 1).

With few exceptions, we found significant date, treatment, date x treatment interaction effects for soil exchangeable cations for both shallow (0-10 cm

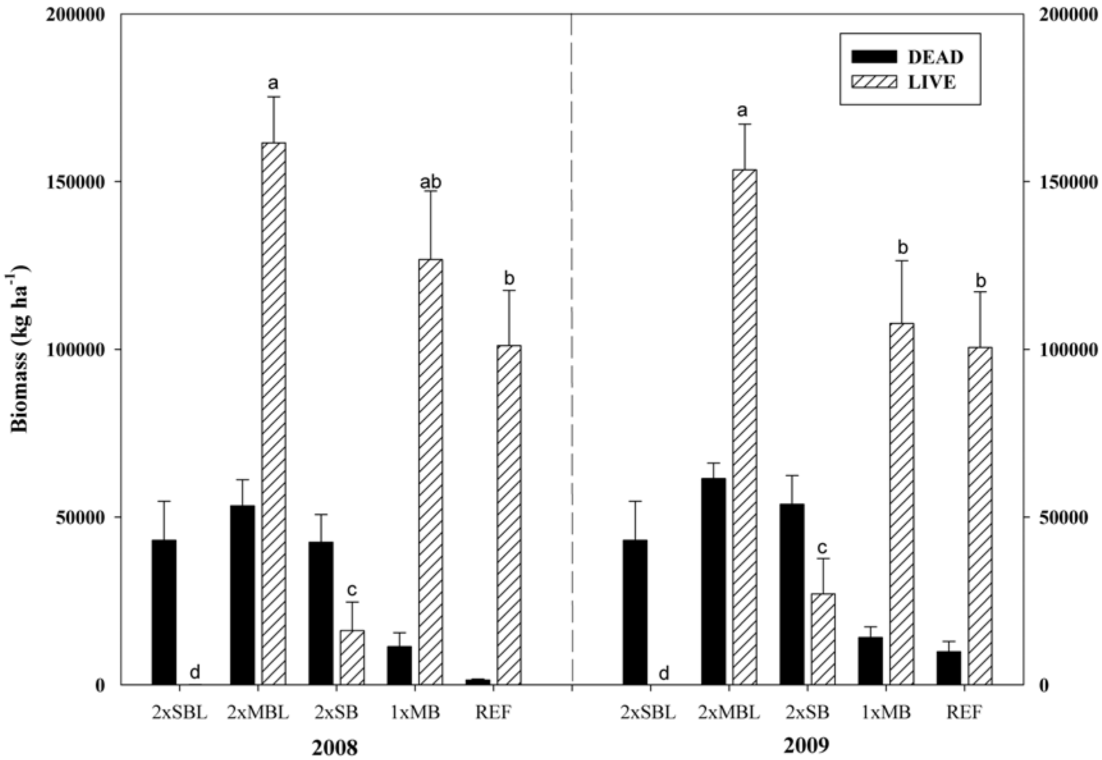

Figure 1.—Overstory (≥ 5.0 cm diameter at breast height [d.b.h.]) biomass with standard error bars of live and standing dead trees for the five treatment areas the first (2008) and second (2009) growing seasons after wildfire in Linville Gorge Wilderness, western North Carolina. The treatments were severely burned twice (2000 & 2007) plus dolomitic lime application (2xSBL); moderately burned twice with lime (2xMBL); severely burned twice, no lime (2xSB); moderately burned once (2002), no lime (1xMB); and an unburned and unlimed reference area (REF). Bars with different letters denote significant ($\alpha \leq 0.05$) differences among treatments within a year.

depth) and deep (10-30 cm depth) soils (Table 2). Soil exchangeable Ca^{2+} had a significant treatment effect, but no date or date x treatment interaction effects (Table 2). Soil Ca^{2+} was greater on 2xMBL and 2xSBL than 2xSB, 1xMB, and REF for shallow soil, and greater on 2xMBL than 2xSBL for deep soil (Table 2). Soil extractable aluminum (Al^{3+}) was less on 2xSBL than 2xSB, 1xMB, and REF, which resulted in higher Ca/Al ratios on 2xSBL and 2xMBL than 2xSB, 1xMB, and REF.

Wildfires are landscape-scale disturbances and have the potential to significantly impact biogeochemical processes by altering pools and fluxes of carbon and nutrients (Debano and others 1998, Knoepp and others 2005). The magnitude and duration of these responses depends on the interactions among preburn conditions, burn severity, postfire precipitation regime, topography, soil characteristics, and vegetative recovery rate (Robichaud 2005). We expected that the effects of the most recent wildfire and lime application would be observed over a 2-year period, possibly longer, as Ca and Mg leach into the soil where it can be taken up by the recovering vegetation (Elliott and others 2002, Knoepp and Swank 1997). In our study, the moderately burned site with lime application had the highest soil and foliar Ca suggesting that the lime application was subsequently taken up by vegetation.

In contrast, the severely burned site with lime had greater soil Ca but lower foliar Ca than the moderately burned site even though both sites received lime application. Soil Al was lowest on 2xSBL, suggesting that the lime addition did improve the soil Ca/Al ratio since it was higher on this site than the others. Al^{3+} increased and Ca/Al ratio decreased over time for both the shallow and deep soil depths (Table 2). Thus, the lime addition did improve the soil Ca/Al ratio, but the response was transitory. Soil Ca/Al ratios remain well below the toxicity threshold <1.0 (Cronon and Grigal 1995) on all treatment areas in LGW.

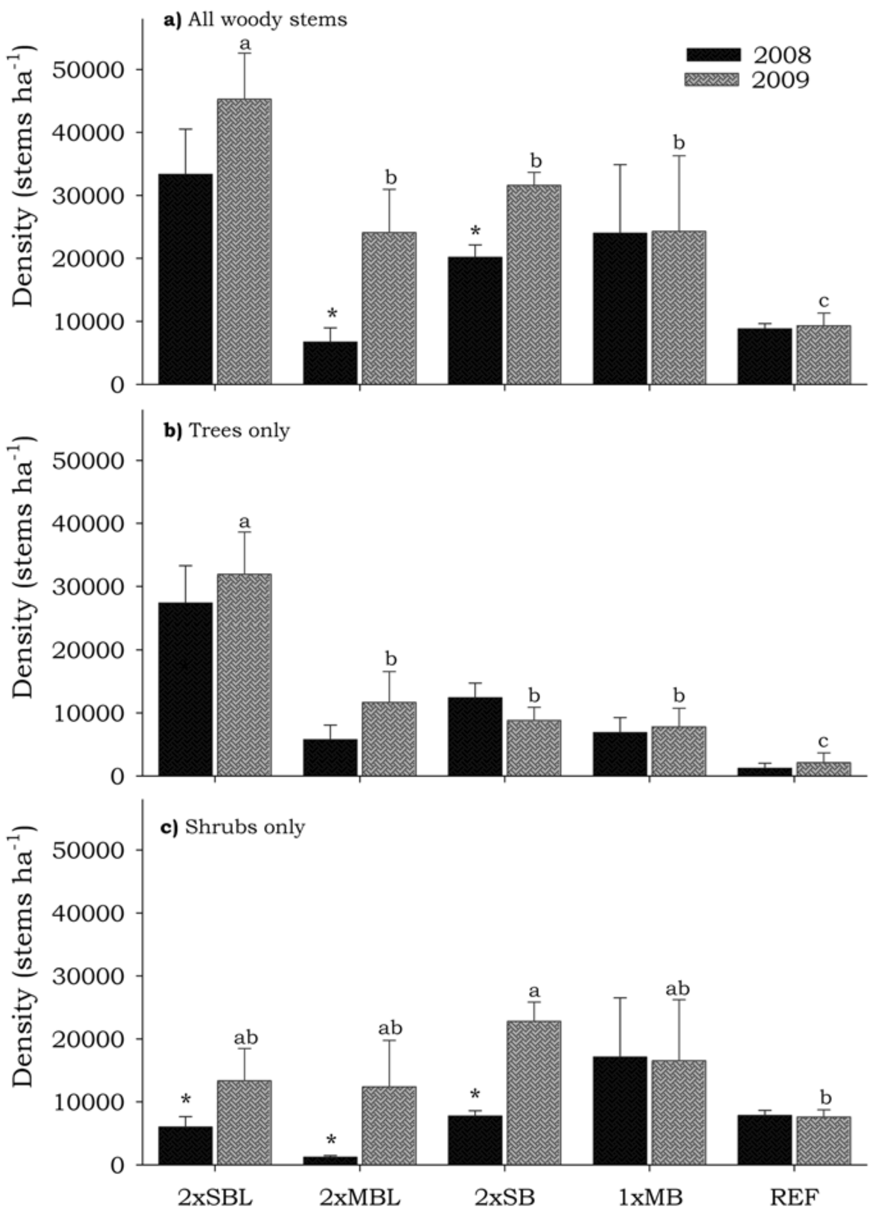

Figure 2.—Understory (shrubs and trees <5.0 cm d.b.h.) density with standard error bars: a) all woody species, b) trees only, and c)shrubs only for the five treatment areas the first (2008) and second (2009) growing seasons after wildfire in Linville Gorge Wilderness, western North Carolina. Treatment labels are the same as in Figure 1. Bars with different letters denote significant (α≤0.05) differences among treatments within a year. Asterisk denotes significant (α≤0.05) difference between years within a treatment.

Table 1.—Nutrient concentrations (with standard errors in parenthesis) of evergreen and deciduous leaf tissue for the five treatment areas: severely burned twice (2002 and 2007) plus dolomitic lime application (2xSBL); moderately burned twice plus lime (2xMBL); severely burned twice, no lime (2xSB); moderately burned once (2002), no lime (1xMB); and an unburned and unlimed reference area (REF).

	2xSBL	2xMBL	2xSB	1xMB	REF
N (percent)					
Evergreen	0.779 (0.032)	0.961 (0.021)	0.877 (0.032)	0.961 (0.064)	0.923 (0.052)
Deciduous	1.696 (0.067)	1.775 (0.132)	1.553 (0.059)	1.639 (0.057)	1.712 (0.042)
Ca (μg g^{-1})					
Evergreen	6687 b (373)	10214 a (1001)	7180 b (313)	8878 ab (646)	10163 a (506)
Deciduous	4927 b (470)	6697 a (158)	5503 ab (560)	5164 b (510)	5014 b (556)
P (μg g^{-1})					
Evergreen	736 (33)	780 (18)	707 (28)	670 (48)	739 (68)
Deciduous	1506 ab (36)	1612 a (53)	1445 ab (42)	1249 c (29)	1357 bc (24)
Al (μg g^{-1})					
Evergreen	52 (6)	59 (5)	47 (4)	169 (65)	193 (92)
Deciduous	91 b (30)	104 ab (21)	138 ab (9)	166 a (28)	84 b (11)

Notes: Values followed by different letters are significantly different ($p < 0.05$) among treatments (SAS 2002-2003).

Table 2.—Mean soil chemistry (with standard errors in parenthesis) for the five treatment areas in the Linville Gorge Wilderness, western North Carolina, USA. Treatment labels are the same as in Table 1. All values are in $cmol_c$ kg^{-1} except for Ca/Al molar ratio.

	2xSBL	2xMBL	2xSB	1xMB	REF
0-10 cm soil depth					
Nitrogen	0.107 b (0.021)	0.164 a (0.011)	0.143 ab (0.012)	0.145 ab (0.005)	0.139 ab (0.005)
NO_3–N	0.0009 (0.0004)	0.0003 (0.0003)	ND†	ND	ND
NH_{4+}-N	0.118 (0.021)	0.105 (0.022)	0.070 (0.005)	0.104 (0.011)	0.072 (0.005)
HPO_4^{2-}	0.236 ab (0.009)	0.250 a (0.015)	0.195 b (0.008)	0.241 ab (0.014)	0.251 a (0.010)
Ca^{2+}	7.322 ab (0.502)	9.616 a (1.662)	1.854 c (0.363)	4.009 b (0.630)	0.626 c (0.059)
Al^{3+}	76.36 c (6.234)	167.55 a (8.232)	124.00 b (6.404)	110.09 b (4.752)	109.40 b (7.700)
Ca/Al	0.102 a (0.006)	0.072 ab (0.017)	0.015 c (0.003)	0.037 bc (0.006)	0.006 c (0.001)
10-30 cm soil depth					
Nitrogen	0.046 b (0.009)	0.070 a (0.004)	0.078 a (0.006)	0.062 ab (0.003)	0.060 ab (0.003)
NO_3–N	ND	ND	ND	ND	ND
NH_{4+}-N	0.040 b (0.005)	0.044 b (0.003)	0.052 a (0.003)	0.053 a (0.003)	0.038 b (0.002)
HPO_4^{2-}	0.111 b (0.013)	0.142 ab (0.005)	0.143ab (0.008)	0.136 ab (0.009)	0.162 a (0.014)
Ca^{2+}	0.956 b (0.038)	1.519 a (0.251)	0.574 bc (0.146)	0.606 bc (0.063)	0.218 c (0.015)
Al^{3+}	57.83 c (4.982)	116.57 a (4.868)	87.55 b (6.530)	76.42 bc (3.719)	81.11 b (2.747)
Ca/Al	0.018 a (0.001)	0.016 a (0.003)	0.006 bc (0.001)	0.009 b (0.001)	0.003 c (0.0002)

†ND = samples below the detection limits of our methods. Values within a soil depth followed by different letters are significantly different ($p < 0.05$) among sites (SAS 2002-2003). Values were averaged across time; and then, mean values and standard errors were based on the five plots per treatment area ($n = 5$).

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COMMUNITIES AND DENSITY ESTIMATES OF HISTORICAL AND CURRENT MISSOURI OZARKS FORESTS

Brice B. Hanberry, Hong S. He, Daniel C. Dey, John M. Kabrick, and Brian J. Palik

Research Associate (BBH) and Associate Professor (HSH), Department of Forestry,
203 Natural Resource Building, University of Missouri, Columbia, MO, 65211;
Research Forester (DCD, JMK, BJP). BBH is corresponding author:
to contact, call 573-875-5341 ext. 230 or email at hanberryb@missouri.edu.

INTRODUCTION

Forests in Missouri changed after European settlement, due in part to timber harvest, fire suppression, grazing, and conversion to agricultural and urban land use. There are few current examples of fire-mediated composition and structure present in oak and pine savannas and woodlands. Reconstruction of historical landscapes can provide a reference for restoration efforts.

METHODS

The United States General Land Office (GLO) surveyed trees during 1815-1864 in the Missouri Ozarks. The U.S. Forest Service's Forest Inventory and Analysis (FIA) program surveys current forests. Using data from 2004-2008, we selected live trees with a diameter at breast height (d.b.h) ≥ 12.7 cm. Our ecological unit was a subsection divided into land types (ECOMAP 1993).

For community assignment in an ecological unit, the number of trees had to be ≥200 per ecological unit, and percent composition of species had to be ≥10 per ecological unit to be a dominant species in the community. Community order was based on descending mean percent composition for all GLO trees.

For density estimation in an ecological unit, FIA plots had to be 100 percent forest land. For historical forests, we used the Morisita plotless density estimator and applied corrections for surveyor bias. For current forests, we calculated trees per acre using the expansion factor of 6.018046, based on one tree representing the inverse of the plot area in acres (i.e., $1/[4*0.04154172]$). To predict densities continuously for 835,000 Soil Survey Geographic (SSURGO) Database polygons (mean area of 10 ha), we used calculated plot density, 24 soil and topographic predictor variables, and random forests, an ensemble regression tree method.

RESULTS AND DISCUSSION

Across the Missouri Ozarks, there has been a decrease in oaks from 79 percent of species composition to 57 percent of species composition. This has resulted in changes in communities. Dominant oak species (as defined by ≥ 10 percent composition) have been reduced and replaced by a mixture of many species, of which only eastern redcedar and maples have become dominant. Current forest densities are approximately two times greater than historical densities. Current forests are about 300 to 400 trees/ha (d.b.h. ≥ 12.7 cm), with little deviation. Historical forests had greater deviation and ranged from a mean of 75 to 320 trees per ha. The greatest density increases were in the drier western Ozarks close to the prairies, where historical densities were lower.

Although climate change, insects, fungal pathogens, and deer herbivory probably all contribute to oak declines, fire appears to be the primary factor that determines community composition and structure. Regionally distinct, oak-dominated and fire-dependent communities have been replaced by denser, fire-intolerant species. Currently, many oak-dominated forests have understories of shade-tolerant species, and in the future, upland eastern forests may become duplicates of mesic forests without management. Oak restoration will become more difficult if the overstory is no longer oak. Oak forest management and research priorities include commitment to both prescribed burns and opening the canopy and subcanopy where there is an oak understory.

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RESTORING OAK REGENERATION AFTER OVERGRAZING AND FIRE IN ZAGROS FORESTS

Azad Henareh Khalyani, Audrey L. Mayer, and Manouchehr Namiranian

Research Assistant (AHK) and Assistant Professor (ALM), School of Forest Resources and Environmental Science, Michigan Technological University, 1400 Townsend Dr., Houghton, MI, 49931; Professor (MN). AHK is corresponding author: to contact, call 906-231-1186 or email at ahenareh@mtu.edu.

INTRODUCTION

Zagros oak forests have a long history of use by local communities. Palynological research provides evidence of stock breeding, grazing, and agriculture at least since the beginning of the fifth millennium cal B.P. (Djamali and others 2009, Sumner 1990, Wright and others 1967). The traditional sustainable management was maintained by a homeorhetic regulation in pre-technological land use (Naveh 1988). This regulation was acquired by a balance between the anthropogenic utilization of the forests for livelihood necessities and stabilizing feedbacks from a lack of food, water, and social components, generating a resource protection mechanism. Resource utilization activities such as coppicing and cultivation were practiced by rotations (Ghazanfari and others 2004). Landscapes maintained a patchwork formed by the combined effects and interaction of fire, grazing, and coppicing.

However, contemporary technological progress and maladaptive policies have eliminated the economical feedbacks from population control by providing food, water, and medicine for man and livestock regardless of water supply, even though the lack of economical development should provide economic incentives to reduce pressures on the forests. The social stabilizing feedback was eliminated by a transition to governmental ownership and new regulations. Forests were nationalized in 1963, and grazing and coppicing became illegal. The Iranian government has tried to protect the forest (and encourage regeneration) from livestock grazing through the use of various exclosures, but these have not been effective in prohibiting overutilization in all areas because of the economical needs of locals. In this situation, the landscapes are no longer formed by the combined effects and interaction of fire, grazing, and coppicing because these influential factors are separated from each other in the landscape and have led to the formation of the two extreme models in Zagros landscapes: with and without exclosures. In areas without government exclosures, there is no or very limited fire frequency because of the quick consumption of understory grasslands by livestock. But overgrazing by livestock (mainly goats) leads to the lack of regeneration with consequent soil erosion. The additional pressure from climate change-induced droughts further catalyzes the transition of the forest landscapes to deserts. In areas with government exclosures that restrict access of people and livestock, the forest cover is frequently affected by fires during the dry season. Forest stands cannot recover quickly from the wildfires which devastate the ecological condition because of the dry and hot conditions in summer. If the fires are too frequent, they also reduce fauna and flora and inhibit regeneration through topsoil destruction.

The social and economical feedbacks, therefore, need to be restored in order to reactivate the dynamic self-stabilization and maintain ecological sustainability in patchy landscapes formed by the interaction of anthropogenic uses and fire. Restoration projects thus need to be carefully targeted, socially accepted, and implemented by direct participation of local communities using their traditional ecological wisdom. In this study we implement an adaptive management approach to: (1) gradually eliminate the boundaries between the two extreme conditions by reintroducing livestock grazing to avoid frequent extreme wildfires; (2) establish seed based regeneration in the context of grazing and coppicing by local communities; (3) restore the social feedbacks by giving access and utilization rights to local communities and use their participation in the project; and (4) evaluate the potential of the sites for regeneration establishment and compare different technical approaches for the regeneration establishment in the area.

STUDY AREA

The study area was 15 ha of Armardeh forests (45°48' N, 35°56' E) in the northern Zagros region in Baneh city, Kurdistan province, Iran (Fig. 1). These xeric forests are dominated by *Quercus* spp. with other deciduous species occurring occasionally. Forest cover rarely exceeds 70 percent. The average elevation is 1675 m a.s.l., and average annual precipitation is 615 mm. The bedrock is mainly formed by quartzite. The soil type is mainly formed of inceptisols and in some areas is heavily destructed and forms entisols.

METHODS

Thirty 1000 m^2 circular plots were randomly located in the study area. The height, diameter at breast height (d.b.h.), and crown diameter at two directions were measured for all trees in the circular plots (Fig. 1). One soil profile was dug in each plot, and soil physicochemical characteristics were analyzed. Four restoration techniques were established in each of

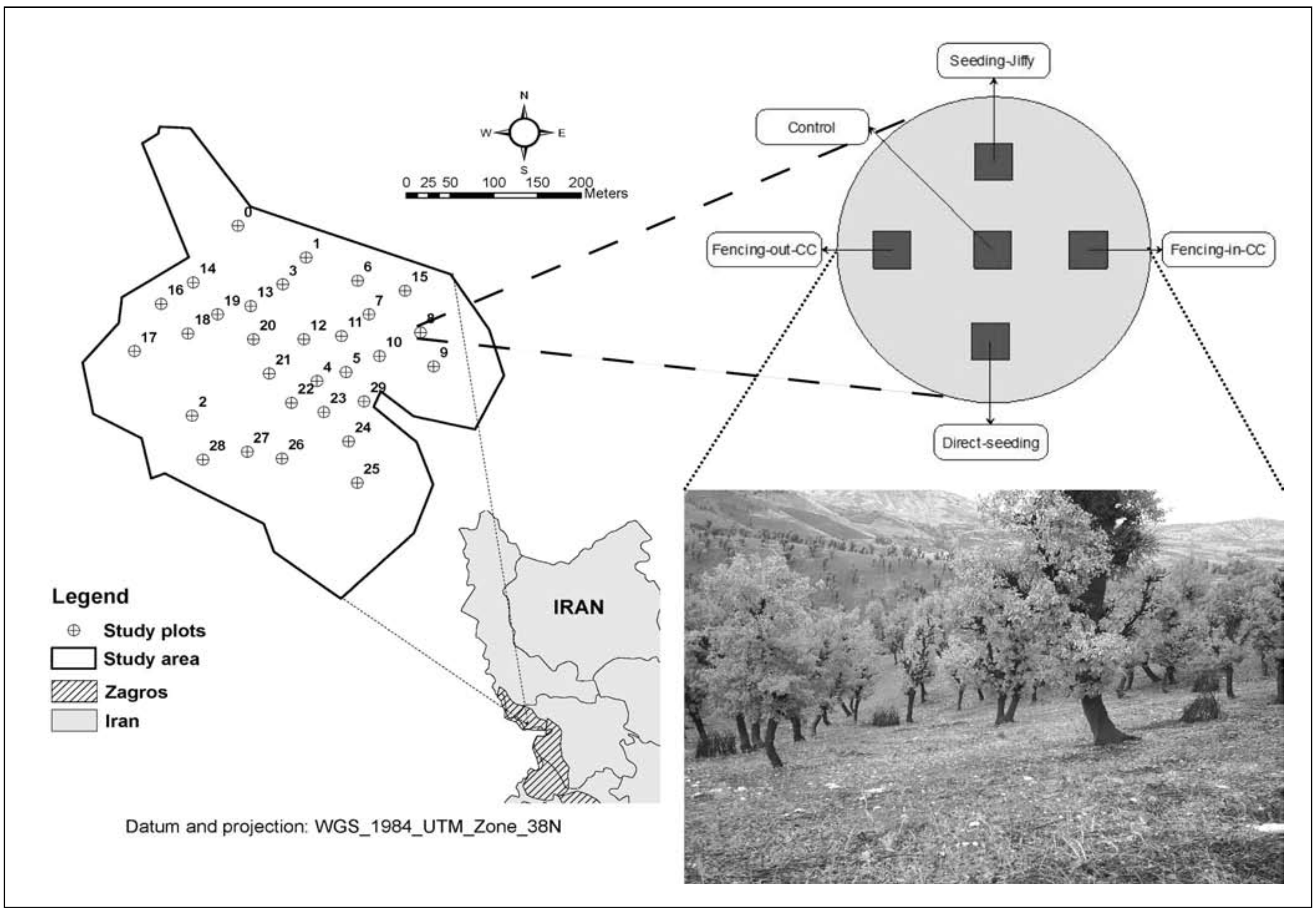

Figure 1.—Location of study area in Iran, position of the 1000 m^2 circular plots in the study area, and the five 1 m^2 micro plots in each circular plot.

the circle plots as study treatments in 1 m^2 micro plots, along with a control micro plot at the center of the plots in autumn. The study was conducted in the framework of a randomized complete block design (RCBD). The restoration approaches (treatment micro plots) were:

1. Fencing outside the crown coverage
2. Fencing under the crown coverage
3. Direct seeding outside the crown coverage
4. Seeding using Jiffy technique (Skoupy and Hughes 1971) outside the crown coverage

The treatment micro plots were fenced, and the control micro plots were left unfenced under the crown coverage of each plot's center tree (Fig. 1). The response variables were the proportion of established seedlings and the heights of the saplings in the spring. Analysis of Variance (ANOVA) and Tukey's post hoc test were used to test the difference between treatments at 0.05 significance level.

RESULTS

The quantitative measurements showed that 74.1 percent of the trees in the study area were Lebanon oak (*Quercus libani* Olivier). Gall oak (*Quercus infectoria* Olivier), Mann oak (*Quercus brantii* L.), pistachio (*Pistacia atlantica* Desf.), and wild pear (*Pyrus glabra* Boiss.) formed 24.8 percent, 0.6 percent, 0.3 percent, and 0.2 percent of the trees, respectively. Mean d.b.h., height of the trees, and canopy cover were 29 cm, 6.6 m, and 1314 m^2/ha, respectively. Aspects did not show significant effects on tree characteristics and treatment effects. Physicochemical analysis of soil characteristics in two layers showed that in spite of being mechanically compacted and destructed, it had proper physicochemical characteristics for regeneration (Henareh 2005, Namiranian and others 2007). There was a significant difference in restoration success between fenced and unfenced study plots (Fig. 2). Areas in exclosures demonstrated regeneration while controls with no fencing did not. Planting with Jiffy pellets resulted in significantly taller saplings compared to other approaches (Fig. 3). Direct seeding treatments were significantly lower in height than planting Jiffy pellets but taller than other treatments.

DISCUSSION

Higher percentage of Lebanon oak indicates forest stands with better ecological conditions, since this species requires higher humidity and deeper soils compared to other oak species. Forest stand measurements and soil analysis indicated that the potential for natural regeneration still exists. Applying exclosures over large areas increases the understory

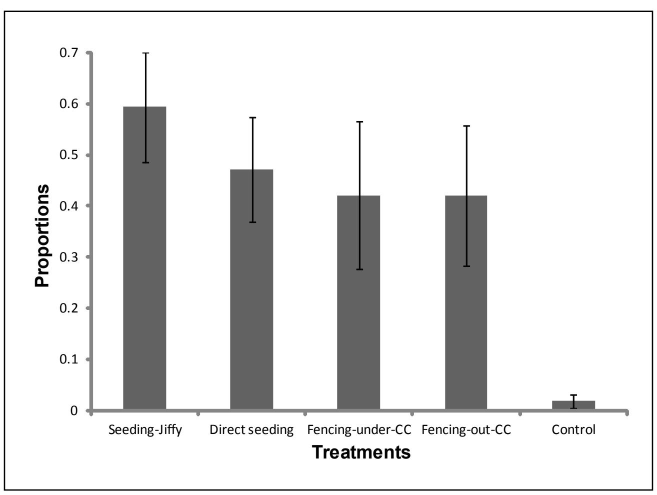

Figure 2.—Mean proportion of established seedlings for the five treatments in early spring (error bars represent standard errors).

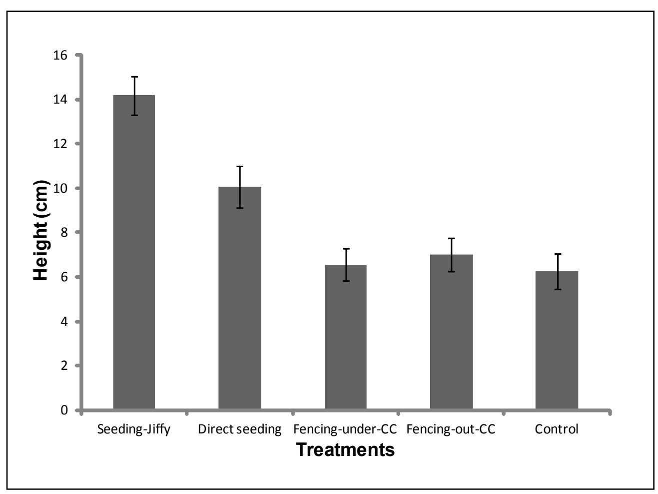

Figure 3.—Mean heights of saplings for the five treatments in summer (error bars represent standard errors).

dry vegetation and risk of extreme wildfires as well as social problems by forcing local communities out of forests. However, the very small exclosures tested here do not suffer from either of these problems and demonstrated that regeneration in a context of grazing and coppicing is feasible. Fencing at a small scale (1 m^2), even outside of the crown coverage and without applying artificial restoration techniques, ensured the establishment of regeneration significantly higher than in unfenced areas, where no regeneration occurred. No significant difference was found between natural and artificial restoration methods regarding the proportion of established seedlings. This suggests that the simple treatment of building a small fence around small, scattered areas is a low-cost, practical management approach that local communities can use to help restore forest regeneration into these landscapes. However, a significant difference was found between artificial and natural methods regarding the heights of the resultant saplings in the summer; the artificial methods resulted in taller samplings. These artificial methods may be a necessary addition in areas that have lost a significant amount of topsoil and where natural regeneration might grow too slow relative to the need for the resource base that the forests represent.

The study demonstrated that conservation at small scales in a context of livestock grazing and coppicing by local communities can result in forest regeneration and fire avoidance, while simultaneously being acceptable to local communities. Applied across the region, this approach could eliminate the boundary between the two dominant extreme conditions of fire and grazing and maintain patchy landscapes formed by the interaction of all factors including fire, grazing, and coppicing.

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INITIATING RESTORATION OF WOODLAND COMMUNITIES: THE EFFECT OF THINNING AND PRESCRIBED FIRE ON STAND STRUCTURE IN THE OZARK HIGHLANDS

Carter O. Kinkead, John M. Kabrick, Michael C. Stambaugh, and Keith W. Grabner

Graduate Research Assistant (COK); Research Forester (JMK), U.S. Forest Service, Northern Research Station, 202 Natural Resources Building, Columbia, MO 65211; Research Associate (MCS); Ecologist (KWG). JMK is corresponding author: to contact, call 573-875-5341 ext. 229 or email at jkabrick@fs.fed.us.

INTRODUCTION

In the last two centuries, Missouri Ozark forests have undergone immense transformation as a result of timber harvests and fire suppression (Dey and Hartman 2005, Ladd 1991). As fuel accumulation and canopy closure have increased (Kolaks 2004), so has interest in restoring and maintaining the woodland and savanna ecosystems that once prevailed across the Ozark landscape (Johnson and others 2002, Ladd 1991, Nuzzo 1986). Woodland communities are characterized by open understories and dense ground flora comprised of forbs, grasses, and sedges (Nelson 2005, Nuzzo 1986, 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). With fire-based management making its way to the forefront of this effort, a number of science-based efforts are aimed at quantifying the response of natural systems to various fire or disturbance regimes.

The purpose of this study was to gain insight as to the implications of various treatments, from prescribed burning to thinning harvests and the combination. Our objective was to examine changes in forest structure, such as diameter distribution, canopy cover, and species composition, as a function of fire, thinning, and aspect to better understand how to restore oak and oak-pine woodlands.

The project area is located in the Missouri Ozarks at Logan Creek and Clearwater Creek Conservation Areas, both managed by the Missouri Department of Conservation. Each site lies within the Black River Oak/Pine Woodland/Forest Hills Ecological Landtype Association that is characterized by steep hillslopes with cherty, low-base soils (Nigh and Schroeder 2002). The fully stocked, second-growth forests on each site have had no known management or fire for over 40 years.

METHODS

Four treatments (burn, harvest, harvest-burn, and control) were applied to three (north slope, ridge, and south slope) ecological landtypes (ELTs) across three replicated blocks. Treatments were paired with the ELT to create 12 experimental units per block. Timber harvests (overstory thinning) occurred during the summer and early fall 2002 prior to the first burn. Harvests reduced stocking to approximately 45 ft^2/acre by thinning from below, leaving primarily the dominant trees with preference for retaining shortleaf pine (*Pinus echinata* Mill.) and white oak (*Quercus alba* L.). We use the term "thinning" because the goal was not to regenerate the stand but rather to enhance the growth of the residual stand including the forbs, legumes, and graminoids in the understory. Two prescribed fires have been conducted on burn and harvest-burn units in each of the three blocks. The first occurred in early April of 2003 and the second in April 2005 for blocks one and two. Because of unsuitable weather conditions, block three was not burned until March 2006. Consequently it was excluded from analysis.

Pretreatment data were collected in the summer of 2001, and posttreatment data were collected in the summer of 2003 and 2005. Permanent vegetation plots were installed along transects during the summer of 2001. Overstory vegetation plots consisted of three 1/3 acre circular plots randomly located within each treatment/aspect class. All live trees (diameter at breast height [d.b.h.] > 1.5 inch) were identified to species and d.b.h. was measured. Ground flora data were collected at a total of 1,080 quadrats (size = 10.8 ft^2, 30 per experimental unit). Within each quadrat, all live herbaceous plants and shrubs were identified to species, and cover by species was estimated to the nearest percent.

RESULTS AND DISCUSSION

Average pretreatment stocking for all stands was 94.2 percent, composed primarily of trees greater than 10.5 inches d.b.h., even after treatments. Prescribed fire alone decreased overall stocking nearly 10 percent from pretreatment conditions, having significant reductions in all trees less than 10 inches. In fact, 2-5 inch stems declined 22 percent after a single fire, and 52 percent after a second fire. However, due to the minimal impacts on larger diameter trees, the effects of prescribed fire to overall stocking were not significantly different from the control. No significant ($P < 0.05$) interaction between aspect and treatment on stocking or ground flora coverage was detected.

The effects of harvest-burn treatments on stocking did not significantly differ from harvest only treatments (Fig. 1). These findings indicate that prescribed fire has the ability to decrease the abundance of small diameter stems without drastically altering stand structure. From a woodland restoration standpoint, it is desirable to decrease the abundance of understory woody plants to provide more growing space for the herbaceous plants. In the absence of fire, hardwood seedlings and seedling sprouts proliferate, which can shade herbaceous plants and reduce their coverage (Hutchinson and others 2005). This is especially the case in the presence of increased canopy openness, such as after a harvest. Although our results show a minor decrease in woody stems one year after harvest-burn treatments, preliminary results from recently collected data (2011) suggests that without continued burning, small trees and shrubs begin to limit all other life forms in the understory.

Life forms documented in the quadrats include forbs, graminoids, legumes, shrubs, and vines. All of these physiognomic groups increased with both prescribed fire and the combination of burning and harvesting. Since many restoration efforts are aimed at increasing forb and graminoid communities, it is important to note that harvest-burn treatments more than doubled the percent coverage of these life forms compared

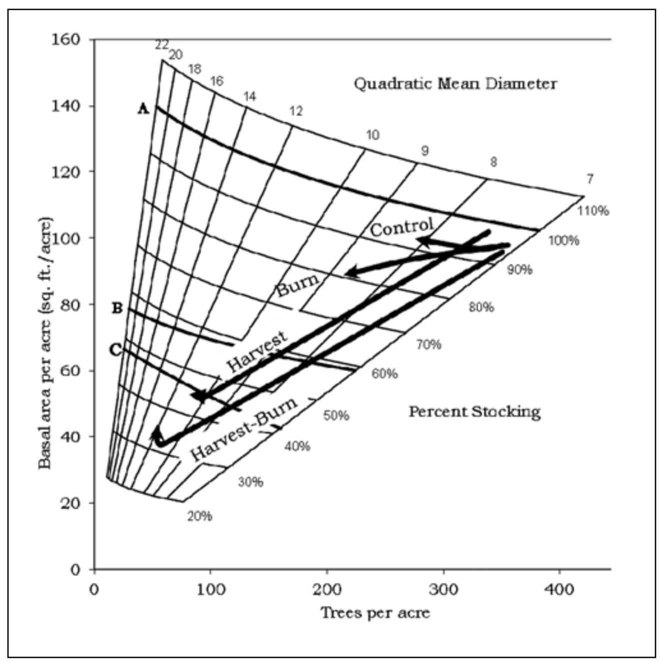

Figure 1.—Gingrich (1967) diagram portraying changes in stocking for each treatment from 2001 to 2006. Actual stocking percentages are slightly misrepresented because the parameters used are based on "typical upland oak-hickory" stands rather than calibrated to the quadratic mean diameter of our study site (Johnson and others 2002).

to burning alone and harvesting alone. Analyses of a class of woodland indicator species (i.e., a collective group of forbs, graminoids, legumes, and shrubs) (Table 1) showed this class increased 42 percent in relative cover on plots that were harvested and burned, a near three- to fourfold increase compared to burn only and harvest only treatments (Fig. 2). Woodland indicators within control plots increased 2 percent.

Overall, the results of this study suggest that commercial overstory thinning followed by two prescribed fires has the potential to restore woodland structure and composition (Nelson 2005, Taft 2009). Our findings also demonstrate the relationships between stand structural metrics such as stocking to woodland ground flora and shrub cover in the presence or absence of prescribed fire. These relationships have seldom been quantified and documented in the woodland restoration literature. This sequence of management techniques most effectively resulted in stands characteristic of woodland conditions (i.e., basal area 30-50 ft^2/acre, abundant forb and grass populations, and minimal midstory tree density [Nelson 2005]). While mechanical removal of the midstory and overstory trees fulfills these restoration objectives, increased burning becomes necessary to manage the understory woody response to canopy openness. Prescribed fire alone may pose a sustainable option for woodland restoration where overstory stocking is decreased over time, thereby subtly achieving canopy openness and suppression of woody regeneration.

Table 1.—List of woodland indicator species used in understory vegetation sampling

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

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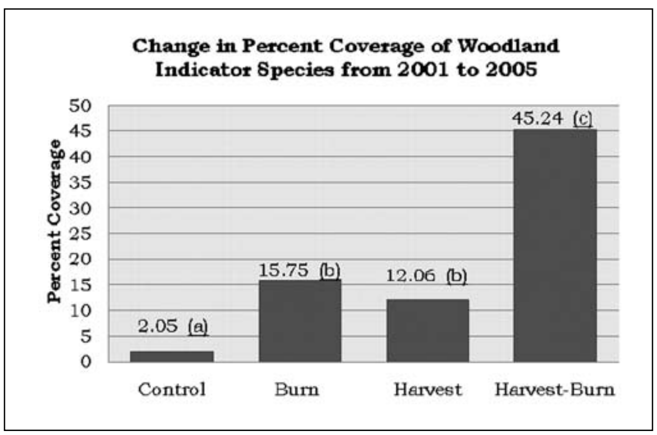

Figure 2.—Changes in percent coverage of woodland indicator species (see Table 1) for all treatments. Significant differences ($P < 0.05$) among treatments are indicated with differing letters.

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EFFECTS OF MULTIPLE INTENSE DISTURBANCES AT MANLEY WOODS, WILSON'S CREEK NATIONAL BATTLEFIELD

Sherry A. Leis and Kevin James

Fire Ecologist (SAL), Biology Department, Missouri State University, 901 South National Avenue, Springfield, MO 65897; Plant Ecologist (KJ). SAL is corresponding author: to contact, call 417-836-8919 or email at sleis@partner.nps.gov.

ABSTRACT

Natural communities in southwest Missouri are often challenged by ecological disturbances of a catastrophic nature. Windthrow, wildfire, flooding, drought, and ice storm events all occur periodically and range in spatial extent from local to regional. Although in this region local disturbances are common in singularity, it is unusual for several disturbances to affect the same locality within a short time period. Since 2003, however, the Ozark Highlands region has seen widespread events including tornados, fires, floods, and ice storms, many occurring in succession on the same piece of land. Understanding the change in vegetation composition and recovery rates resulting from multiple severe disturbances is crucial to natural resource management planning.

In 2003, a tornado struck the Manley Woods area of Wilson's Creek National Battlefield (WICR) affecting 60 ha. The affected area included long-term monitoring sites sampled both 3 years prior to and post-tornado. In the post-tornado years, three additional notable disturbances occurred, but each disturbance differentially affected each of the sites. Increased fuel loads resulting from the tornado damage caused concern for elevated wildfire risk, so park staff made plans to reduce fuel loads. In 2005, a contractor removed salvageable timber from Manley Woods, and the following year a prescribed fire was completed. In 2007 a January ice storm further affected communities in Manley Woods. The aim of this study was to determine the effectiveness of the fuels reduction activities, and to describe the composition of overstory and understory plant communities currently inhabiting Manley Woods.

Fuel loads at Manley Woods were quite high post-tornado, especially at site four (Fig. 1). The fuels reduction process, log removal, and fire reduced mean fuel loads by approximately half, leaving an average of 18.5 tons/ha. Mean fuel loads were significantly reduced at site 4 that received the most damage and notable reductions were seen at site 5.

Changes occurred in several aspects of the plant community. Overstory tree density was reduced by 4.31 m^2 per ha, mainly by the tornado (Fig. 2). In 2006, trees in the larger size classes (35+ cm diameter at breast height [d.b.h.]) could only be found at the less disturbed sites in the woods, and canopy closure trended towards a savannah structure. Regeneration (seedlings and saplings) was reduced overall, including a significant decline in oak and hickory species. Diversity of the herbaceous understory plants dropped significantly during the period of heaviest disturbance (2003-2006). Understory plant community response lagged 1 year behind disturbance events (Fig. 3). Plant

community composition shifted towards more early seral woodland species after 2004 at all sites, but the greatest change occurred at the most heavily disturbed areas in the woods. Furthermore, 2007 composition appeared to be on a path back toward previous community composition. Plant guilds, particularly woody plants, pulsed after the tornado but returned to similar predisturbance conditions. Exotic species remained low and did not change significantly over time.

This analysis may aid park natural resource managers to plan to achieve the desired conditions in Manley Woods. Prior to the 2003 tornado, park managers used fire to work towards a savannah community type. The disturbances had a heterogeneous effect on the whole management unit, hastening the restoration process in parts of Manley Woods, but not greatly affecting others.

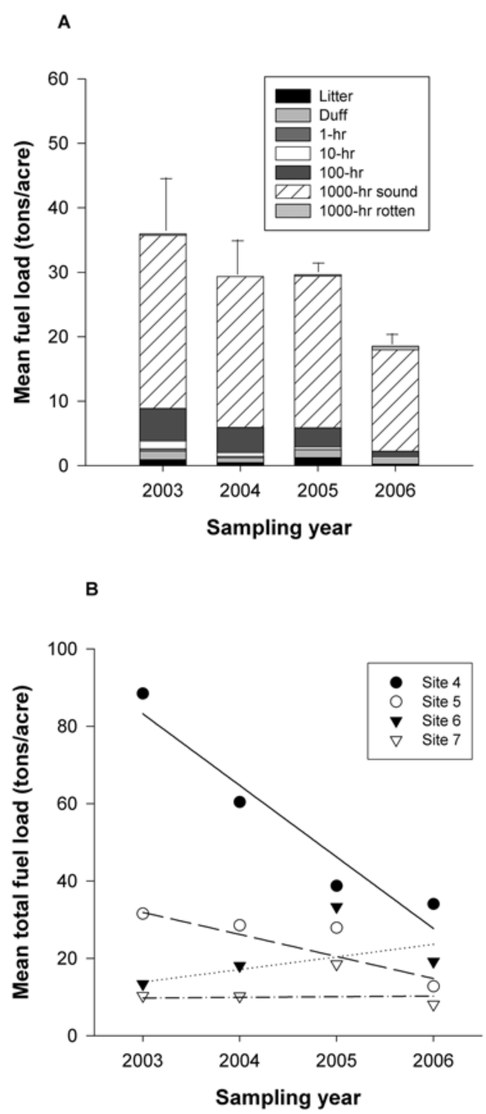

Figure 1.—(A) Mean fuel loading by time lag class for Manley Woods. Error bars represent one standard deviation for mean total fuel load (not time lag class), (B) mean fuel loading by sampling site for Manley Woods. Asterisk indicates loadings at site 4 were significantly different through time ($P = 0.04$, $r^2 = 0.92$).

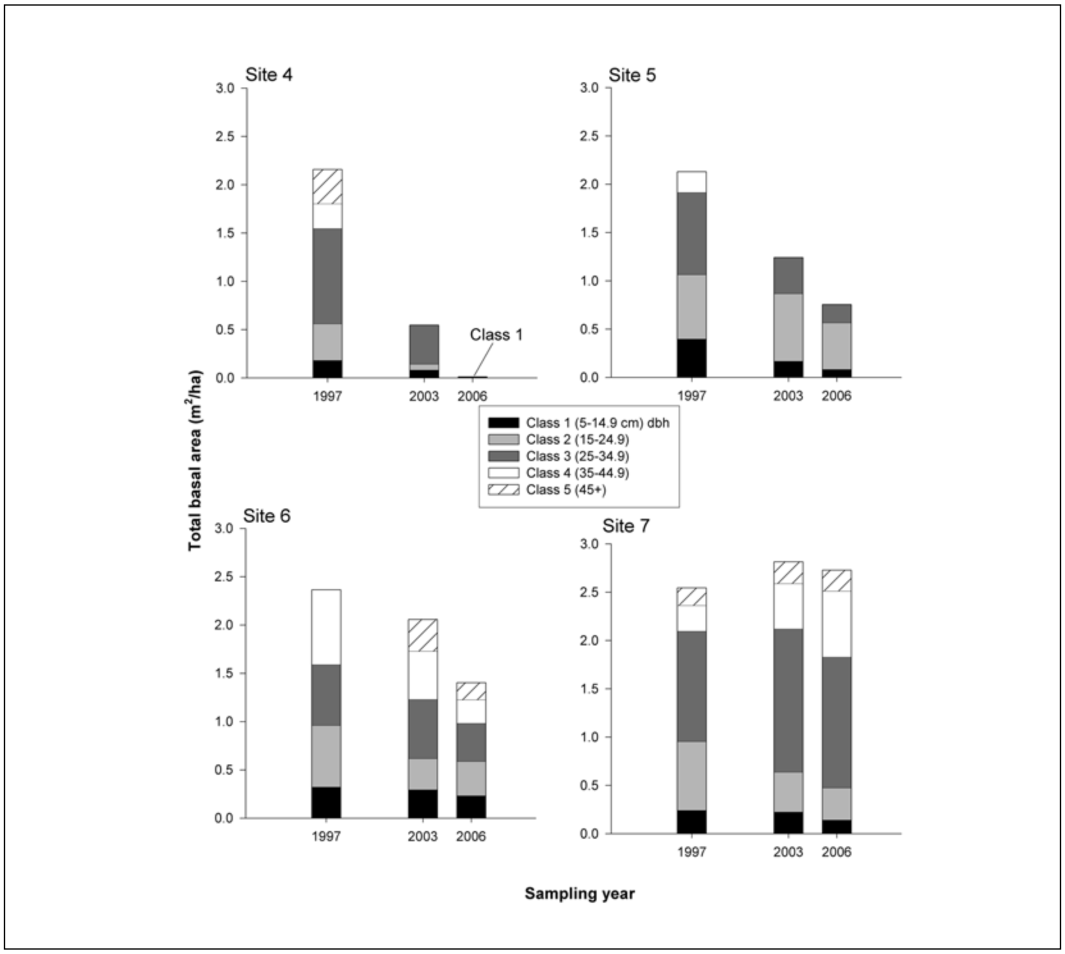

Figure 2.—Total basal area (m^2/ha) of trees in each size class throughout monitoring history at Manley Woods, (diameter at breast height = d.b.h.).

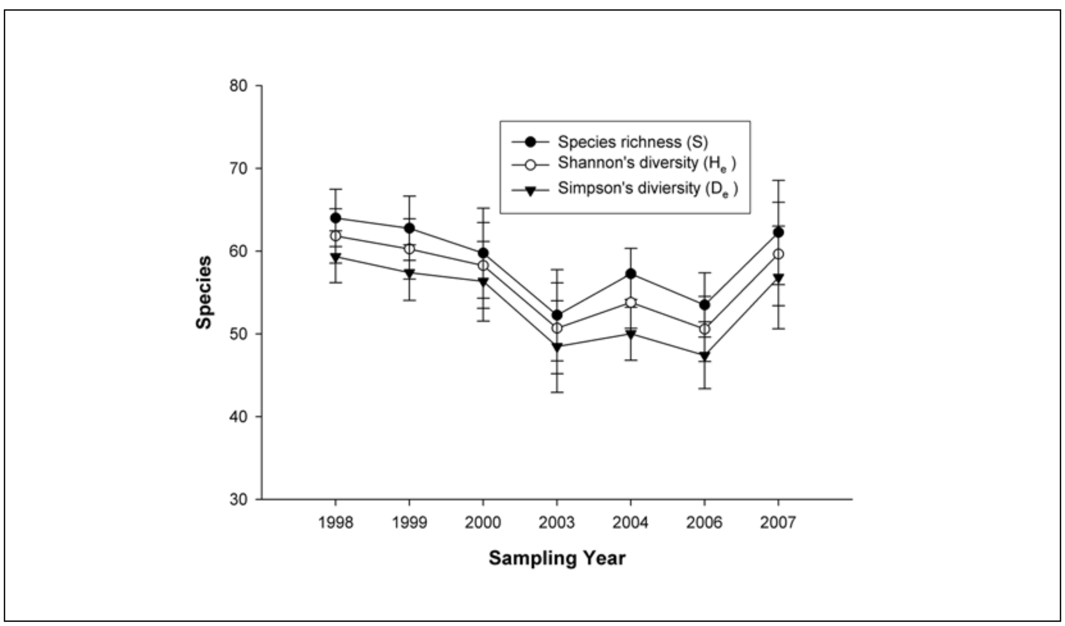

Figure 3.—Mean diversity measures with standard error bars for the understory community in Manley Woods long-term monitoring plots. Measures were converted to effective number of species for comparison.

POTENTIAL EFFECTS OF FIRE AND TIMBER HARVEST ON TERRESTRIAL SALAMANDER MICROHABITAT USE IN A MISSOURI OZARK FOREST

Katherine M. O'Donnell, Frank R. Thompson III, and Raymond D. Semlitsch

Graduate Student Researcher (KMO), Division of Biological Sciences, University of Missouri, 105 Tucker Hall, Columbia, MO 65211-7400; Research Wildlife Biologist (FRT); Curators Professor (RDS). KMO is corresponding author: to contact, call 573-882-1421 or email at kmozrc@mail.missouri.edu.

BACKGROUND

Prescribed fire and timber harvest are human-caused disturbances that can have substantial effects on forest ecosystems. It is generally accepted that some methods of timber harvest adversely affect forest-dependent wildlife, including amphibians (e.g., Petranka and others 1993, Semlitsch and others 2009). However, wildlife responses to fire are not as well understood; they vary considerably among species, and are affected by fire seasonality and severity (Lyon and others 2000). As the use of prescribed fire becomes more common, it is essential to better understand the effects of controlled burns on forest-dependent wildlife.

Fire and Amphibians

Fire effects on amphibians remain incompletely understood but likely vary among species, geographic regions, and habitat types (Pilliod and others 2003). Amphibians require wet or moist places for breeding, nesting, and respiration, and are very susceptible to heat and desiccation. Therefore, they often retreat under cover objects such as rocks and logs to retain moisture when the leaf litter is dry. Amphibians have limited movement capacity (Russell and others 1999), which likely forces them to stay and cope with habitat changes postfire. This unique suite of characteristics may make amphibians more vulnerable to fire than other wildlife taxa.

Recent studies have found little evidence of amphibians being directly affected by prescribed fire (burns, mortality from fire) in the short-term (Pilliod and others 2003, Russell and others 1999). However, fire can indirectly affect wildlife by altering components of their habitat. These indirect effects are likely key for amphibians because of their sensitivity to changes in microhabitat, such as leaf litter and downed wood, and could be long-lasting (Pilliod and others 2003).

Focal Species

We chose to focus our study on the southern redback salamander (*Plethodon serratus*) (Fig. 1). Redback salamanders and other species in the Plethodontidae family are fully terrestrial; they do not have an

Figure 1.—Southern redback salamander (*Plethodon serratus*).

aquatic larval stage like many other amphibians. They undergo direct development resulting in hatchlings that resemble miniature adults. Plethodontid salamanders are also lungless and respire directly through their skin, and therefore, are very moisture-dependent. These salamanders typically have very small home ranges. Previous studies have shown that the southern redbacks are most surface-active in the spring and fall when temperatures are moderate (Herbeck and Semlitsch 2000). They spend most of the summer months in underground refugia to avoid desiccation and heat stress. They are also nocturnal and come to the surface at night to forage. During the day, they usually retreat underground or under cover objects.

Importance

Terrestrial salamanders like the southern redback are the dominant vertebrate predators in many forest ecosystems and represent a large portion of the vertebrate biomass (Burton and Likens 1975), although they are often overlooked because of their fossorial nature. Several studies suggest that these types of salamanders may play an integral role in forest ecosystem dynamics, such as nutrient cycling, leaf litter decomposition, and even carbon sequestration (Burton and Likens 1975, Welsh and Droege 2001, Wyman 1998). Disturbance events such as timber harvest cause drying of the soil and leaf litter and make forests less capable of supporting amphibians (Semlitsch and others 2009, Welsh and Droege 2001). Because prescribed fires also affect leaf litter and cover objects, there may be negative consequences for amphibians. We are investigating the effects of prescribed fire and timber harvest on terrestrial salamanders. We are interested in how these forestry practices affect salamander abundance and behavior. This paper focuses on the predicted outcomes of the study, based on pretreatment data, on the distribution of salamanders among microhabitats.

STUDY AREA AND METHODS

The study is taking place in the Sinkin Experimental Forest, located within the Mark Twain National Forest in the Ozark Highlands region of south-central Missouri. The site consists of mature, fully-stocked oak-hickory stands. Five 5-ha replicate experimental plots per treatment will be subjected to one of the following treatments: prescription burn, shelterwood harvest, or midstory herbicide. Five plots will serve as controls. We sampled two locations (upslope, downslope) within each of the 20 sampling plots in the spring and fall of 2010. We conducted 3 x 3 m area-constrained searches of natural cover objects and leaf litter, and measured snout-vent length (SVL) and recorded the capture location of all salamanders (southern redback salamander, western slimy salamander [*Plethodon albagula*], and dark-sided salamander [*Eurycea longicauda melanopleura*]).

RESULTS

We captured 1025 southern redback salamanders, 18 western slimy salamanders, and 4 dark-sided salamanders. We found most salamanders (74 percent) within the leaf litter; the remaining 25 percent were found under natural cover objects such as logs and rocks (Fig. 2). We captured more salamanders during downslope surveys (566) than upslope surveys (481). Salamander density was strongly correlated with recent rainfall (Fig. 3). We also found no significant pretreatment differences between groups.

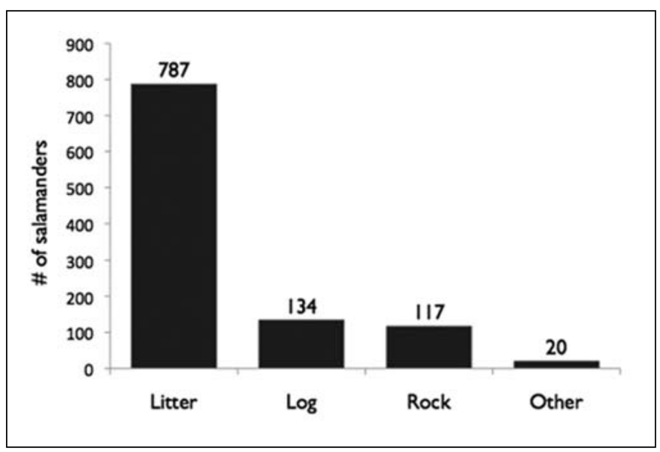

Figure 2.—Location of salamander captures.

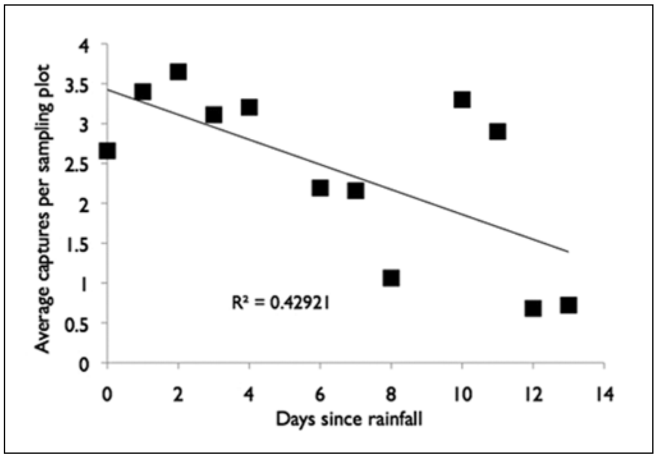

Figure 3.—Relationship between number of salamander captures and recent rainfall.

CONCLUSIONS AND PREDICTIONS

Our pretreatment results suggest that redback salamanders are more likely to be found in leaf litter than expected. Many studies on terrestrial salamanders focus on sampling natural cover objects and do not include leaf litter searches. We believe our sampling technique results in a more complete understanding of salamanders' distribution among microhabitats, and it will allow us to detect changes in cover object use following prescribed fire and timber harvest. We predict that competition for cover objects will increase following fire and timber harvest due to decreases in leaf litter and drier surface conditions. We expect to reveal more about the role of leaf litter loss in salamander population persistence.

It is important for managers to consider the unique characteristics of amphibians when implementing forest management plans. Because amphibians have different microhabitat requirements than other vertebrate taxa such as lizards or snakes, they are likely to have different responses to fire. Terrestrial salamanders, such as the southern redback, may be even more susceptible to fire than other amphibians.

ACKNOWLEDGMENTS

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SOIL AND LITTER NUTRIENTS FOLLOWING REPEATED AND PERIODIC BURNING IN THE MISSOURI OZARKS

Felix Ponder, Jr.

For more information, contact Daniel C. Dey, Research Forester, Northern Research Station,
202 ABNR Bldg., University of Missouri-Columbia, Columbia, MO 65211-7260.
To contact, call 573-875-5341 or email ddey@fs.fed.us

ABSTRACT

Extractable nutrients in the surface litter and soil of oak-hickory (*Quercus* spp. and *Carya* spp.) forest plots subjected to annual and periodic prescribed burning were studied on plots located in the Chilton Creek basin in southeast Missouri on land owned and managed by The Nature Conservancy (TNC). Soils are generally poor and excessively drained and have developed over dolomite and sandstone bedrock. The TNC has intensively monitored vegetation responses to the high frequency, low intensity fire regime at the site including such variables as frequency, timing, intensity, and fuel conditions since pretreatment in 1996 and posttreatment in 1998 (Table 1). Little work has been performed at the Chilton Creek Management Area to examine soil and litter nutrient changes that may result from fire activities on the highly weathered soils on the site.

In 2007, the crew monitoring vegetation was permitted to collect soil and litter samples on vegetation plots during the same visit. After processing samples, they were analyzed for various chemical properties including pH, total organic carbon (TOC), N, P, K, Ca, Mg, and S.

Nitrogen and S in the upper 0 to 15 cm of soil were significantly lower ($\alpha = 0.05$) for plots burned annually than in plots burned periodically (Table 2). Several nutrients (P, K, Ca, Mg, and S) in the surface litter also differed between burning treatments. Differences were not consistent among treatments, suggesting possible differences in plant species contributing to litter composition as a consequence of burning treatments.

Differences in nutrients attributed to aspect showed litter N, P, and K and soil N, P, K, and S were lowest on north facing slopes and usually highest on south facing slopes (Table 3). Differences in site moisture and temperature between aspects are likely explanations for the response. Correlations for aspect were significant for soil P ($r = 0.45$, $P = 0.016$) and for litter S ($r = 0.45$, $P = 0.034$). Except for N and P, concentrations of nutrients in litter were not strongly related to the concentration of nutrients in the soil. Further, among nutrients, only the concentration of P in the soil and its corresponding level in the litter were significantly correlated ($r = 0.49$, $P = 0.012$).

Overall, TOC, although somewhat lower in soil for the annual burn treatment, did not differ between burn treatments for soil or litter. Unfortunately, this study does not include a “control” or “no burn” treatment for comparison. However, considering the lack of a control treatment, based on soil nutrient concentrations and litter nutrition, these results suggest that annual and periodic burning has had minimal effects on soils within the Chilton Basin, and perhaps the most ecologically significant effect was the loss of soil

N and S associated with annual burning. While these data represent a one-time view after 9 years of annual and periodic prescribed burns, they do give insight on soil and litter nutrient changes associated with burning treatments within the Chilton Creek Management Area in the Missouri Ozarks.

ACKNOWLEDGMENTS

The author thanks the Nature Conservancy for permission to do this study.

Table 1.—Burn schedule for fire units in the Chilton Preserve (Missouri Ozarks)

Location	Aspect	Size (ac)	Treatment (Burn interval over 9 years) 1	2	3	4	5	6	7	8	9	Sequence	Return Interval
Kelly	North	600	X	X	X	X	X	X	X	X	X	Annual	1.00
Kelly	South	410	X		X	X			X			1.3.4.7	2.25
Chilton	East	460	X				X	X			X	1.5.6.9	2.25
Chilton	North	465	X	X	X			X				1.2.3.6	2.25

Table 2.—Effect of fire sequence and interval on soil quality characteristics in the upper soil layer in the Chilton Creek Preserve located in the Missouri Ozarks. EC=electrical conductivity. Means plus standard errors in parentheses within a row followed by different letters are significantly different at α = 0.05 (Tukey's HSD).

	Burn sequence Interval (Years burned over 9 years)				
Property	Annually 1.00	1.3.4.7 2.25	1.5.6.9 2.25	1.5.7 3.00	1.2.3.6 2.25
Soil					
pH	5.3(1.0)	5.8(0.9)	5.5(0.8)	4.9(0.4)	5.0(0.9)
EC	132.4(112)	117(66)	120.6(21)	83.4(14)	77(41)
Carbon (g kg^{-1})	1.5(0.5)	2.7(0.9)	1.8(1.5)	2.6(0.9)	2.3(0.7)
N (%)	0.08(0.03)b	0.17(0.08)a	0.11(0.03)ab	0.10(0.02)ab	0.12(0.02)ab
P (mg kg^{-1})	10.4(4.4)	17.8(4.1)	15.9(3.3)	12.2(3.5)	13.7(6.8)
K (mg kg^{-1})	0.6(10)	85.0(43)	85.7(14)	47.1(15)	61.0(46)
Ca (mg kg^{-1})	419(370)	833(621)	525(484)	184(129)	140(109)
Mg (mg kg^{-1})	106.5(74)	226.2(168)	69.9(55)	24.3(11)	36.1(18)
S (mg kg^{-1})	10.4(3.0)a	14.2(3.7)ab	15.1(4.5)ab	15.5(1.2)ab	17.4(4.6)b
Litter					
Carbon (g kg^{-1})	44.1(5.1)	43.8(2.3)	46.0(1.2)	46.0(2.1)	44.1(5.3)
N (g kg^{-1})	0.91(0.2)	1.2(0.1)	1.1(0.2)	0.99((0.1)	1.22(0.3)
P (g kg^{-1})	0.05(0.01)c	0.08(0.00)a	0.06(0.02)abc	0.05(0.00)bc	0.07(0.01)ab
K (g kg^{-1})	0.14(0.04)b	0.13(0.02)b	0.16(0.02)b	0.16(0.01)b	0.25(0.06)a
Ca (g kg^{-1})	1.5(0.4)ab	2.1(0.3)a	1.6(0.3)ab	1.2(0.5)b	1.3(0.4)ab
Mg (g kg^{-1})	0.20(0.07)a	0.14(0.05)ab	0.16(0.03)ab	0.10(0.03)b	0.14(0.03)ab
S (g kg^{-1})	0.09(0.02)ab	0.12(0.01)a	0.11(0.01)ab	0.08(0.01)b	0.10(0.02)ab

Table 3.—Influence of site aspect on soil quality characteristics in the upper soil layer (0 to 15 cm) and surface litter following burnings in the Chilton Creek Preserve located in the Missouri Ozarks. Means plus standard errors in parentheses within a row followed by different letters are significantly different at α = 0.05 (Tukey's HSD).

	Site aspect		
Property	**North**	**South**	**East**
Soil			
pH	5.2(0.8)	5.4(0.6)	5.5(0.9)
Ec	116.1(85)	98.5(43)	120.6(59)
Carbon (g kg-1)	1.9(1.4)	2.5(1.3)	1.8(0.9)
N (%)	0.09(0.04)b	0.14(0.07)a	0.11(0.6)ab
P (mg kg-1)	10.9(4.0)b	15.9(6.4)a	15.9(7.5)a
K (mg kg-1)	42.8(12.2)b	73.0(9.8)a	85.7(47.6)a
Ca (mg kg-1)	341.1(262)	513.2(354)	526.4(432)
Mg (mg kg-1)	79.1(42.5)	138.4(185.5)	69.9(57.1)
S (mg kg-1)	11.1(2.3)b	15.7(4.2)a	15.1(4.9)a
Litter			
Carbon (g kg-1)	44.7(3.2)	44.0(3.8)	46.0(1.2)
N (g kg-1)	0.93(0.12)b	1.21(0.017)a	1.12(0.16)ab
P (g kg-1)	0.05(0.01)b	0.07(0.01)a	0.06(0.02)a
K (g kg-1)	0.14(0.02)b	0.16(0.04)ab	0.20(0.02)a
Ca (g kg-1)	1.41(0.45)	1.63(0.33)	1.59(0.30)
Mg (g kg-1)	0.16(0.05)	0.14(0.04)	0.16(0.03)
S (g kg-1)	0.09(0.01)	0.10(0.01)	0.11(0.01)

HERBACEOUS UNDERSTORY AND SEED BANK RESPONSE TO RESTORATION EFFORTS AT AN OAK-PINE BARRENS IN CENTRAL WISCONSIN

Jeffrey L. Ralston and James E. Cook

Master of Science candidate (JLR) and Professor of Forestry (JEC), College of Natural Resources, University of Wisconsin-Stevens Point, 800 Reserve St., Stevens Point, WI 54481.
JLR is the corresponding author: to contact, call 920-810-6103 or email at jralston@new.rr.com.

ABSTRACT

The Wisconsin Department of Natural Resources (WDNR) has designated barrens communities as high priority for restoration. These communities are globally imperiled and support many endangered and threatened species. Many barrens are in various stages of degradation or are unmanaged. Restoration of structure and fire may increase understory richness of true prairie species (Tester 1989). An intermediate term evaluation of management efforts at the Quincy Bluff and Wetlands Area (Quincy Bluff) demonstrated only small changes with harvesting and return of fire (Nielsen 2003). This discrepancy in results may reflect the length of fire suppression and the degree of degradation at Quincy Bluff. Prior to initiating management, fire had been absent from Quincy Bluff for approximately 70 years, while other restoration sites have reported fire suppression of approximately 30 years.

Quincy Bluff is located in Adams County, Wisconsin and consists of over 1500 ha of several community types, including considerable oak-pine barrens. The objectives of this study were to test the impact of prescribed fire, timber harvest, and the seed bank on understory plant diversity at Quincy Bluff. Thirty-nine permanent transects were established prior to initiation of restoration efforts in five of The Nature Conservancy (TNC) management units and in the unmanaged WDNR property. An additional 26 newly established, independent transects and the 39 previously established transects were monitored in 2010. The following were determined: cover and presence of all living vegetation <1 m tall; shrub and small tree cover and density; and tree cover, density, and basal area. Soil samples were obtained for greenhouse analysis of the seed bank. Species richness and ground layer cover in 2010 were lowest in management units treated with prescribed fire alone and were 50 percent to 150 percent greater in units with substantial reduction in canopy cover and basal area. Structural changes resulted from timber harvest and/or a tornado that struck one management unit in 2004. After a literature review, we generated a list of species likely to be indicators of barrens vegetation (Bray 1960, Curtis 1959, Will-Wolf and Stearns 1999). In 2010, these barrens indicator species had the largest percent cover in the control unit, however, cover of barrens indicator species declined in all units. In units with reduced canopy cover and basal area, barrens indicator species richness increased by two to three species and declined or remained the same in the other units. Frequent fires eliminated shrubs and small trees. Greenhouse analysis of the seed bank demonstrated low species richness and produced few germinates of any restoration value.

Barrens, if left unmanaged, degrade to closed canopy forests with low species richness and low ground layer cover. Frequent fires over 17 years were ineffective at reducing canopy cover or basal area, and understory species richness remained low. Manipulation of the tree structure through timber harvest and/or natural processes (i.e., tornado) increased understory species richness, however, species determined to indicate healthy barrens communities decreased in relative cover, and richness only increased marginally. These results confirm that the resiliency of this system may be exceeded by the long period without management and the seven decades of fire suppression.

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PRESCRIBED FIRE AND THINNING IMPACTS ON FINE FUELS AT THE WILLIAM B. BANKHEAD NATIONAL FOREST, ALABAMA

Callie Jo Schweitzer and Stacy L. Clark

Research Forester (CJS), U.S. Forest Service, Southern Research Station,
P.O. Box 1568, Normal, AL 35762; Research Forester (SLC). CJS is corresponding author:
to contact call 256-372-4230 or email at cschweitzer@fs.fed.us.

INTRODUCTION

Prescribed burning and thinning are intermediate stand treatments whose consequences when applied in mixed pine-hardwood stands are unknown. The William B. Bankhead National Forest in north Alabama has undertaken these two options to restore unmanaged loblolly pine (*Pinus taeda* L.) plantations to hardwood-dominated stands. Our objective was to evaluate differences in fine fuel and duff dynamics in a combination of thinning and fire regimes.

METHODS

We installed a randomized complete block design with a 3 x 2 factorial treatment arrangement and four replications of each treatment. Treatments were three residual basal areas (50 ft^2 ac^{-1}, 75 ft^2 ac^{-1}, and an untreated control) with two burn frequencies (burns every 3 years and an unburned control). Two prescribed burns were implemented. We collected measurements before and after each treatment implementation. A 1 ft^2 wooden frame was used to collect samples, which were sorted into duff, leaves, 1-hour fuel (0.1-0.25 inches), 10-hour fuel (0.26-1 inch), fruit, and bark categories. Samples were oven dried until no change in weight was detected and then weighed to the nearest 0.01 g.

RESULTS

Main effect of thinning was significant for duff, 10-hour, 1-hour, and bark immediately after thinning. Compared to controls, thinning increased duff (+4.3 and +5.7 tons ac^{-1} for 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1}, respectively), 1-hour (+0.2 tons ac^{-1} for both thinning treatments), 10-hour (+1.6 and +1.4 tons ac^{-1} for 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1}, respectively), and bark loads (+0.4 and +0.3 tons ac^{-1} for 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1}, respectively). Three years following thinning, only leaf litter had significant differences with a reduction of -1.1 and -0.8 tons ac^{-1} in 50 ft^2 ac^{-1} and 75 ft^2 ac^{-1} treatments, respectively. Burning and its interaction with thinning were not significant after the first burn. After the second burn, leaf litter decreased by 0.5 tons ac^{-1} compared to controls, and bark increased by 0.1 tons ac^{-1}. Leaf litter decreased by 0.5 tons ac^{-1} after the second burn, and bark increased by 0.1 tons ac^{-1}. Duff, 1-hour, 10-hour, and fruit were not affected by the first or second burn.

CONCLUSIONS

Burning alone and in conjunction with thinning as applied in this study had minimal effects on fine fuels and duff. Thinning produced more of a reduction in fine fuels than did burning, likely due to higher decomposition rates related to increased sunlight hitting the forest floor. Based on these results, we do not recommend using prescribed fire in these stands to reduce fine fuel and duff loads, but recognize fire has other benefits not measured in this study for vegetation diversity and wildlife habitat.

SIMULATING TREATMENT EFFECTS IN PINE-OAK FORESTS OF THE OUACHITA MOUNTAINS

Ayn Shlisky, Kori Blankenship, and Shawn M. Cochran

Regional Analyst (AS), U.S. Forest Service, Pacific Northwest Region, 333 SW. First Ave., Portland, OR 97204; Fire Ecologist (KB), The Nature Conservancy; Wildlife Biologist (SMC), U.S. Forest Service, Ouachita National Forest. AS is corresponding author: to contact, call 503-808-2284 or email at ajshlisky@fs.fed.us.

ABSTRACT

Effective land management decisionmaking depends on scientifically sound analyses of management alternatives relative to desired future conditions and environmental effects. We used a state-and-transition model to evaluate likely future landscape conditions in a pine-oak forest on the Ouachita National Forest, Arkansas based on current and potential future alternative management actions. Our objectives were to: (1) demonstrate the use of state-and-transition models in project planning, and (2) create simple "what if" scenarios to supplement the project environmental impact assessment and facilitate more informed decisionmaking through relative comparisons. We used the model to simulate and compare the effects of several management alternatives:

A. Current Management
B. Regeneration Harvest/Thinning
C. Woodland Management
D. Regeneration Harvest/Thinning + Climate Change
E. Woodland Management + Climate Change

The effects of these alternatives were also compared against modeled ecological reference conditions.

At the time of the study, a national forest interdisciplinary team was completing a project-level environmental assessment of alternative management scenarios across the 16,700 acre Lower Irons Fork/Johnson Creek watershed. The watershed is located within the Ouachita National Forest near the town of Mena in western Arkansas. It is comprised primarily of pine-oak forest and woodland and has a history of active fire management. The watershed is a drinking water source for the town of Mena, Arkansas, offers recreational opportunities including hunting and fishing, and is home to two federally-endangered species: the red-cockaded woodpecker (*Picoides borealis)* and the harperella plant (*Ptilimnium nodosumis*).

We modified the LANDFIRE Ozark-Ouachita Shortleaf Pine-Oak Forest and Woodland model (USDA FS and USDI 2009) using stand exam and other forest data to represent current landscape structure and disturbance probabilities. Timber harvest volume per acre coefficients were estimated from a similar nearby project, and smoke production values for particulate matter (PM 10 and 2.5) were estimated from the First Order Fire Effects Model. Forest colleagues and U.S. Forest Service Southern Research Station scientists provided peer review. We used the Vegetation Dynamics Development Tool and the Path Landscape Model, a state-and-transition modeling framework and platform, to simulate the effects of the various alternatives after 10 years.

Our results indicate that a woodland management emphasis generally yielded landscape structure and fire frequencies closer to the desired future condition specified in the 2005 Ouachita National Forest Revised Forest Plan (USDA FS 2005) compared to a regeneration harvest/thinning emphasis (Figs. 1 and 2). When potential climate effects were considered, the woodland management emphasis also yielded greater smoke (Fig. 3) and woody biomass harvest outputs (Fig. 4) than the regeneration harvest/ thinning emphasis.

These findings suggest that Forest Plan revisions should reevaluate the desired future conditions for pine-oak forest in light of the fact that it does not currently include a standard for mid-seral forest structure, and that the existing desired future condition standard for late seral open woodland is lower than LANDFIRE reference conditions. While the model outputs have proven to be useful, the process forced the team to test assumptions and document knowledge, two intangible but valuable outcomes.

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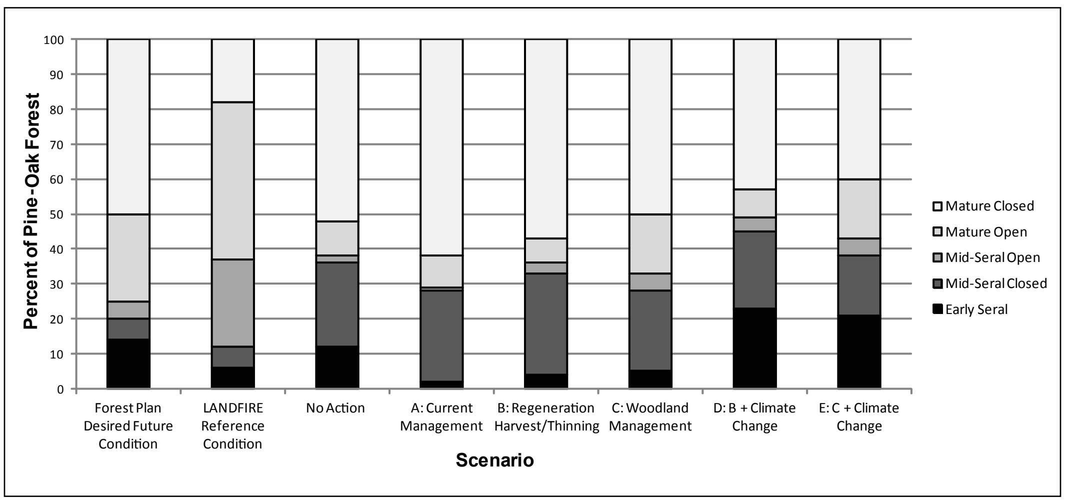

Figure 1.—Proportions of pine-oak forest and woodland structural stages resulting from modeled management alternatives, ecological reference conditions, and revised forest plan desired conditions after a 10-year simulation.

The content of this paper reflects the views of the author(s), who are responsible for the facts and accuracy of the information presented herein.

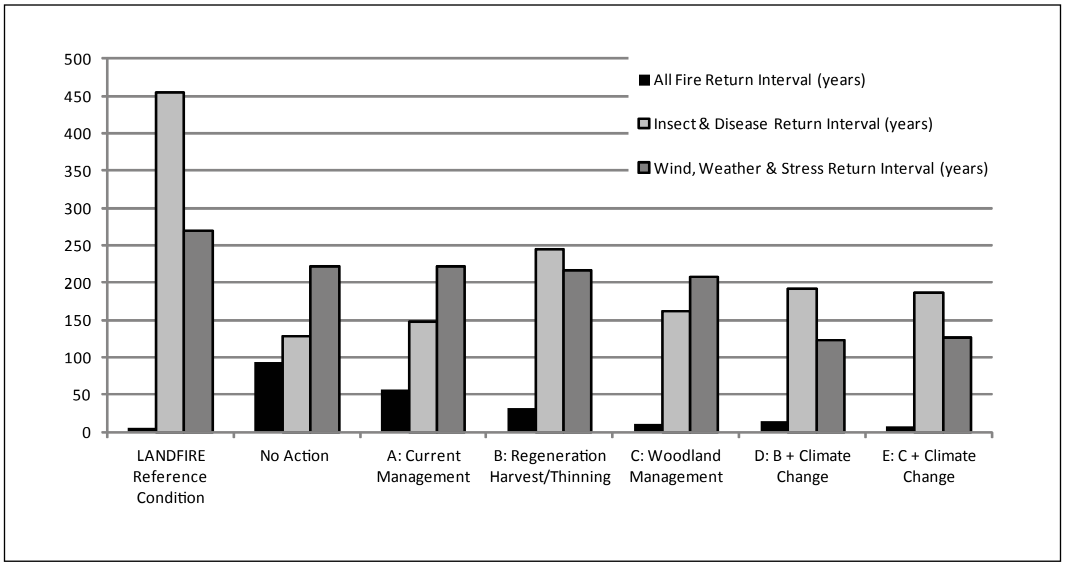

Figure 2.—Comparative return intervals for fire, insects and disease, and wind/weather/stress disturbances resulting from modeled management alternatives and ecological reference conditions.

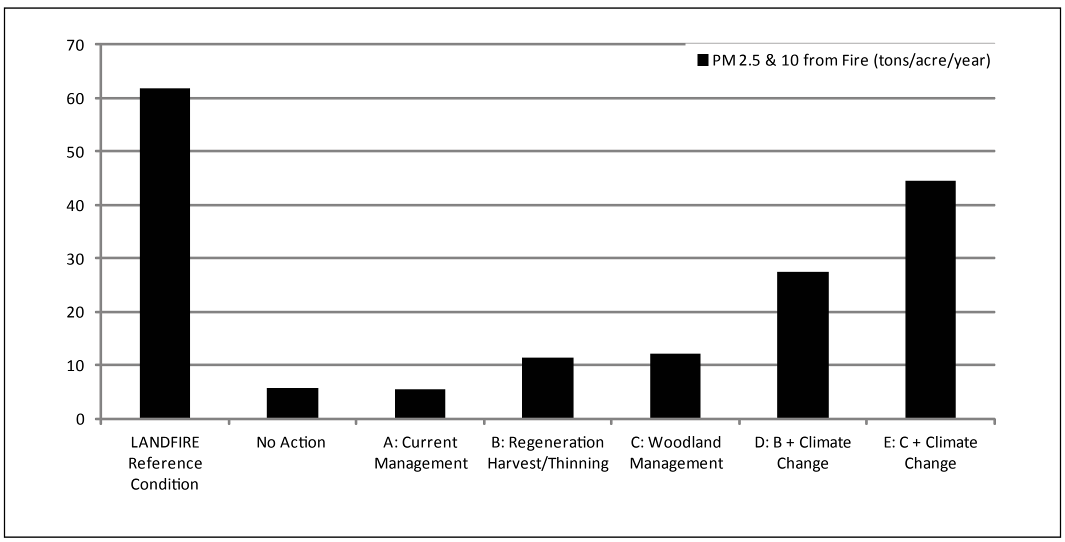

Figure 3.—Comparative smoke emissions resulting from modeled management alternatives and ecological reference conditions.

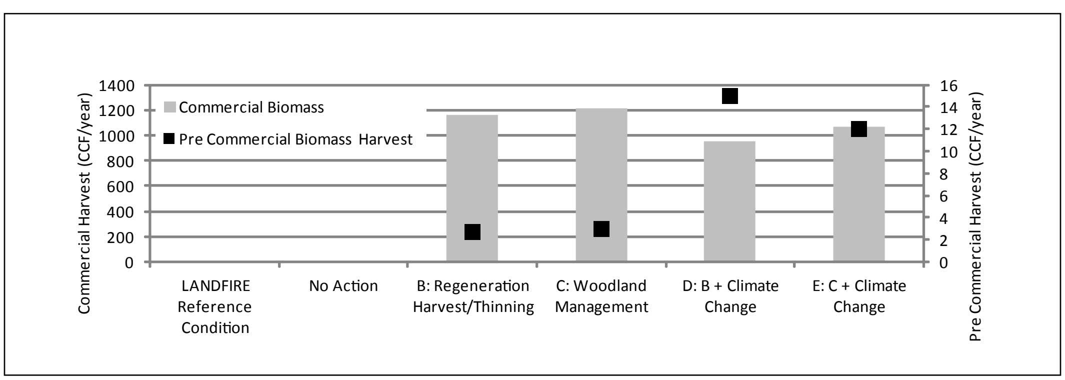

Figure 4.—Comparative biomass harvest resulting from modeled management alternatives.

VEGETATION STRUCTURE AND SOIL WATER CONTENT AFTER 60 YEARS OF REPEATED PRESCRIBED BURNING IN THE MISSOURI OZARKS

Kirsten Stephan and Jason A. Hubbart

Assistant Professor of Biology (KS), Department of Life and Physical Sciences,
Lincoln University of Missouri, 816 Chestnut St., Jefferson City, MO 65101;
Assistant Professor of Forest Hydrology (JAH). KS is corresponding author:
to contact, call 573-681-5116 or email at stephank@lincolnu.edu.

INTRODUCTION

Two important interrelated factors determining ecosystem structure and function are vegetation composition and soil moisture. Both are affected directly and indirectly by fire (Hart and others 2005). Few studies have characterized the long-term effects of repeated spring prescribed burning on these ecosystem characteristics. As an exception, Lewis and Harshbarger (1976) found that 20 years of repeated prescribed burning had significant and consistent effects on shrubs and herbs. However, post-fire soil moisture has been shown to increase (Klock and Helvey 1976), decrease (Campbell and others 1977), or remain the same (Wells and others 1979) after single fire events. This could be attributed to the timing of measurements, sampling design, and/or differential fire effects on the many factors that influence soil moisture, such as interception by plants or forest floor, evaporation from the canopy or the soil, transpiration, and infiltration rates.

The objectives of this study were to quantify the effects of 60 years of repeated spring prescribed burning at annual and 4-year intervals on a) vegetation composition and structure, b) soil physical properties, and c) their effects on soil moisture in a Missouri oak-hickory forest.

STUDY AREA

The research was conducted at University Forest Conservation Area located in the Ozark Mountains near Wappapello in southeast Missouri. Data were collected from a total of six 40 m by 40 m plots in two sites. Site 1 and site 2 are approximately 3.2 km apart and were established in 1949 and 1951, respectively. At each site, plots are separated by a 10 m buffer. At each site, one of the studied plots is burned every spring (annual burn treatment, A), every 4 years (periodic burn treatment, P), or never (control, C). The last periodic burns prior to data collection took place in 2007 (site 2) and 2009 (site 1).

METHODS

Vegetation cover and forest floor mass data were collected in 2010. Cover of herbaceous and woody understory vegetation (<0.5 m height) and woody midstory vegetation (0.5-2 m height) was assessed using 32 sampling quadrats (1 m by 1 m) per plot. Quadrats were placed every 5 m along transects spaced 10 m apart. Cover was estimated to the nearest percent. Cover of shrubs and trees in the midstory and overstory >1.4 m was assessed via the Leaf Area Index (LAI) using a ceptometer in nine locations per plot.

Forest floor mass was quantified by collecting material from eight circular areas (30 cm diameter) per plot. Forest floor material was air-dried; woody and nonwoody components were separated and weighed.

Soil physical properties were assessed in eight locations per plot in 2009. Infiltration capacity was measured via double-ring infiltrometers. Bulk density was determined from soil cores (5 cm diameter, 5 cm long). Soil volumetric water content (VWC) was continuously measured at 15 cm depth in one location per plot at 30 minute intervals from 2008 to 2010.

RESULTS & DISCUSSION

Tree and shrub cover (>1.4 m) was highest in the controls (LAI: A 2.3, P 2.1, C 3.4). This was attributed to fire-related tree mortality at the beginning of the study in annual and periodic burns. We predicted that interception, canopy evaporation, and transpiration losses by midstory and overstory trees lowered soil moisture in the controls more strongly relative to other treatments.

Cover of understory plants was highest in annual burns (cover in percent: A 40.0, P 37.2, C 14.0); cover of understory and midstory (<2 m) plants together was highest in the periodic burns (cover in percent: A 40.0, P 64.5, C 31.0). This was due to lack of woody seedling survival in annual burns but abundant resprouting in periodic burns. We predicted that interception, canopy evaporation, and transpiration losses by these plants reduced soil moisture in the periodic burn more strongly than in other treatments.

Forest floor (mainly leaf litter) was least abundant in annual burns because of frequent combustion (forest floor content in t ha^{-1}: A 4.7, P 8.5, C 15.4). The predicted effect on soil moisture was ambiguous. On one hand, low interception by and evaporation from a thin litter layer would facilitate precipitation to enter into the mineral soil. On the other hand, once in mineral soil, water is more likely to evaporate from a soil lacking litter insulation.

Infiltration capacity was lowest in annual burns (A 11, P 41, C 107 cm h^{-1}). This was likely due to a compacted top layer of soils from raindrop impact on bare ground in annual burns (bulk density in g cm^{-3}: A 0.99, P 0.96, C 0.94). We predicted that lower infiltration capacity decreased soil moisture in annual burns relative to other treatments.

In all treatments, direct measurements of soil moisture showed the expected temporal pattern of decreased soil moisture during the growing seasons due to increased evapotranspiration. However, there was no consistent fire effect on soil moisture. While VWC in annual burns was intermediate to the other treatments at both sites, VWC of the control was highest in site 1 but lowest in site 2 for the majority of the measurement period. This indicates the existence of significant microsite differences between sensor locations, masking any potential fire effect.

CONCLUSIONS

While burn treatments affected vegetation and soil physical characteristics in a similar way in both sites, there was no consistent treatment effect on soil volumetric water content between the two sites. We hypothesize that this is likely due to large microsite variability commonly observed in soils. Several VWC probes per plot would be required to capture plot-scale soil moisture levels. Despite the low intensity of spring prescribed burning, repeated burning over decades has dramatic effects on vegetation characteristics and soil physical properties. A mechanistic understanding of the effects of long-term prescribed burning will result in better informed forest management decisions in Ozark oak-hickory forest.

ACKNOWLEDGMENTS

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ABSTRACTS

RESTORATION OF HIGH ELEVATION RED OAK WITH FIRE IN THE SOUTHERN BLUE RIDGE

Peter Bates, Dean Simon, Gary Kauffman, Margit Bucher, Jon Shaffer, and Rob Lamb

Associate Professor (PB), Department of Geosciences and Natural Resources,
Western Carolina University, 331 Stillwell Building, Cullowhee, NC 28723;
Wildlife Forester (DS), North Carolina Wildlife Resources Commission; Botanist (GK),
U.S. Forest Service, National Forests of North Carolina; Assistant Director (MB),
The Nature Conservancy; Associate Forester (JS) and Forester (RL), Forest Stewards, Inc.
PB is corresponding author: to contact, call 828-227-3914 or email bates@email.wcu.edu.

ABSTRACT

Prescribed fire is most often used to restore fire-adapted plant communities on xeric and mesic sites since these are the areas where frequent, low-intensity fires are most common. In southern Appalachian forests, these sites are typically found on south- and west-facing slopes and on ridges that historically supported varying pine and oak communities. The Southern Blue Ridge Fire Learning Network (SBRFLN) was created with the goal of restoring historical fire regimes across the southern Blue Ridge. The SBRFLN has targeted four plant communities for restoration. Three of the four forest communities are typical of relatively xeric sites. These include Shortleaf Pine-Oak, Pine-Oak-Heath, and Dry-Mesic Oak-Hickory. However, the fourth, High Elevation Red Oak (HERO), is unique in that it occurs at high elevations that are generally more cool and moist. This poster will present information on the extent of the HERO type in the southern Blue Ridge as well as the current composition and structure of stands being targeted for restoration. We will compare current stand structure to HERO restoration models developed by SBRFLN and others. We will also present preliminary results from several prescribed burns that have been performed in these stands and discuss fuel load changes and burn performance.

COMPARING THE IMPORTANCE VALUE OF OVERSTORY AND UNDERSTORY TREES IN OAK FORESTS

Chad Bladow

Fire Manager, The Nature Conservancy, Indiana Field Office,
Efroymson Conservation Center, 620 East Ohio Street, Indianapolis, IN 46202.
To contact, call 317-829-3807 or email at cbladow@tnc.org.

ABSTRACT

In 2007, The Nature Conservancy began a long-term project at a site within the Brown County Hills of Indiana to manage dry-mesic oak forests. The understory of the oak forest is currently dominated by more mesic forest tree species. In the event of a canopy disturbance, there is little chance for oak and hickory saplings to compete with mesic forest species for canopy dominance. By comparing the importance value [IV = (relative basal area + relative density)/2] of tree species in the forest canopy (≥ 8 inch diameter at breast height [d.b.h.]) with the forest understory (4.5 ft tall - 8 inch d.b.h.) we can get a better idea of the severity of this problem. This monitoring will help determine if management activities are improving the IV of oak trees within the forest understory.

Baseline data were collected from fifty-three 0.1-acre monitoring plots in the summer of 2007. Oak species had an IV of 0.88 in the forest canopy compared to less than 0.04 in the understory. Data collected the first year following thinning and burning treatments in the fall/winter 2007 showed no increase in oak species' IV in the understory, but there was a 20.7 percent reduction in basal area and 6.25 percent reduction in canopy closure. The monitoring of the forest overstory vs. understory will repeat every 5 years plus 1 year following any management treatment and/or natural disturbance to determine if, over time, the IV of oaks in the understory increases. The long-term goal is to show whether forest management techniques used in the project can shift the dry/mesic forest understory from beech/maple dominance to oak/hickory.

THE EFFECTS OF SEASONALLY PRESCRIBED FIRE ON A DOLOMITE GLADE

Mary Ann Blinkhorn and D. Alexander Wait

Graduate Assistant (MAB) and Associate Professor (DAW), Missouri State University, 24058 East Cedar Point Rd, Lebanon, MO 65536-7719. MAB is corresponding author: to contact, call 417-718-7247 or email at Blinkhorn777@live.missouristate.edu.

ABSTRACT

The diverse vegetation that dominates glade systems is dependent on fire for long-term sustainability; therefore, prescribed fires are an important management tool. However, fire management has focused on spring and dormant season burns, with spring burns favoring grasses and dormant season burns favoring forbs. Growing season fires are known to kill woody plants that are not fire adapted; however, the effects of growing season burns on species composition are not well documented. In an effort to determine optimum methods of glade management, we are studying the effects of prescribed fire during different seasons in a dolomite glade at Ha Ha Tonka State Park in Camden County, MO. Three sites have been established, each containing four treatments. Within each site, one treatment area is designated as an unburned control, and each of the remaining treatment areas will have fire applied during the spring, dormant, or growing season. Initial observations and application of fire were conducted from July to September of 2010 with subsequent applications scheduled for February and April of 2011. Final data collection and analysis will extend through September of 2011. Data will be comprised of both pre- and posttreatment observations, as well as observations timed to coincide with spring/early summer and late summer flowering periods. Quantification of the effects of seasonal burns will be accomplished through population studies conducted within and among plots. The results of this study should enable land managers to schedule burns in order to achieve a desired vegetation response. Baseline floristic assessment and initial postburn results were presented at the 4th Fire in Eastern Oak Forest Conference.

RESEARCHING EFFECTS OF PRESCRIBED FIRE IN HARDWOOD FORESTS

Stacy L. Clark, Kathleen E. Franzreb, Cathryn H. Greenberg, Tara Keyser, Susan C. Loeb, David L. Loftis, W. Henry McNab, Joy M. O'Keefe, Callie Jo Schweitzer, and Martin Spetich

Research Forester (SLC, TK, DLL, WHM, CJS), Research Wildlife Biologist (KEF, JMO), Research Ecologist (CHG, SCL), Research Forest Ecologist (MS), U.S. Forest Service, Southern Research Station, 200 W.T. Weaver Boulevard, Asheville, NC 28804. CHG is the corresponding author: to contact, call 828-667-5261 ext.118 or email at kgreenberg@fs.fed.us.

ABSTRACT

The Upland Hardwood Ecology and Management Research Work Unit (RWU 4157) is a group of research teams located across the South, strategically placed to conduct research in physiographic sub-regions of the upland hardwood ecosystems including the southern Appalachian Mountains, the Cumberland Plateau, the Boston Mountains, and the Missouri Plateau. Our RWU is one of 16 maintained under the Southern Research Station by the U.S. Forest Service.

Our mission is to develop and disseminate knowledge and strategies for restoring, managing, sustaining, and enhancing the vegetation and wildlife of southern upland hardwood forests. Through experimental studies and modeling, our research program focuses on learning and predicting how upland hardwood-dominated forests and wildlife are affected by natural disturbances or silvicultural activities, and how plant and animal responses differ across environmental gradients such as elevation, moisture, and fertility.

One of our focal research areas is fire ecology and fire effects on hardwood forests and the wildlife communities they support. Understanding how fire affects upland hardwood forest communities will help land managers to develop scientifically-based methods to meet their management and restoration goals. Here we highlight some of our current studies on fire ecology.

FIRE ECOLOGY STUDIES

Historical Fire Frequency

We are using tree cores and fire scars to assess the frequency of fire in upland hardwood ecosystems prior to fire suppression efforts starting in the 1930s. For instance, in the Boston Mountains of Arkansas, we found that widespread fire occurred more often during drought years in the 1700s with fires likely achieving sizes unprecedented during the last century. Early transitional (1810-1830) settlement by Cherokees at population densities under 0.26 humans/km^2 was highly correlated ($r = 0.90$) with the number of fires per decade in the interior region of the Boston Mountains. Multiple regression analyses further implicated humans as well as short- and long-term climate variability such as forced by the El Niño/Southern Oscillation (ENSO) and Atlantic Multidecadal Oscillation (AMO). Understanding presettlement fire frequencies will help land managers in ecosystem restoration efforts.

Fire and Fire Surrogate Study (FFS)

Scientists with RWU 4157 are participating in the wildlife component of the national collaborative Fire and Fire Surrogate Study (FFS). This long-term study is assessing how ecological components or processes may be changed or lost if fire surrogates, such as cuttings and mechanical fuel treatments, are used instead of fire or in combination with fire. Virtually no comparative data exist on how these treatments mimic ecological functions of fire, or how bird, reptile, amphibian, or small mammal communities respond to prescribed fire or fire surrogate treatments.

Regional Oak Study

Scientists within our RWU have partnered with the North Carolina Wildlife Resources Commission, the Stevenson Land Company, the Northern Research Station, and the Mark Twain National Forest in a regional study of how hardwood tree species respond to prescribed fire and other silvicultural treatments across a productivity gradient and across the Central Hardwood Region. We and our collaborators are also studying the response of herbaceous plants, seed banks, acorn viability, artificial northern red oak regeneration, fuels, bat, bird, reptile, amphibian, and small mammal communities to prescribed fire and other oak regeneration treatments. This regional oak study includes three independent, fully replicated study areas representing different physiographic areas of the Central Hardwood Region including the Southern Appalachian Mountains (NC), the Cumberland Plateau (TN), and the Ozark Highlands (MO). University collaborators with our regional oak study include the University of Tennessee, Alabama A&M University, North Carolina State University, and the University of Missouri.

Indiana Bats and Prescribed Fire

Scientists in our RWU are looking at the compatibility of prescribed fire in the Southern Appalachians with the conservation of the federally endangered Indiana bat. In cooperation with the Nantahala National Forest, Cherokee National Forest, and Great Smoky Mountains National Park, we are examining the effects of prescribed fire on snag population dynamics, Indiana bat roost tree availability in relation to fire history, and Indiana bat roost tree selection in relation to fire history and stand and landscape characteristics. This study will provide land managers with the information they need to manage Indiana bats and restore pine-oak habitats throughout the southern Appalachians.

Using Prescribed Fire to Restore Oak-Dominated Upland Hardwood and Hardwood-Mixed Pine Systems

Scientists are studying the use of regeneration and intermediate silviculture prescriptions coupled with fire to manage and restore upland hardwood systems. We have implemented a large-scale silvicultural assessment designed to examine the efficacy of stand-level prescriptions in reducing the potential impacts of gypsy moth infestations and oak decline on upland hardwood forests on the Daniel Boone National Forest, Kentucky. Early assessments showed a slight increase in tree vigor as determined by crown cover and position of residual trees in shelterwood with reserves, thinning, and oak woodland treatments. In a process to move a mixed-pine hardwood forest towards hardwood-dominated stands on the William B. Bankhead National Forest in Alabama, scientists found that following the initial thinning and burning treatments there was a 30 percent reduction in percent canopy cover, and light penetration through the canopies ranged from 5 to 25 percent pretreatment to 29 to 60 percent posttreatment. The cool, slow-moving burns had no discernable effect on the overstory trees. Avian and herpetofaunal population dynamics appear to be influenced more by the thinning than the fires.

Amphibians and Prescribed Fire in Longleaf Pine-Wiregrass Sandhills

Scientists are studying amphibian and reptile use of isolated sinkhole ponds in both regularly burned and long-unburned Florida longleaf pine-wiregrass sandhills. This study will help land managers assess how prescribed fire affects herpetofaunal populations in the long term.

Artificial Regeneration and Prescribed Fire

Scientists are studying how high quality seedlings of planted oak (*Quercus* spp.) and American chestnut (*Castanea dentate* [Marsh.] Borkh.) respond to prescribed burning. Preliminary results indicate that seedlings can withstand burning several years after planting if root collar diameters are relatively large when established.

Fire and Oak Decline

Scientists used LANDIS to model oak decline in the Boston Mountains of Arkansas 150 years into the future under two fire return intervals. The analysis delineated potential oak decline sites and established risk ratings for these areas. This is a further step toward precision management and planning.

SMALL MAMMAL OCCURRENCE IN OAK WOODLANDS AND RESTORED SAVANNAS

Valerie J. Clarkston and Patrick A. Zollner

Graduate Research Assistant (VJC) and Assistant Professor of Wildlife Ecology (PAZ), Department of Forestry and Natural Resources, Purdue University, 195 Martstellar Street, West Lafayette, IN 47907. VJC is corresponding author: to contact, call 765-969-4016 or email at vclarkst@purdue.edu.

ABSTRACT

Oak savannas are rare in the United States, and few quantitative data exist on how the small mammal community will respond to the restoration of this ecosystem. We present the baseline data for an ongoing savanna restoration research project at the Jasper-Pulaski State Fish and Wildlife Area in northwest Indiana. During the summer of 2009, we investigated the density, abundance, and microhabitat preferences of small mammal species in oak woodlands versus existing oak savannas to better predict if shifts in species composition would occur after the restoration process. We used one-way analysis of variance to compare abundance of each species to habitat type and multiple linear regressions to compare their abundance with microhabitat variables. Significant differences in species abundance were not observed between oak woodlands and oak savannas. White-footed mice were positively correlated with percent herbaceous cover and basal area of white oaks, but negatively correlated with percent cover of soft mast. Southern flying squirrels were trapped only in oak woodlands and were positively correlated with basal area of black oaks. Eastern chipmunks were positively correlated with percent herbaceous cover and negatively correlated with woody stem density. Red squirrels were only captured in oak woodlands and were positively correlated with soft mast and basal area of white and black oaks, but were negatively correlated with percent herbaceous cover. Our data suggest that oak woodlands converted to oak savannas will lose woodland obligate species, and microhabitat characteristics are better predictors of species occurrence than the macrohabitat.

VECTOR ANALYSIS OF RESPONSE OF ADVANCE REPRODUCTION TO REPEATED PRESCRIBED BURNING

Zhaofei Fan and Daniel C. Dey

Assistant Professor (ZF), Department of Forestry, Mississippi State University, Box 9681, Mississippi State, MS 39762-9681; Research Forester (DCD), U.S. Forest Service, Northern Research Station. ZF is corresponding author: to contact, call 662-325-5809 or email at zfan@cfr.msstate.edu.

ABSTRACT

Advance reproduction is important to restore and maintain oak dominance. Response of oaks (*Quercus* spp.) and associated species to prescribed fire depends on a wide array of factors including site condition, fuel characteristics, fire frequency and intensity, and size or age of advance reproduction. It will be useful to quantify species response to fire for prescribing burning treatments that promote oaks while curbing their competitors. We used the 10-year Chilton Creek prescribed burning experiment data and vector analysis to simultaneously compare oak and associated species growth and resprouting ability, two important traits to determine future oak status. The species and size groups/classes identified based on the vector analysis can be directly applied to future prescribed burning practices.

WOODLAND RESTORATION IN THE OZARK HIGHLANDS OF ARKANSAS

Allison Fowler, Douglas Zollner, and John Andre

Field Ecologist (AF) and Director of Conservation Programs (DZ), The Nature Conservancy Arkansas Field Office, 601 North University Ave., Little Rock, AR 72205; Ecologist (retired) (JA), U.S. Forest Service. AF is corresponding author: to contact, call 501-614-5084 or email at allison_fowler@tnc.org.

ABSTRACT

Fire is one of the most important ecosystem disturbance processes in the Ozark-Ouachita Highlands. Conversely, the exclusion of landscape-scale fires from these ecosystems for the past several decades has changed the structure and species composition of the forest. Fire exclusion has allowed an increase of shade-tolerant, fire-intolerant tree species. The ecosystem restoration project on the Big Piney and Pleasant Hill Ranger Districts of the Ozark-St. Francis National Forest encompasses eight project areas designed to reduce hazardous fuels and increase forest health. Current restoration activities being implemented by the Forest Service include prescribed fire and commercial and noncommercial thinning. In order to assess changes in forest health over time, we monitored plant community structure and composition within the 102,120 acre ecosystem restoration project areas using 127 randomly placed macroplots. Data were collected during the summers of 2003-2006 (baseline) and 2007-2009 (remeasure). Data were stratified based on topographic position. Tree densities (stems per acre) decreased significantly in the north slope, south slope, and ridgetop communities. Overall tree density decreased as well. The majority of the decrease in stems per acre was from the midstory tree layer. Species richness in the ground layer increased in all communities, with a significant increase in the ridgetop community. Current restoration activities appear to be adequately producing the desired changes in forest structure. The continuation of prescribed fire is crucial to improving forest health and bringing the forest closer to desired ecological condition. In addition, continued monitoring can track progress toward desired condition and help guide forest management planning.

RESPONSE OF SEASONAL BIRD SPECIES DENSITIES TO HABITAT STRUCTURE AND FIRE HISTORY ALONG AN OPEN–FOREST GRADIENT

Ralph Grundel and Noel B. Pavlovic

Research Ecologist (RG), U.S. Geological Survey, Great Lakes Science Center,
1100 North Mineral Springs Road, Porter, IN 46304; Ecologist (NBP).
RG is corresponding author: to contact, call 219-926-8336 ext. 428
or email at rgrundel@usgs.gov.

ABSTRACT

Along an open-forest landscape gradient in northwest Indiana, we assessed associations of bird species with grasslands, savannas, woodlands, scrublands, and forests by relating fire frequency and vegetation characteristics to seasonal densities of 72 bird species distributed across the open–forest gradient. About one-third of the species did not exhibit statistically significant relationships with any combination of seven vegetation characteristics that included vegetation cover in five vertical strata (bare ground; vegetation 0.3-1 m height; vegetation 1-2 m height; living woody shrubs, sprouts, saplings, or small trees 2.5-10 cm diameter at breast height [d.b.h.]; and living trees >10 cm d.b.h.), dead tree density, and tree height. For 40 percent of the remaining species, models best predicting species density incorporated tree density. Therefore, management based solely on manipulating tree density may not be an adequate strategy for managing bird populations along this open–forest gradient. When fire frequency, measured over 15 years, was added to vegetation characteristics as a predictor of species density, it was incorporated into models for about one-quarter of species, suggesting that fire may modify habitat characteristics in ways that are important for birds but not captured by the structural habitat variables measured. Among those species, similar numbers had peaks in predicted density at low, intermediate, or high fire frequency. Given these avian compositional variations along the open-forest gradient, managers considering restoration of landscapes will face a fundamental challenge. What should the habitat composition of the restored landscape look like? We developed a model for evaluating the desirability for birds of different landscape habitat compositions by quantifying an important conservation tradeoff inherent in making restoration decisions, the tradeoff between the landscape's ability to promote avian species diversity and the landscape's use by threatened avian species. This quantification allowed us to evaluate the ability of different landscape compositions to achieve preferable tradeoff compromises, such as maximizing diversity for a given level of landscape use by threatened species. Managers can use such tradeoff results to evaluate which landscape compositions are associated with particular conservation and management priorities.

THE TRACE OF FIRE IN EASTERN NATIVE AMERICA

Richard P. Guyette, Michael C. Stambaugh, Daniel C. Dey,
Rose M. Muzika, and Joseph M. Marschall

Research Professor (RPG), Missouri Tree-Ring Laboratory, University of Missouri, 203 ABNR Building, Columbia MO 65211; Research Associate (MCS); Research Forester (DCD), U.S. Forest Service, Northern Research Station; Associate Professor (RMM); Research Specialist (JMM). RPG is corresponding author: to contact, call 573-882-7741 or email at GuyetteR@missouri.edu.

ABSTRACT

Written in the rings of trees is a history of fire in Native America that tells of humans, drought, and their interactions. These fire histories in eastern Native America move through generations and territories from 1600 to 1850. These quantitative histories are based on thousands of fire scars found on oak and pine trees. Each fire scar has a date, location, and associated human population. Here we examine the connections between the occurrence of wildland fire in Native America and the people who lived there, the Algonquin, Cherokee, Chippewa, Osage, Menominee, and others. The documentation of fire history ranges from the ecosystems of Appalachia, the Great Lakes, the Southeast, and the Midwest. We found changes in fire frequency associated with Native American populations in Alabama, Arkansas, Kentucky, Michigan, Missouri, Tennessee, Oklahoma, Ontario, Pennsylvania, and Wisconsin. Many fire regimes in eastern Native America are found to have a temporal human "footprint", that is, an abrupt or rapid change in fire frequency not related to climate. The interactions of drought and human migrations and ignition are detected in the fire scar record. During years with large fires, severe drought is the predisposing factor, and human ignitions represent the inciting factor associated with the occurrence of widespread fires.

CLIMATE AND PHYSICAL CHEMISTRY IN EASTERN FIRE REGIMES

Richard P. Guyette, Michael C. Stambaugh, Rose M. Muzika, and Daniel C. Dey

Research Professor (RPG), Research Associate (MCS), and Professor (RMM), School of Natural Resources, University of Missouri, 103 Anheuser-Busch Natural Resources Building, Columbia, Mo 65211; Research Forester (DCD), U.S. Forest Service, Northern Research Station. RPG is corresponding author: to contact, call 573-882-7741 or email at GuyetteR@missouri.edu.

ABSTRACT

Knowledge of the temporal-spatial variability in fire intervals is needed for understanding the changing effects of climate on fire regimes. This work focused on translating the physical chemistry of ecosystem processes and climate into fire regimes. This was achieved by using empirical, process, and multiple regression modeling to translate the laws and units of physical chemistry into the processes of ecosystem fire regimes. The Physical Chemistry Fire Frequency Model (PC2FM) is based on the Arrhenius equation and was calibrated with fire scar data, charcoal studies, and expert interval estimation. The model predicts the climate forcing of mean fire intervals (MFIcf) from temperature, precipitation, their interactions, and the partial pressure of oxygen. We used fire interval data from 166 sites in North America and elsewhere to calibrate the PC2FM. The PC2FM was calibrated with data from pre-Euro-American periods to reduce the effects of climate change, land use, fire suppression, and other nonclimatic factors affecting fire regimes. Details of the model's chemistry and statistics are presented. The model is applied to ecosystems at scales from 1 km^2 or larger, but can span multiple time periods and climate scenarios. Mean fire intervals are mapped for the historic period from approximately 1600 to 1820 using the PC2FM at regional scales in the eastern United States, at a broader scale in the United Sates, and at continental scales. Since the model does not directly include vegetation, natural or human ignitions, management, or topography, it only predicts fire regime characteristics that are influenced by climate.

INFLUENCES OF TOPOGRAPHY AND FIRE ON EASTERN RED CEDAR (*JUNIPERUS VIRGINIANA*) DISTRIBUTION IN A PRAIRIE-FOREST TRANSITION ZONE

Lyndia D. Hammer and Michael C. Stambaugh

Research Assistant (LDH) and Research Associate (MCS), Department of Forestry, University of Missouri, 203 ABNR Building, Columbia, MO, 65211. LDH is corresponding author: to contact call 573-882-8841 or email at ldhp54@missouri.edu.

ABSTRACT

Historic information indicates that eastern redcedar (*Juniperus virginiana* L.; ERC) was restricted to rocky bluffs and fire-protected areas throughout much of its range during the pre-Euro-American settlement period. As a consequence of settlement, dramatic changes occurred in land use, grazing, fire, and human populations, all of which coincided with a widespread increase in eastern redcedar's distribution and density. To more closely identify the relative importance of these factors, we initiated a study addressing ERC demographics and growth in a relatively intact post oak (*Quercus stellata* Wangenh.) woodland complex within the Wichita Mountains National Wildlife Refuge (WMNWR) located in southwestern Oklahoma. Tree density, ages, and site information were measured in 353 plots distributed throughout a 14,000 hectare area comprised of plains, riparian zones, and rugged mountains. Preliminary analysis suggests a positive correlation exists between tree age and local topographic roughness and rocky soil substrate. Overall, younger trees (<100 years old) dominated the age distribution refuge wide. Expansion appeared to occur more widely in moderately rough terrain compared to the extremes found in grasslands (gentle) and peaks (very rough). In areas with more open forest canopy conditions (e.g., savannas, open woodlands), ERC appears to exhibit nurse tree characteristics, whereby younger ERC more commonly establish at the bases of older oak trees. Older ERC trees (200-500+ years old) occupy increasingly topographically rougher terrain. We hypothesize that topography was historically an important barrier with respect to limiting fire spread and that a topographic roughness index (in addition to other variables) could be used to delineate the historic spatial distribution of ERC that existed during the past few centuries when frequent burning was well documented in these woodlands.

DENDROCHRONOLOGICAL DETERMINATON OF HISTORICAL FIRE OCCURENCE AND RECRUITMENT IN A SOUTHERN ILLINOIS OAK-HICKORY FOREST

J.M. Harris, Charles M. Ruffner, M. Therrell, and C.K. Nielsen

Graduate Student (JMS), Department of Forestry, Mailcode 4411,
Southern Illinois University Carbondale, Carbondale, Illinois 62901; Associate Professor (CMR);
Assistant Professor (MT); Assistant Professor (CKN). JMS is corresponding author:
to contact, call 618-401-2295 or email at jeffrey.harris24@siu.edu.

ABSTRACT

Throughout the central hardwoods, fire return interval dramatically increased during the period of Euro-American settlement. Fire was used as a tool for clearing land and improving forage for grazing. The Shoal Creek study site is located in Jackson County, Illinois, 8 km southwest of Murphysboro. Shoal Creek is situated at the northern extent of the Illinois Ozark Hills, classified as a Subsection of the Ozark Highlands Section. The region is unglaciated, and loess caps are 10 m deep on the ridgetops and 1-3 m deep on side slopes. Preliminary results suggest the site was frequently burned during postsettlement, with a mean fire interval (MFI) of 2.95 years from 1887 to 1946. Fire waned from the site in the 1930s, and the last major fire occurred in 1946. By this time, Shawnee National Forest had become established in southern Illinois, and fire suppression was the preferred management technique. Thirty-three fire scarred cross sections were opportunistically sampled from trees on a southwest aspect of a *Quercus*-*Carya* forest. Cross sections were sanded to 600 grit, and skeleton plots were used to determine signature years for cross-dating purposes. Year and seasonality of individual fire scars and approximate pith date were determined for each sample and were utilized in FHX2 fire history software. Recruitment history revealed that overstory *Quercus*-*Carya* species established under favorable conditions in the early 20th century. Timber was harvested from the site around 1900, and intense fires followed for the next 30 years. A small pulse of *Acer*-*Fagus* germinated as fire frequency decreased on-site during the 1930s, and a significant pulse established immediately after the last recorded fire in 1946. Superposed epoch analysis (SEA) determines the influence of immediate weather patterns and overall climate trends surrounding fire event years. SEA was run to compare fire event years at Shoal Creek with Palmer Drought Severity Index (PDSI) climate reconstructions. For the 95 percent confidence interval, there was not a significant association between fire and climate. In the central hardwoods, lightning is associated with rainstorms, and fires burn in both dry years and wet years, so the relationship between fire and climate is not strong. If rehabilitation of *Quercus*-*Carya* dominated forest stands is the management objective, the results of this study will aid in fire cycle planning of frequency and seasonality. Managers may consider the MFI for rehabilitation burns, and range of fire intervals for long-term maintenance burns. However, prescribed burns are not the only answer for managers. Fire must be used in accordance with silvicultural techniques that mimic natural disturbance regimes such as timber stand improvement (TSI) and shelterwood harvests which create large overstory gaps suitable for oak-hickory recruitment.

OAK-GRASSLAND RESTORATION AT LAND BETWEEN THE LAKES NATIONAL RECREATION AREA: PRELIMINARY RESULTS

Jaime A. Hernandez and Elizabeth F. Danks

Forester (JAH), Land Between the Lakes National Recreation Area, U.S. Forest Service, 100 Van Morgan Drive, Golden Pond, KY 42211; NEPA Editor/Wildlife Biologist (EFD). JAH is corresponding author: to contact, call 270-924-2073 or email at jahernandez@fs.fed.us.

ABSTRACT

Two areas totaling 8,630 acres within the Land Between the Lakes National Recreation Area (LBL) were designated as oak-grassland demonstration restoration areas in 2004. Dormant and growing season prescribed fires along with several thinning operations occurred between 2006 and 2010. Prescribed fire frequency ranged from one to three ignitions within this time period. We conducted a comparison between the changes in basal area, canopy cover, and the occurrence or composition of understory herbaceous cover within areas treated by one fire with areas burned more than once, and areas that were both burned and thinned. Burned and thinned areas were also compared by thinning regime: cut-and-leave versus commercial thinning. Over the last 4 years, we have already seen that the vegetation management programs at LBL have affected forest structure and composition. Prescribed burning has had an effect on understory composition, and to a lesser degree, midstory composition. However, any structural changes accomplished by just the use of prescribed burning were limited to the most xeric sites. The combination of prescribed burning and thinning treatments had an effect on both species and structural composition across all canopy levels, with all sites types with herbaceous development increasing as well. Oak-grassland structure has yet to be completely developed within the two sites; however, several of the areas have developed a woodland structure. In conjunction with further prescribed burns, this combination should provide the higher light levels that are key to sustaining an understory dominated by herbaceous species.

THE EFFECTS OF PRESCRIBED BURNING ON THE BLACK KINGSNAKE: GOING BEYOND THE DEMOGRAPHIC DATA

Christopher A.F. Howey and Willem M. Roosenburg

Ph.D. Candidate (CAFH), Department of Ecology and Evolutionary Biology, Ohio University, Athens, OH 45701; Associate Professor (WMR). CAFH is corresponding author: to contact, call 609-413-5942 or email at chris.howey@gmail.com.

ABSTRACT

Species responses to habitat alteration are most frequently studied by estimating changes in population numbers. However, subtle changes in the habitat may cause indirect effects that go unnoticed in the short term yet can cause adverse population effects in the long term. Although demographics may be informative to reveal how snakes respond directly to habitat alterations, understanding how landscape changes may alter prey abundances, preferred microhabitat availability and use, predation intensity, and foraging strategy would assist biologists in knowing more of the collective effects caused by landscape alterations. The objective of this project was to determine how the black kingsnake (*Lampropeltis getula nigra*) is affected by changes to the landscape that are caused by prescribed burning. In the summer of 2010, I began this project at Land Between The Lakes National Recreational Area in southwestern Kentucky. I set up four study plots (each 800 m x 800 m; 64 ha) within the Franklin Creek Area burn unit (~960 ha in total size) and four study plots of equal size in an adjacent unburned habitat with similar topography. The Franklin Creek Area was burned in April 2007 and again in September 2010. Drift fences with funnel and pitfall traps were erected in the center of each plot, and an array of coverboards were placed throughout each plot. During the summer of 2010, a total of 848 reptiles, amphibians, and small mammals were captured, marked, and released. Reptile species richness and diversity indices (DI) were lower in burned plots (13 species, DI = 2.03) than in control plots (17 species, DI = 2.37). Biophysical copper models were deployed in each plot to measure the potential body temperatures a black kingsnake could achieve. Mean temperatures in burned plots were consistently warmer than in control plots and more frequently exceeded the critical thermal maximum of black kingsnakes (42 °C) suggesting that habitat in burned plots may become too warm for black kingsnakes, thus limiting the amount of time that they could be active (foraging for food or searching for a mate). Available habitat was measured within all plots prior to the second burn, and burn plots were characterized by fewer understory trees and higher ground temperatures. Burned plots also had higher air temperatures, a lower percentage of leaf litter, and shallower depths of leaf litter than control plots. The preferred body temperature (T_{pref}) was also measured for four captured kingsnakes in a thermal gradient arena, and mean T_{pref} was determined to be 28.2 ± 1.6 °C. Of these captured kingsnakes, one was large enough to have a radio-transmitter surgically implanted, and I am currently tracking this individual within a burned plot. This project is currently in the second of three field seasons.

MULTIPLE BURNS PAVE THE WAY FOR IMPROVED OAK-HICKORY REGENERATION IN CANOPY GAPS IN SOUTHERN OHIO FORESTS

Todd F. Hutchinson, Elaine Kennedy Sutherland, Robert P. Long, Joanne Rebbeck, and Daniel A. Yaussy

Research Ecologist (TFH), Research Plant Physiologist (JR), and Research Forester (DAY), U.S. Forest Service, Northern Research Station, 359 Main Road, Delaware, OH 43015; Supervisory Research Biologist (EKS), U.S. Forest Service, Rocky Mountain Research Station; Research Plant Pathologist (RPL), U.S. Forest Service, Northern Research Station; TFH is corresponding author: to contact, call 740-368-0064 or email at thutchinson@fs.fed.us.

ABSTRACT

In 1995, we began a study of prescribed fire as a tool to sustain mixed-oak forests. We hypothesized that repeated fires would reduce stand density "from below", favoring the development of oak-hickory advance regeneration relative to the shade-tolerant species (e.g., red maple) that dominated the understory. Our study consisted of four 60 acre stands burned two to five times from 1996 to 2005, and two unburned stands, located in the Vinton Furnace Experimental Forest, OH. We collected overstory and regeneration data on a total of fifty-four 0.3 acre plots located across the full range of upland topographic moisture conditions in the six stands. Though fire reduced the density of 1-4" d.b.h. (diameter at breast height) saplings (-74 percent) and 4-10" d.b.h. midstory trees (-30 percent) on these permanent plots by 2008, canopy cover remained high (>85 percent), and the relative abundance of oak-hickory advance regeneration was not significantly greater in burned vs. unburned plots. In 2003, canopy gaps formed in these same stands during a regional white oak decline. In 2008, we quantified understory structure and tree regeneration in 52 canopy gaps (separate from the permanent plots); 28 gaps were in three stands burned three to five times and 24 gaps were in three unburned stands. Gaps were formed from the death of one to nine canopy trees (mean = 4). The understory structure of burned gaps was much more open than unburned gaps, which had a dense layer of saplings and midstory trees. Burned stands had significantly more oak-hickory advance regeneration in gaps (range 3,725 to 5,590 stems/ac) than did unburned stands (607 to 2,308 stems/ac); and oak-hickory dominated the larger advance regeneration layer (stems 1ft tall to 1 inch d.b.h.) in burned gaps. Sassafras advance regeneration was also more abundant in burned gaps while shade tolerant species and all other intolerant species were equally abundant in burned and unburned gaps. While fire alone did not clearly benefit oak-hickory regeneration in closed canopy forests, canopy gaps that formed after multiple burns became dominated by large oak-hickory regeneration. Our results suggest that, given enough time, repeated fires can increase the probability that oak and hickory will retain dominance after a canopy disturbance.

ASSOCIATION OF FIRE SEASON, FIRE FRONT, AND FIRE TEMPERATURE ON THE MORTALITY OF EARLY SUCCESSIONAL PERSIMMON AND SWEETGUM TREES

Paul Killian, Joe Robb, and Darrin Rubino

Graduate Student (PK), Division of Biology, Kansas State University, 116 Ackert Hall,
Manhattan, KS 66506; Refuge Manager (JR); Associate Professor (DR).
PK is corresponding author: to contact, call 317-402-9904 or email at killianp@k-state.edu.

ABSTRACT

Land managers commonly treat grasslands with prescribed fire to decrease encroachment of woody vegetation and to maintain grassland biodiversity. The focus of our study was to assess the mortality of two early successional invaders, sweetgum and persimmon, in response to various fire regimes. We compared: (1) tree mortality between spring dormant-season (March-April) and fall growing-season (October) fires; and (2) the relationship between fire intensity and tree mortality. To examine these objectives, we sampled tree mortality from four burns between the fall of 2007 and the spring of 2009 at Big Oaks National Wildlife Refuge in southeastern Indiana. Our results suggested that the fire season, fire front, and fire temperature were all positively associated with mortality of these two early successional trees in a species-specific manner.

High intensity fires associated with head fires, and spring dormant season fires had the greatest impact on the mortality of persimmon trees. However, fall growing season fires of either high intensity or long duration had the greatest impact on sweetgum trees. Full mortality of these trees was difficult to achieve due to prolific root and collar sprouting occurring shortly after the fires.

POSTFIRE SUSCEPTIBILITY OF HABITATS TO INVASION BY SEED OF ORIENTAL BITTERSWEET

Stacey A. Leicht-Young, Noel B. Pavlovic, and Ralph Grundel

Botanist (SAL-Y), U.S. Geological Survey, Lake Michigan Ecological Research Station;
Ecologist (NBP, RG), U.S. Geological Survey, Great Lakes Science Center,
1100 North Mineral Springs Road, Porter, IN 46304. NBP is corresponding author:
to contact, call 219-926-8336 ext. 428 or email at npavlovic@usgs.gov.

ABSTRACT

Prescribed fires are a common management tool in forested ecosystems. These fires help to maintain forest structure as well as species biodiversity. Fires also create disturbance which could allow for invasion of exotic species. Oriental bittersweet (*Celastrus orbiculatus*) is an exotic liana (woody vine) that is aggressively moving its way west from the eastern United States. This species is a major invader of forests and can disrupt natural succession. We set out to investigate if habitats that have been burned provide more conducive conditions for bittersweet germination than those that have not. We examined six different habitat types including sand prairie, moraine prairie, oak savanna, oak-hickory forest, beech-maple forest, and oak forest. In each habitat type we had replicated 6 x 6 m plots that were divided into four different treatments: high fire intensity, low fire intensity, litter removed, and a control. We burned these plots in either the spring (prairies and oak savannas) or fall (oak-hickory, beech-maple, and oak forest) dormant seasons. After the burns (April), we introduced 25 seeds of oriental bittersweet to each of the four treatments within the plot and monitored their germination and height monthly throughout the growing season (May-September). When examining the maximum number of seedlings per sampling period over the 2 years, the forested habitat plots (beech-maple and oak forest) had significantly greater percent germination than the open plots (prairie, savanna); however this comparison is confounded with year of sowing and differences in weather between years. Plots on moraine soils had greater overall percent germination than those on sandy soils. In the prairie and savanna habitats, the control had the highest percent germination, while the litter removed had the lowest. The high and low intensity fire treatments were not significantly different from the control nor the litter removed plot. In the oak and beech-maple forest habitats, the germination percentage in the control treatment was significantly lower than the low and high fire treatments. The litter removed treatment was not significantly different from the control or the two burn treatments. Thus, in some habitat types, fire could make the plant community more susceptible to invasion, while in others, it does not. This result is largely a function of how exposed the seeds become in the habitat in the absence of litter. In more open habitats (prairie and savanna) these seeds are exposed to more sun and do not germinate well, while in the forested plots, having less litter is conducive to germination since the canopy of the forest protects the seeds from excessive sun and heat. These results will assist land managers in making decisions when burning in forested areas with high amounts of oriental bittersweet.

DETERMINING LUMBER VALUE CHANGES IN FIRE INJURED OAK TREES

Joseph M. Marschall, Richard P. Guyette, Michael C. Stambaugh, and John Dwyer

Research Specialist (JMM), Professor (RPG), Research Associate (MCS),
Associate Professor (JD), Department of Forestry, University of Missouri-Columbia,
203 ABNR Building, Columbia, Missouri 65211. JM is corresponding author:
to contact, call 573-882-8841.

ABSTRACT

Prescribed fire is used for a variety of management tasks in woodlands with only limited information on how it affects lumber product values. We analyzed how fire related injuries affect lumber volume and grade in red oaks (*Quercus coccinea* and *Q. velutina*) harvested from two sites in southern Missouri. Trees (n=58) with varying degrees of external damage (scar size), time since fire, and log size were harvested and milled into dimensional lumber. Lumber grade and scale changes due to fire related injuries were tracked on individual boards (n=423; 4160 board feet). To estimate lumber product value losses, grade and scale changes were compared to an expected grade and scale as if no fire injuries were present. Preliminary analysis indicates tree size, scar size, and time since fire to be important predictors in decreased lumber product values, and minimal value loss occurs within the first 10 years after fire-caused injuries occur. Current field sampling has occurred on mid-quality woodland sites where lumber product quality is typically low. Future field sampling (approximately 30 additional trees) and analysis will target higher quality sites.

ESTIMATING FUEL CONSUMPTION DURING PRESCRIBED FIRES IN ARKANSAS

Virginia L. McDaniel, James M. Guldin, and Roger W. Perry

Forestry Technician (VLM), Supervisory Ecologist (JMG), and Research Wildlife Biologist (RWP),
U.S. Forest Service, Southern Research Station, Hot Springs, AR 71901.
VLM is corresponding author: to contact, call 501-623-1180 ext. 112
or email at vmcdaniel@fs.fed.us.

ABSTRACT

While prescribed fire is essential to maintaining numerous plant communities (especially those dominated by oak and pine), fine particles produced as smoke can impair human health and reduce visibility in scenic areas. As a result of these concerns, the Arkansas Voluntary Smoke Management Guidelines were established by the Arkansas Forestry Commission to mitigate the impact of smoke from prescribed fire on people's health and to ensure adherence to air quality regulations. These guidelines use standard fire behavior and fuel models developed elsewhere in the United States. The accuracy of these models for determining fuel loading and consumption in Arkansas, however, is unknown. We established 120 modified Brown's transects in 15 burn units and three community types on the Mena, Oden, and Poteau Ranger Districts of the Ouachita National Forest in Arkansas to determine fuel loads before and after prescribed fires. The three community types were pine-oak (*Pinus echinata*-*Quercus* spp.) forest, oak forest, and pine woodland. In addition to the ordinary Brown's methodology of measuring litter and duff depth and tallying woody fuels, we also clipped attached vegetation and collected 1 and 10 hour fuels in adjacent quadrats before and after the prescribed fires. This enabled us to estimate the live fuel component not sampled by Brown's transects and test the accuracy of ordinary Brown's transects in terms of woody fuel consumption. We used localized bulk density values to convert inches of litter and duff into tons per acre. We then used FFI (Fire Ecology Assessment Tool-Firemon Integrated) software to quantify fuel consumption on 7 of the 15 prescribed fires. Preliminary analyses showed that the fuel consumption occurring in the Ouachita Mountains was consistent with expected values based on standard fire behavior and fuel models (Table 1), and that fuel consumption in restored woodlands was significantly less than that in closed canopy forests (Fig. 1).

Table 1.—Comparison of fuel consumption values predicted by standard fire behavior fuel models and Brown's transect data collected on dormant season prescribed fires on the Ouachita National Forest in Arkansas, 2010 and 2011

Community Type	Standard Fuel Model	Actual Brown's Transect Data*
	tons/acre	tons/acre
Pine-oak forest	3.0–4.4	3.0–5.4
Oak forest	0.8–2.5	2.0–3.8
Pine woodland	1.5–5.9	0–1.9

* Includes Brown's transect data only (N = 32, 14, and 9 for pine-oak forest, oak forest, and pine woodland, respectively).

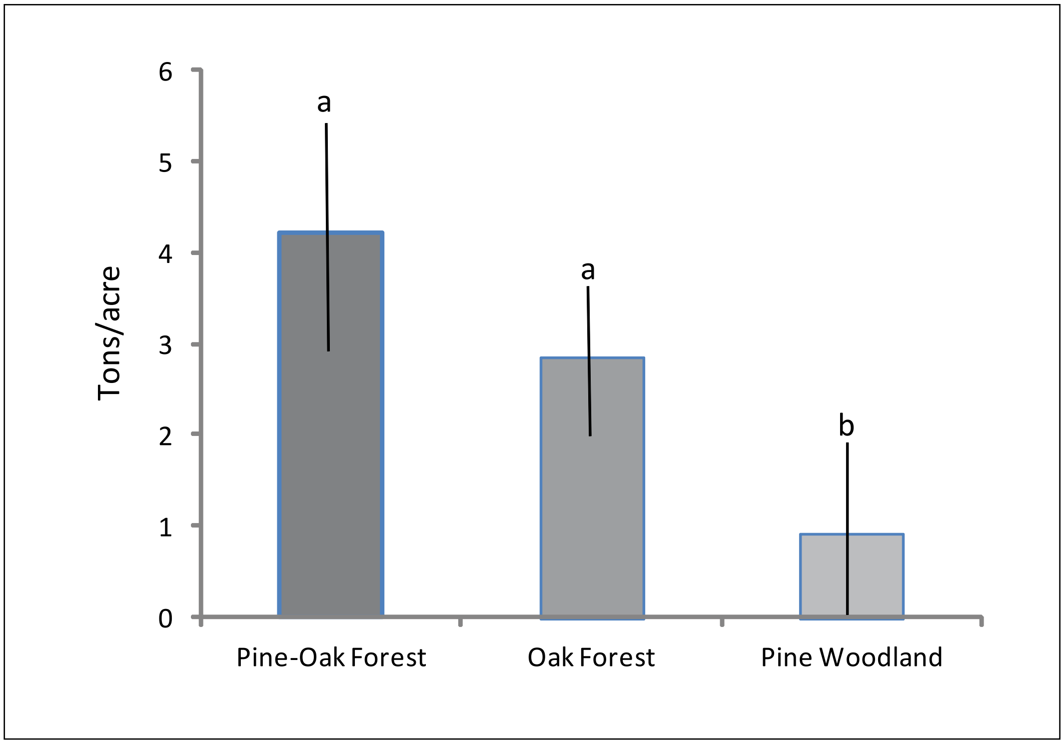

Figure 1.—Overall fuel consumption in pine-oak forest, oak forest, and pine woodland using only Brown's transect data on the Ouachita National Forest in Arkansas, 2010 and 2011. Different letters indicate a significant difference ($p \leq 0.05$, mean ±2se) among forest types.

FIRE EFFECTS ON THE RESPROUTING AND TOTAL NONSTRUCTURAL CARBOHYDRATES OF THE HIGHLY INVASIVE ORIENTAL BITTERSWEET

Noel B. Pavlovic, Stacey A. Leicht-Young, and Ralph Grundel

Ecologist (NBP), U.S. Geological Survey, Lake Michigan Ecological Research Station,
1100 North Mineral Springs Rd., Porter, IN 46304; Botanist (SAL-Y); and Ecologist (RG).
NBP is corresponding author: to contact, call 219-926-8336 ext. 428
or email at npavlovic@usgs.gov.

ABSTRACT

An important evolutionary strategy for surviving fire is the ability to resprout. Resprouting ability is determined by resistance to burning, the location of meristems, and storage of special chemical reserves for regrowth. Total nonstructural carbohydrates (TNC) in the roots can be an important source of readily available reserves for rapid resprouting. Oriental bittersweet (*Celastrus orbiculatus*) is a highly invasive liana that is migrating westward across the central portion of North America. This aggressive liana can girdle trees, reach and cover forest canopies, increase tree susceptibility to ice damage, and alter successional trajectories. While it is known that this species can vigorously root sprout after a fire or from cutting, no quantitative study on this prolific resprouter has been done. We are conducting a replicated experiment on sand and moraine soil substrates to examine the effects of burning, cutting, and the combination of the two on resprouting and regrowth of Oriental bittersweet. We initiated eight experimental blocks with control, burn, cut, and cut and burn treatments in the spring and fall. Extreme weather prevented us from conducting growing season treatments (which were moved to the fall dormant season), so we added an additional cutting treatment in early July. Oriental bittersweet density and cover were measured in four 1 m^2 subplots in each treatment plot (10 × 10 m) with a pretreatment inventory done in July 2009 and a posttreatment remeasurement done in July 2010. Stems were classified into six size categories: seedlings; <2.5 mm in diameter; 2.5-5.0 mm; 5.1-10 mm; 10.1-15 mm; and >15 mm. We also collected three root segments of Oriental bittersweet from each plot in March, May, and July for TNC analysis. We used the differences in Oriental bittersweet cover, stem counts, and diameters as our response variables for the preliminary comparisons between 2009 and 2010. We found that burning and cutting plus burning reduced bittersweet cover more than just cutting and no treatment, but the reduction was less on the richer morainal soils than on sand. We found that the cut and burn treatment had significantly greater numbers of resprouts compared to the cut treatment ($F_{2,1} = 2.1$, $P = 0.1$). In addition, the number of resprouts increased with size class ($F_{4,1} = 9.8$, $P < 0.001$). When we examined the percentage of plants that were killed in each size class we found, as expected, that the largest size class had the most survival overall. The spring cut and burn had the most killed stems, but also had the highest number of resprouts. Seasonally, TNC declined from a peak in March to moderate levels in May and then increased by July. Cutting bittersweet in early July resulted in a 75 percent reduction in TNC compared to dormant TNC levels. Our results have important implications for developing effective strategies for controlling and eliminating Oriental bittersweet.

CHANGES IN FOREST UNDERSTORY ASSOCIATED WITH REDCEDAR ENCROACHMENT IN FIRE SUPPRESSED *QUERCUS STELLATA* FORESTS

Paul van Els, Rodney E. Will, Michael W. Palmer, and Karen R. Hickman

Ph.D. student (PvE), University of Las Vegas; Associate Professor (REW), Department of Natural Resource Ecology and Management, Oklahoma State University, 008C Agricultural Hall, Stillwater, OK 74078; Professor (MWP, KRH), Oklahoma State University.
REW is corresponding author: to contact, call 405-744-5444 or email at rodney.will@okstate.edu.

ABSTRACT

Fire suppression in oak woodlands of Oklahoma, Missouri, and Arkansas often leads to the establishment of a redcedar (*Juniperus virginiana*) midstory that alters species richness and understory productivity. In a *Quercus stellata* dominated forest in northern Oklahoma, we compared vegetation in forest gaps, forests without redcedar, at the inner and outer edge of redcedar canopies, and near trunks (200 plots total). Species richness (11 to 6 spp. m^{-2}) and cover (53.3 to 12.7 percent) declined with proximity to redcedar trunks. Regression analysis indicated that richness ($R^2 = 0.08$) and cover ($R^2 = 0.18$) were best explained by redcedar litter mass. Partial canonical correspondence analysis revealed two strong canonical axes, one related to litter/light and the other to cover of *Quercus* spp. versus redcedar. Tree seedlings and woody vines dominated near redcedar. Forbs, graminoids, and *Quercus* spp. seedlings were more common in areas without redcedar. Our study indicates that litter is the main determinant of understory vegetation declines associated with midstory redcedar encroachment in these fire-suppressed forests. Decreases in herbaceous litter loads, which historically contributed to the accumulation of fuel beds, will have a positive feedback effect on midstory encroachment. Declines in recruitment of *Quercus* spp. that were related to increasing abundance of redcedar and consequent increases in litter loads eventually may lead to changes in overstory composition.

FROM SPRING EPHEMERALS TO LIGHT TO FUNGI: PLANT DYNAMICS IN AN OZARK OAK/HICKORY-FOREST/WOODLAND COMMUNITY MANAGED FOR WILDLIFE THROUGH PRESCRIBED BURNS

D. Alexander Wait and Mary Ann Blinkhorn

Associate Professor (DAW), Missouri State University, Temple Hall 248, 901 South National Ave, Springfield, MO 65897; Graduate Assistant (MAB). DAW is corresponding author: to contact, call 417-836-5802 or email at alexanderwait@missouristate.edu.

ABSTRACT

This research examines the effects of prescribed fire on community structure and function in Ozark forests. The research aim is to inform decisions about future monitoring and management strategies for forests in southwest Missouri managed for wildlife, and more generally, to understand how heterogeneity in light through a canopy affects community and ecosystem function. Eighteen circular 0.1-ha plots have been established across three "treatments/habitats" at the Drury-Mincy Conservation Area (DMCA) in Taney County, MO. In addition, belt transects and 100 m^2 areas have been sampled in conjunction with circular plot sampling. Burning was prescribed on approximately 70 percent of closed forest at DMCA in 1999 with the goal of reestablishing open woodlands, with subsequent burns in 2001, 2003, 2008, and 2010. Prior to this, the closed forest areas had not burned in over 50 years. Small areas of DMCA have been burned since the early 1980s and are sampled as "reference sites", while areas unburned for over 50 years are sampled as "control sites". All prescribed burns have been conducted in March and April on the same forest areas. Fire is clearly opening the canopy and resulting in greater heterogeneity in photosynthetically active radiation reaching the forest floor. However, after 12 years, recently burned forests do not have significant oak regeneration, and plant community structure and function are still more typical of a closed forest than open woodland.
We will present data on physiology of oak and hickory saplings, herbivory on oak saplings, soil respiration, leaf litter inputs, fungal abundance, overstory production, light penetration to understory, spring ephemeral diversity, and overall understory, midstory, and overstory plant species richness and cover. Finally, we will summarize interactions towards the goal of increasing our ability to predict plant and animal population dynamics and anticipate and minimize habitat degradation and pest species invasions in Ozark forests.

PRESCRIBED BURNING IN AN UPLAND OAK FOREST REDUCED LITTER N, SOIL ORGANIC C, AND GRAM NEGATIVE BACTERIA

Ryan J. Williams and Steve W. Hallgren

Graduate Research Assistant (RJW), Ecology, Evolution and Organismal Biology, Iowa State University, 3424 Oakland Street, Ames, IA 50014; Associate Professor (SWH), Oklahoma State University. RJW is corresponding author: to contact, call 515-450-6809 or email at ryanw@iastate.edu.

ABSTRACT

Prescribed burning is a management technique used for reducing fuel loads, preventing exotic species invasion, and maintaining wildlife habitat. In upland oak (*Quercus* spp.) forests, prescribed burning prevents the process known as mesophication where the dominant canopy species shift from fire tolerant oaks to fire intolerant tree species. Recently, it has become increasingly important to manage forests from an ecosystem service perspective, especially for soil carbon storage. However, little data exists that explores the relationship between prescribed burning at different frequencies and forest soil properties. We assessed the effect of prescribed burning at 0, 2.5, and 5 fires per decade (FPD) on litter, soil, and the soil microbial community in an upland oak forest in central Oklahoma. Results indicated burn treatments had 26 percent lower litter total nitrogen, resulting in a higher C:N ratio ($P = 0.0017$ and $P = 0.0039$ respectively). The litter lignin:N ratio was lower under 0 FPD when compared to 2.5 FPD but was no different from 5 FPD ($P = 0.019$). Areas burned at 5 FPD had 2.5 times less soil organic matter (SOM) and 2.7 times less soil organic carbon (SOC) than other treatments ($P = 0.0383$). Carbon storage dropped from 36.0 Mg ha^{-1} at 0 FPD to 15.7 Mg ha^{-1} at 5 FPD. Differences in SOM and SOC were associated with increased soil bulk density ($P = 0.0039$). Phospholipid fatty acid analysis (PLFA) indicated that gram-negative bacteria were significantly less abundant under 5 FPD ($P = 0.0378$). Differences in litter chemistry could cause heterogeneity in ecosystem functioning between forest areas with different fire histories, yet changes in litter chemistry did not reflect the differences that occurred in soil chemistry. Differences between the response of litter and soil to fire show an uncoupled nutrient cycling relationship between these two pools. Less litter N may limit the availability of N in soil pools after repeated burning over time as reflected by the C:N and lignin:N ratios. However, nitrogen limitation may not be as dramatic under 5 FPD due to lower amounts of recalcitrant lignin in litter. The reduction in SOM and SOC could affect soil properties like cation exchange capacity and porosity, affecting the availability of nutrients and water in the soil. Shifts in the microbial community could occur because of the reduction of SOC or changes in other nutrients not measured in this study. Our sampling occurred long enough after burning (1.5 years) to indicate that shifts in the soil microbial community were due to the soil environment rather than the burning itself. It is unknown how this change in the soil microbial community will effect nutrient cycling, but it is evident that prescribed burning can alter the soil microbial community at a broad taxonomic level. The differences found in SOM, SOC, and gram-negative bacteria illustrate how burning at a high frequency may be destructive to SOM and SOC, while maintaining an intermediate burn frequency maintains a soil environment like that of an unburned forest.

Dey, Daniel C.; Stambaugh, Michael C.; Clark, Stacy L.; Schweitzer, Callie J., eds. 2012. **Proceedings of the 4th fire in eastern oak forests conference**; 2011 May 17-19; Springfield, MO. Gen. Tech. Rep. NRS-P-102. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 292 p.

Contains 14 full-length papers and 40 abstracts of posters that were presented at the 4th Fire in Eastern Oak Forests conference, held in Springfield, MO, May 17-19, 2011. The conference was attended by over 250 people from 65 different organizations and entities, representing 22 states and 1 Canadian province.

KEY WORDS: fire, eastern forests, forest management, oak

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