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The Hardwood Ecosystem Experiment: A Framework for Studying Responses to Forest Management









Abstract

Conditions in forested ecosystems of southern Indiana are described before initiation of silvicultural treatments for the Hardwood Ecosystem Experiment (HEE). The HEE is a 100-year study begun in 2006 in Morgan-Monroe and Yellowwood State Forests to improve the sustainability of forest resources and quality of life of Indiana residents by understanding ecosystem and human responses to forest management. Human-focused articles detail the history of disturbance from pre-Columbian times, the evolution of forest policy and management, values associated with the state forests that host the HEE, human perceptions of modern forest management in Indiana, and outreach efforts designed to inform stakeholders about the HEE. Interest in forest, wildlife, and fish resources is high in Indiana, and four of five residents have visited a state forest. Support for timber harvest as a management tool also is high if done professionally and not exclusively to supply lumber. Among users and neighbors of the forests that contain the HEE, educational intervention increased the acceptability of forest management and reduced the acceptability of not harvesting.

To study ecological responses to silviculture, nine research core areas were randomly assigned one of three treatments implemented in 2008 (uneven-aged, even-aged, or no harvest). Ecologically focused articles describe pre-treatment conditions during 2006-08 for targeted species (cerulean warbler, eastern box turtle, timber rattlesnake) and communities (woody plants, breeding birds, terrestrial salamanders, small mammals, bats, wood-boring beetles, and moths). Despite the broad diversity of taxa studied, few differences were noted during the pre-treatment period. Differences that did exist were related either to inter-annual variation, variation associated with local habitat features, or variation tied to differences occurring at the landscape scale. Predicted responses of plants and animals to silvicultural treatments are provided, and future challenges and opportunities associated with the HEE are discussed.

Cover Photos

Top Left: Hardwood Ecosystem Experiment sign in front of 2008 clearcut; used by permission of Rebecca A. Kalb, Purdue University

Top Right: Female cerulean warbler (*Setophaga cerulea*); used by permission of Jennifer R. Wagner, Ball State University

Bottom Left: Northern red oak (*Quercus rubra*) acorn production; used by permission of Sally S. Weeks, Purdue University

Bottom Right: Timber rattlesnake (*Crotalus horridus*); photo by Indiana Department of Natural Resources

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THE HARDWOOD ECOSYSTEM EXPERIMENT REPORT—AN OUTLET FOR PEER-REVIEWED INFORMATION

Charles H. Michler

This report is intended to be the first in a series of compiled scientific information and results that update interested readers on the latest findings of the Hardwood Ecosystem Experiment. In this report, each manuscript was peer reviewed by at least two anonymous experts from the scientific and practitioner community. Reviewed manuscripts were returned to authors for revision, and revised manuscripts were edited and formatted by one of the co-editors (M.R. Saunders, R.A. Kalb, G.S. Haulton, C.H. Michler) and the senior editor (R.K. Swihart). Final submission was made to the U.S. Department of Agriculture (USDA), Forest Service, Northern Research Station for uniform format and type style. Forty-eight percent of the manuscripts required major revision, 43 percent required moderate or minor revision, and 9 percent were accepted without revision. As required in the peer-reviewed literature, authors were responsible for accuracy and content of their manuscripts. The mention of trade names and pesticides, fertilizers, and other chemicals within the manuscripts does not constitute an endorsement by the USDA Forest Service or the U.S. Department of Agriculture. Statements of the contributors external to the U.S. Forest Service may not necessarily reflect the policy of the Agency.

Updated information on the Hardwood Ecosystem Experiment may be found at www.HEEForestStudy.org. For additional information, readers may email info@HEEForestStudy.org; or call Charles Michler, Purdue University, Department of Forestry and Natural Resources, at 765-494-3591 or email at cmichler@fs.fed.us or michler@purdue.edu.

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CONTENTS

HISTORICAL AND CURRENT PERSPECTIVES ON FOREST MANAGEMENT 1

The History of Human Disturbance in Forest Ecosystems of Southern Indiana 2
Michael A. Jenkins

Indiana Forest Management History and Practices 12
Sam F. Carman

Indiana Residents' Perceptions of Woodland Management 24
Daniel J. Witter, Shannon M. Amberg, David J. Case, and Phillip T. Seng

INTRODUCTION TO THE HARDWOOD ECOSYSTEM EXPERIMENT 35

The Hardwood Ecosystem Experiment: Goals, Design, and Implementation 36
Rebecca A. Kalb and Cortney J. Mycroft

TARGETED WILDLIFE SPECIES 60

The Cerulean Warbler in Morgan-Monroe and Yellowwood State Forests, Indiana: Pre-treatment Data on Abundance and Spatial Characteristics of Territories 61
Kamal Islam, Kyle J. Kaminski, Margaret M. MacNeil, and Lila Prichard Young

Spatial Ecology and Behavior of Eastern Box Turtles on the Hardwood Ecosystem Experiment: Pre-treatment Results 78
Andrea F. Currylow, Brian J. MacGowan, and Rod N. Williams

Spatial Ecology of Timber Rattlesnakes on the Hardwood Ecosystem Experiment: Pre-treatment Results 86
Brian J. MacGowan and Zachary J. Walker

TARGETED WILDLIFE AND PLANT COMMUNITIES 95

Pre-treatment Analysis of Woody Vegetation Composition and Structure on the Hardwood Ecosystem Experiment Research Units 96
Michael R. Saunders and Justin E. Arseneault

Breeding Bird Communities of the Hardwood Ecosystem Experiment 126
Melissa C. Malloy and John B. Dunning, Jr.

Relative Abundance and Species Richness of Terrestrial Salamanders on Hardwood Ecosystem Experiment Sites before Harvesting 142
Jami E. MacNeil and Rod N. Williams

A Pre-treatment Assessment of Small Mammals in the Hardwood Ecosystem Experiment 151
Natasha A. Urban and Robert K. Swihart

Oak Mast Production and Animal Impacts on Acorn Survival in the Central Hardwoods 176
Kenneth F. Kellner, Jeffery K. Riegel, Nathanael I. Lichti, and Robert K. Swihart

Bats of the Hardwood Ecosystem Experiment before Timber Harvest: Assessment and Prognosis 191
Jeremy J. Sheets, John O. Whitaker, Jr., Virgil Brack, Jr., and Dale W. Sparks

Habitat Use by Bats in Two Indiana Forests prior to Silvicultural Treatments for Oak Regeneration 203
Jeremy J. Sheets, Joseph E. Duchamp, Megan K. Caylor, Laura D'Acunto, John O. Whitaker, Jr., Virgil Brack, Jr., and Dale W. Sparks

Pre-treatment Assemblages of Wood-boring Beetles (Coleoptera: Buprestidae, Cerambycidae) of the Hardwood Ecosystem Experiment 218
Jeffrey D. Holland, John T. Shukle, Hossam Eldien M. Abdel Moniem, Thomas W. Mager, Kapil R. Raje, Kyle Schnepp, and Shulin Yang

The Lepidoptera as Predictable Communities of Herbivores: A Test of Niche Assembly using the Moth Communities of Morgan-Monroe State Forest . 237
Keith S. Summerville, Michael R. Saunders, and Jamie L. Lane

SOCIO-ECONOMIC IMPLICATIONS OF FOREST MANAGEMENT 253

Public Acceptability of Forest Management Practices at Morgan-Monroe State Forest 254
Shannon C. Rogers, William L. Hoover, and Shorna B. Allred

Value of the Morgan-Monroe-Yellowwood State Forest Complex 287
William L. Hoover

CONCLUDING SYNTHESIS AND LOOKING FORWARD 314

Science in the Hardwood Ecosystem Experiment: Accomplishments and the Road Ahead 315
Michael R. Saunders and Robert K. Swihart

The Hardwood Ecosystem Experiment: Extension and Outreach 333
Brian J. MacGowan, Lenny D. Farlee, and Robert N. Chapman

Past is Prologue: A Synthesis of State Forest Management Activities and Hardwood Ecosystem Experiment Pre-treatment Results 339
G. Scott Haulton

HISTORICAL AND CURRENT PERSPECTIVES ON FOREST MANAGEMENT

THE HISTORY OF HUMAN DISTURBANCE IN FOREST ECOSYSTEMS OF SOUTHERN INDIANA

Michael A. Jenkins[1]

Abstract.—The forests of southern Indiana have been shaped and defined by anthropogenic disturbance. Native Americans influenced composition and structure through land clearing and burning, but the scale and rate of human disturbance intensified with European settlement. Sustained settlement led to the loss of forest land to agriculture and livestock grazing. Forests were also harvested to meet the needs of a growing population. The unglaciated hills of south-central Indiana proved unsuitable for agriculture, and during the Great Depression degraded lands were abandoned and ultimately incorporated into state and national forests. Today, forest cover has returned to these lands, but vegetation communities still bear the mark of centuries of human disturbance.

INTRODUCTION

The wide expanse of hardwood forest in southern Indiana is perhaps the region's most defining feature. When considered in the context of the agricultural land to the north, the landscape of southern Indiana is more akin to the forests of the Appalachian foothills than to the greater Midwest. Although many factors have led to this contemporary dominance of forest, geologic history played a predominant role. The southernmost extent of the Illinoian glaciation reached into the northernmost portion of the region ca. 130,000 years BP and covered three of the nine present-day Hardwood Ecosystem Experiment (HEE) study units (Fig. 1). However, the maximum southern extent of the most recent glaciation, the Wisconsin of ca. 21,000 years BP, is 10 km (6.2 miles) northeast of the study area and did not cover any of the treatment units (Fig. 1).

This lack of recent glaciation resulted in a landscape consisting of steep hills and valleys with bedrock-derived soils. As discussed below, the rugged topography of this region ultimately led to the abandonment of agriculture and subsequent succession to the forests we see today. In addition, the varied topography of the region has fostered a range of microsite conditions and influenced the frequency and intensity of disturbances, such as fire. The interplay of these factors has contributed to the mosaic of vegetation observed across the contemporary landscape.

According to Homoya et al. (1985), the HEE study sites are all within the Brown County Section (BCS) of the Highland Rim Natural Region of Indiana. This section is characterized by deeply dissected uplands with bedrock composed of early-mid Mississippian siltstone, shale, and sandstone. Small high-gradient intermittent streams are common and drain into larger medium- to low-gradient streams. The process of erosion has produced a local topography that is generally steep, but without bluffs (Logan et al. 1922). All of the HEE study sites are underlain by soils of the Wellston-Berks-Gilpin soil complex (Natural Resources Conservation Service 2011). Wellston and Gilpin soils are classified as fine-silty, mixed, active, mesic Ultic Hapludalfs (Wellston) or Typic Hapludults

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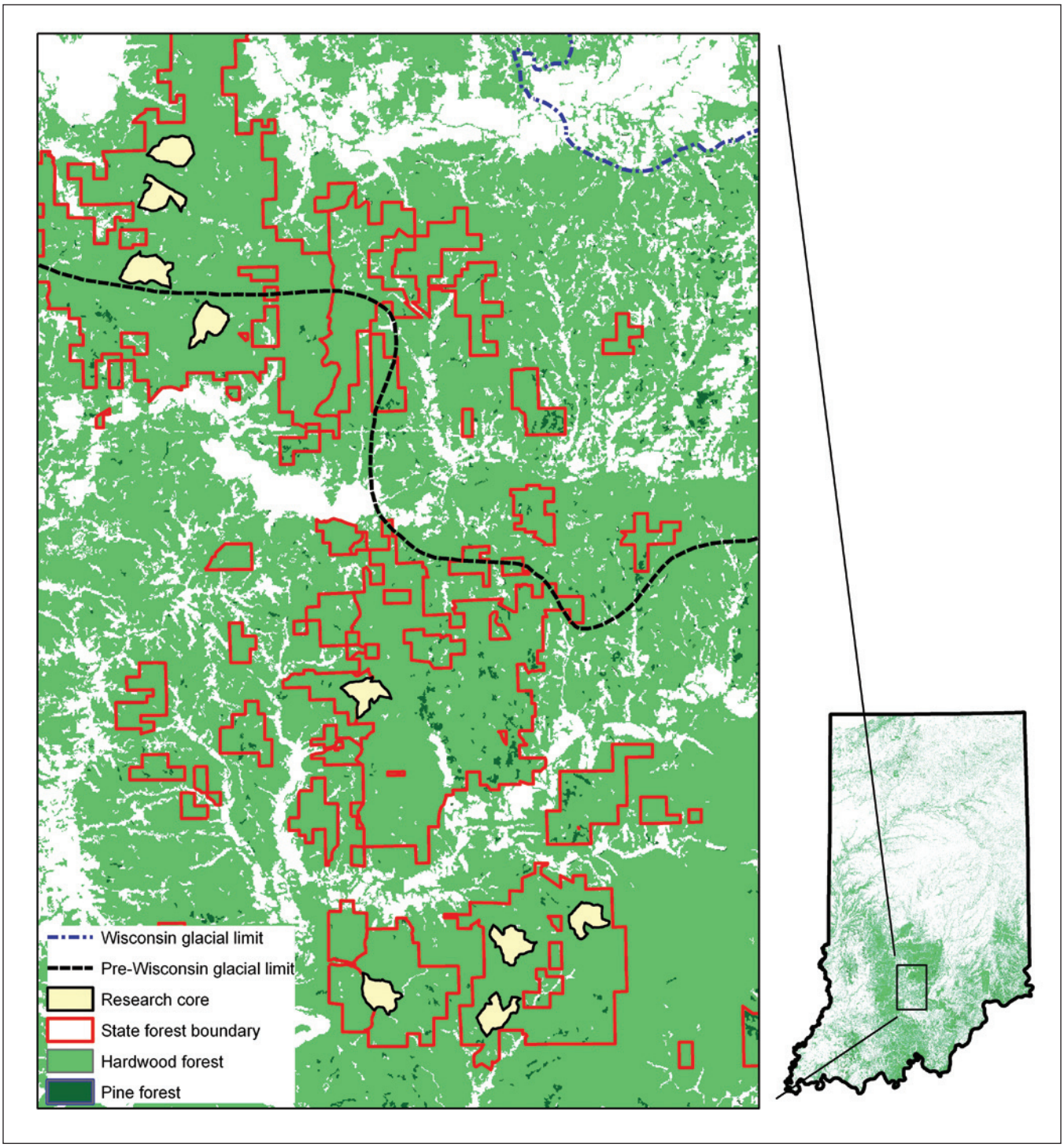


Figure 1.—Hardwood Ecosystem Experiment research core areas within the forest matrix of south-central Indiana. Southern limits of glaciation events are superimposed on the enlarged portion of the figure.

(Gilpin). Berks soils are classified as loamy-skeletal, mixed, active, mesic Typic Dystrudepts. These silt-loam soils occur on moderate to very steep slopes, are well drained, and are highly prone to erosion.

Uplands in the BCS are dominated by oak-hickory (*Quercus-Carya*) forests with chestnut oak (*Q. prinus*) on the most xeric topographic positions. Mesic slopes and ravines are dominated by red oak (*Q. rubra*), American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and white ash (*Fraxinus americana*). Common understory species include black huckleberry (*Gaylussacia baccata*), lowbush blueberry (*Vaccinium pallidum*), and greenbrier (*Smilax* spp.) on upper slopes and ridges. Sugar maple frequently dominates the understory of more mesic sites. Herbaceous species richness is generally low on dry ridges where Boott's sedge (*Carex picta*) typically dominates cover and is an indicator species of the landtype. A more diverse array of forest perennials, ferns, and sedges occurs on more mesic sites.

Disturbance is an ecological process in all ecosystems, but it occurs at varying spatial scales and intensities depending upon the inherent characteristics of an ecosystem. In forests, disturbance ranges in intensity, scale, and frequency from gaps formed regularly by the death of single or scattered trees to landscape-scale crown fires that occur at intervals of more than a century. In forests of eastern North America, including those of southern Indiana, humans have been the dominant drivers of disturbance regimes. In this paper, I discuss how the contemporary forest landscape of southern Indiana has been shaped by centuries of human activities.

DISTURBANCE BY NATIVE AMERICANS

Human occupation of Indiana dates to the end of the Wisconsin glaciation approximately 10,000 years BP, although these earliest inhabitants of the state were few in number and lived in a cooler and changing climate (Jones and Johnson 2008) within a landscape dominated by coniferous forests (Otto 1938). The coniferous forests retreated with the changing postglacial climate, and oak species became dominant approximately 7,000 years BP (Wilkins et al. 1991). More archeological sites exist for this period than for earlier periods because of growth in the human population and increased diversity of activities undertaken to more fully exploit resources that were available within a more varied ecological setting (Jones and Johnson 2008).

Native American populations continued to grow with the development of horticulture of selected plants and the formation of more permanent settlements during the Early Woodland period (ca. 1000-200 B.C). According to Baltz and Ledford (1991), 118 Native American archaeological sites have been found in the Mogan Ridge Area of Hoosier National Forest. Of these sites, 101 were of prehistoric origin. The link between early and protohistoric (post A.D. 1400) cultures in southern Indiana remains a topic of debate (Jones and Johnson 2008), but some scholars have suggested that the Miami and Shawnee, the dominant tribes prior to European settlement, may be linked to some indigenous late-prehistoric cultures (Sieber and Munson 1994).

Although earlier estimates of pre-Columbian Native American populations suggest that North America was largely unpopulated, more recent estimates of Native American populations suggest that North America was more populated than previously thought (DeVivo 1991). Twentieth-century estimates of the pre-Columbian population of North America ranged from 1.1 to 18 million (Denevan 1992, Thornton 2000). Population estimates for the land area that became the United States range from 720,000 to 7 million (Thornton 2000). Pre-Columbian population estimates are lacking at finer scales for most geographic areas, but archeological records do suggest that large river systems in southwestern Indiana contained well-developed population centers.

Native cultures of the Mississippian period (A.D. 1000-1650), evident in the archeological record of Angel Mounds State Historic Site and other sites in southwestern Indiana, practiced intensive agriculture and occupied well-populated cities, villages, and hamlets (Jones and Johnson 2008). The Angel Mounds site is estimated to have had a population between 1,000 and 3,000 at its peak and was linked through trade to other large population centers throughout the Midwest, including Cahokia in Illinois. However, Angel Mounds and other Mississippian sites were abandoned prior to European contact (Cobb and Butler 2002). Archeological evidence has not shown a link between the mound builders and tribes such as the Miami and Shawnee that occupied southern Indiana at the time of European contact (Jones and Johnson 2008, Sieber and Munson 1994). These later cultures were more mobile and did not occupy large permanent population centers, but they likely affected local resources through collection of fuelwood and nuts.

Beyond population density, whether Native Americans influenced their environment has been a subject of study and debate (Denevan 1992, Pyne 1982). Much like modern-day humans, pre-Columbian Native Americans used the tools at hand to create conditions that better addressed their sustenance needs. Burning and tree deadening (girdling by damaging the cambium) were used to clear land for agricultural use (Kellar 1966, Parker 1989). The impact of this disturbance was probably quite limited at any single point in time, but over the centuries the character of natural communities was most likely changed as areas were cleared, cultivated, and abandoned (Parker 1997, Parker and Ruffner 2004). The continuous shifts in land cultivation combined with the use of fire to clear new lands probably created a mosaic of various stand ages and forest types across the landscape (Buckner 1989, Guyette and Cutter 1991, Ruffner and Abrams 2002).

Burning also was used to drive or create conditions that attracted large animals such as bison (*Bison bison*) and white-tailed deer (*Odocoileus virginianus*) (Billard 1979). It is highly likely that these fires burned much larger areas than were intended, especially during drier years (Denevan 1992, DeVivo 1991, Parker 1997). According to survey records from the Government Land Office, presettlement forests (ca. 1820) of unglaciated southern Indiana were dominated by oak and hickory species (Lindsey et al. 1965, Potzger et al. 1956), which likely reflects these early land-use practices. The current overstory dominance of seral oak and hickory species in many old-growth remnants throughout the Central Hardwood Region is also probably the result of pre-settlement land use (Parker 1997).

Fire history records from southern Indiana are limited. In a barren-dry forest complex on the Hoosier National Forest, Guyette et al. (2003) observed regular burning prior to 1674, followed by a 128-year fire-free interval that persisted until 1801. The authors suggested that this interval may have resulted from the retreat of Native Americans from the Ohio Valley into Arkansas and Missouri ahead of Iroquois invaders. Fire frequency increased again after 1801 and frequent burning continued until 1849, likely due to eastern tribes migrating westward ahead of European expansion.

As the European population increased in the eastern United States, the need for new lands for settlement also increased (Sieber and Munson 1994). By 1790 there were already 70,000 European settlers in Kentucky, and many were eager to move into the lands of Indiana, Illinois, and Ohio. As settlers moved into lands formerly held by Native Americans, confrontation was inevitable. The years between 1790 and 1795 and from 1808 to 1814 were marked with numerous scattered fights between European settlers and Native Americans. Between 1790 and 1800, when the Indiana Territory was created, Native Americans controlled two-thirds of Indiana (Cockrum 1907). However, Native American dominance was greatly weakened when the Prophet and his Shawnee

followers were defeated at the Battle of Tippecanoe in 1811 (Indiana State Teachers Association 1914). With the ever-increasing number of settlers, defeat appeared inevitable, and most tribes surrendered their land through various treaties and were forced to move farther west.

EUROPEAN SETTLEMENT TO PRESENT

During the first two decades of the 1800s the population of Indiana grew rapidly, and statehood was granted in 1816 (Sieber and Munson 1994). The early settlers of south-central Indiana were mostly from Kentucky, North Carolina, Virginia, and Tennessee. Ethnically, the majority of these settlers were of English and Scots-Irish descent. The large influx of Scots-Irish settlers into the region made south-central Indiana culturally unique in comparison to the rest of the state. The ethnic heritage of the region's early settlers more closely linked them culturally to the early settlers of the southern Appalachians and the Missouri and Arkansas Ozarks (Rafferty 1980). According to Pyne (1982) the Scots-Irish of the settlement era had a cultural predisposition towards the broadcast burning of woodlands. This burning was conducted to reduce insect pests, improve hunting, and encourage the regeneration of hardwood species. In many areas, such as southwestern Missouri, this cultural burning continues today.

Early European settlers adopted the clear-and-abandon agriculture methods of Native Americans (Parker 1997). Most of these early farms operated at the sustenance level and were fairly ephemeral in nature (Otto 1983). Forests were most desired for farmland, since wetlands were difficult to drain and prairie was not considered good for farming (Urban 2005). Livestock (mostly hogs) were allowed to forage in the forest, thereby adding to disturbance through their feeding and trampling. Bradbury (1817) described land clearing by early settlers: "They have two modes of clearing land; one by cutting the trees around, so as to kill them, and afterwards clearing away the underwood, the quantity of which is very small: this mode is called girdling and is only resorted to by those, to use their own phrase are weak handed. The other mode is by cutting down the trees, dragging them into heaps and burning them."

Much of the land cleared by these early settlers was allowed to return to forest after it was abandoned as westward expansion continued. During the later stages of settlement, continued land clearing for farming and timber served to further reduce Indiana's once extensive forests to smaller, isolated, and heavily disturbed woodlands (Parker 1997). By the mid-1800s, large amounts of forest had been cleared, creating a highly fragmented landscape.

During this period of intensive settlement, numerous animal species of ecological importance became extinct or were extirpated from Indiana and the larger Midwest region. The last record of the passenger pigeon (*Ectopistes migratorius*) in Indiana was in 1893 (Lindsey 1977). Among large ungulates, bison and elk (*Cervus elaphus*) were extirpated from Indiana in the early 1800s (1808 and 1830, respectively) (Whitaker 2010). Though highly abundant on the contemporary landscape, white-tailed deer were extirpated in the 1880s and reintroduced in the 1930s and 1940s (Whitaker 2010). Although the effects of ungulates on vegetation have been widely studied, the historic ecological role of the passenger pigeon is poorly understood. Schorger (1955) estimated the total population of this species at 3 to 5 billion. Interestingly, Neumann (1985) suggested that prior to European contact, the passenger pigeon population in North America was much smaller due to competition with Native Americans for tree nuts. European settlement disrupted this competition and allowed pigeon populations to greatly expand.

As would be expected for such an abundant species, natural history accounts of the passenger pigeon abound from throughout its former range, including

southern Indiana. For example, after the population had been drastically reduced, the Indiana Gazetteer in 1849 described a place called Pigeon Roost in Scott County, IN, as "formerly so much resorted to by Pigeons, that for miles nearly all the small branches of a thick forest were broken off by their alighting in such numbers on them, and the ground was covered with ordure several inches in depth for years afterwards" (as cited by Wright 1911). Obviously, the roosting and feeding activities of this species had a profound effect on the structure and composition of forests.

The pattern of over-harvest and land abuse through excessive grazing, high-grading, and poor farming methods continued into the 20th century (Parker 1997). Continued agriculture in the region's hilly terrain resulted in massive erosion (Fig. 2); sheet erosion was estimated at 99 percent in parts of Crawford County and 92 percent in parts of Brown County (U.S. Geologic Survey data as cited in Sieber and Munson 1994). The farms on these steep hillsides were often referred to as "10-year land," which meant that they could be tilled for 10 years before enough soil eroded away to make further tillage unprofitable (Visher 1937). Woodlands that were not cleared for agriculture were still heavily cut to provide wood resources. However, the stands that regrew on these disturbed sites form most of the contemporary forest in the region.



Figure 2.—Eroded farmland in Brown County, IN, October 1935. Much of this highly degraded land was purchased from willing sellers and incorporated into state and federal forests. (Photo credit: U.S. Department of Agriculture, Farm Security Administration)

Many landowners in the region during the 1920s and 1930s were forced to abandon their holdings due to the poor condition of the land or their inability to pay back taxes (Sieber and Munson 1994). The severe degradation of land combined with the Great Depression plunged the region into a dire economic condition. These desperate conditions led to the depopulation of many areas, as reflected in population trends observed for Brown County. In 1900, the county had a population of 9,727, but by 1930 the population had declined to 5,168, a 47-percent decrease (Forstall 1995). The population of Brown County did not recover to its 1900 level until the 1970s. Following the abandonment of these lands, disturbance frequency and severity decreased due to fire suppression, reduced grazing, reduced farming on marginal lands, and the creation of public lands.

Reforestation has reduced the degree of fragmentation in large parts of the Central Hardwood Region. The incorporation of abandoned marginal farmland into the public domain has served to reduce fragmentation in areas such as southern Indiana, where large areas that were formerly dominated by hillside farms were incorporated into Hoosier National Forest and Indiana state forests during the early 1900s (Carman, this publication). Many landowners were quite willing to divest themselves of their eroded and depleted lands. Sieber and Munson (1994) reported that by August of 1935 the U.S. Forest Service received 2,000 offers to sell more than 80,940 ha (200,000 acres) of land. Throughout the Midwest during the 1930s, workers from the Civilian Conservation Corps and Works Progress Administration replanted large tracts of the most severely eroded ridgetops and stream bottoms with pine species, including white pine (*Pinus strobus*), red pine (*Pinus resinosa*), and shortleaf pine (*Pinus echinata*) (Otis et al. 1986, Sieber and Munson 1994) (Fig. 3).



Figure 3.—Works Progress Administration (WPA) crew planting 1-0 shortleaf pine on the Hoosier National Forest in 1940. Workers from both the WPA and the Civilian Conservation Corps planted pine on abandoned agricultural land throughout southern Indiana. Many of these plantations remain on state and federal land throughout the region. (Photo credit: U.S. Department of Agriculture, Forest Service)

The recovery of lands incorporated into public ownership has been dramatic. An examination of historic land use in the Charles C. Deam Wilderness of the Hoosier National Forest, an area located less than 10 km (6.2 miles) from the southernmost HEE site, found that, in 1939, 33 percent of the wilderness comprised abandoned agricultural land or old fields, 26 percent was open-canopy forest (indicating livestock grazing), and only 41 percent was closed-canopy forest (Jenkins and Parker 2000a). Although ridgetops and stream bottoms composed most of the agricultural land, nearly 20 percent of the slopes were also cleared. By 1990, 86 percent of the wilderness was covered by closed-canopy forest and the overall landscape was much less fragmented, as shown by a greater than 50-percent decrease in total edge.

Closed-canopy forest has returned to the landscape of southern Indiana, but vegetation communities in forests that have regrown on abandoned agricultural land differ from mature secondary forests. Jenkins and Parker (2000b) identified distinct differences in the herbaceous-layer composition of forests located on abandoned agricultural lands resulting from a loss of forest perennials and an influx of exotic and non-forest species. According to Jenkins and Parker (2001), overstory composition on abandoned agricultural land also differed from that of mature secondary forests due to heavy domination by tulip poplar (*Liriodendron tulipifera*). Due to the post-disturbance dominance by tulip poplar, however, overstory composition of abandoned agricultural land was very similar to that of clearcuts and group selection openings created in mature secondary forests.

The reduced frequency and intensity of fire and other disturbance in unmanaged forests since the mid-20th century have allowed succession of formerly oak-dominated forests to eventual domination by shade-tolerant species (Abrams 1992, Lorimer 1984, Ruffner and Groninger 2006). In southern Indiana, sugar maple typically dominates the understories of oak-dominated forests and likely will dominate future canopies in the continued absence of disturbance (Jenkins and Parker 1998). Single-tree selection harvest has been shown to accelerate succession toward dominance by sugar maple and other shade-tolerant species when overstory oak trees are removed (Abrams and Scott 1989, Jenkins and Parker 1998, Ozier et al. 2006). Understanding how the long-term loss of oak species will affect forest structure, processes, and trophic interactions is an important objective of the HEE.

The rugged land of southern Indiana that was incorporated into public ownership is typical of public forest land throughout the Midwest. Flatter, more fertile areas, such as north-central Indiana, that were forested in presettlement times have remained in agriculture. Thus, much of the present-day forest land in the Central Hardwood Region is found in areas with steeper topography, such as south-central Indiana, that were greatly disturbed by poor land-use practices. Consequently, the woodlands of today are probably not reflective of the wide expanses of forests that once covered the region. Instead, the forests of today, according to Parker and Merritt (1995), "represent the culled over remains of once magnificent forests and will continue to bear the scars of mismanagement well into the future."

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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.

INDIANA FOREST MANAGEMENT HISTORY AND PRACTICES

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Abstract.—Indiana's landscape and forests today are largely the result of Ice Age glaciations, Native Americans' use of fire, and over-harvesting in the late 19th and early 20th centuries. Any intentional management of the forest was not generally apparent until the early 1900s. Early visionaries at that time recognized the future impact forest depletion would have on the state's well-being and enacted legislation providing tax incentives to maintain land in forest cover. Some even-aged management has been applied to Indiana's forests and continues to be to some extent, but uneven-aged management is more prevalent, particularly on public lands. Although Indiana has many tree species of economic and ecological significance, a major focus of forest management today is on the regeneration and perpetuation of the oak species.

FROM GLACIERS TOWARD STATEHOOD

More than 700,000 years ago Indiana's landscape was dominated by glaciers that began moving through the area during the Ice Age (Fleming 1997). Moving slowly from north to south, these massive sheets of ice scraped away all land forms that impeded their progress. These glaciers completely altered surface drainage, created lakes, diverted stream channels, and filled large valleys with sediment, leaving little record of their existence. Also swept away in their paths were coniferous forests that prospered in Indiana due to the colder climate.

Although archaeological records for Indiana date back as far as 14,000 years, the first evidence of human habitation appears about 10,000 B.C. with the Paleoindians. By the time European settlers arrived, Native Americans were socially well organized and dependent upon intensive agricultural production, growing crops such as corn, beans, and squash that are familiar today. Towns were developed and connected through trade routes both over land and through rivers (Justice 2006). Cultivation of trees as a crop was still centuries ahead, yet it is likely that the people of this period used the forest resource for dwellings and heat, and perhaps did primitive management to enhance hunting. Jenkins (this publication) provides a more detailed account of prehistoric and presettlement forest conditions in Indiana.

The first Europeans to discover and occupy Indiana were the French, some in search of wealth from beaver pelts and some serving as missionaries (Troyer 1975). They arrived during the 1670s and established their first permanent settlements around 1720 near present-day Fort Wayne, Lafayette, and Vincennes.

Native Americans had lived in this region for thousands of years but had limited interaction with Europeans; most of their encounters occurred in the few decades prior to the Revolutionary War (Sieber et al. 1992). European settlers and their descendants came to the area in far greater numbers after the Revolutionary War. The Land Act of 1800 made it easy for private individuals to acquire federal land, and the first land office in what is now Indiana was opened at Vincennes in 1807.

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INDIANA'S GROWING PAINS: 1810-1900

By 1810, the Indiana territorial population had increased to about 24,000. When peace with European countries prevailed after the War of 1812, Indiana became a state in 1816 and most of central Indiana was open for settlement. Early settlers to Indiana were primarily subsistence farmers who cleared land for crops and pasture much as their ancestors had done to shape the European landscape centuries earlier. Their introduction to this country of metal tools and farming implements along with draft animals enabled them to more easily overcome the perceived barrier to progress that the forest represented.

The new state's population had increased to more than 147,000 by 1820. Road building began in 1832 with the clearing of trees for the Old Michigan Road, a rugged, tree-stump-strewn travel lane connecting the town of Madison in southern Indiana and Michigan City on the state's northern border. Thus began two decades of intensive expansion of transportation routes. In addition to these primitive roads, Indiana transportation was enhanced by steamboats, canals, and railroads during this time. By 1840 settlements were appearing throughout northern Indiana, particularly along the route of the Wabash-Erie Canal. When the Civil War began in 1861, Indiana ranked 6th in population among the 34 United States, and ranked in the top 3 states for the production of wheat, corn, potatoes, hogs, cattle, and sheep (Troyer 1975).

As Indiana's population increased and settlements continued to grow, the demand for lumber during the early to mid-1800s escalated rapidly. Most homes were built of wood and required 20 to 40 cords of wood annually for heating and cooking, and as many as 8,000 fence rails were needed to enclose a 16-ha (40-acre) field (MacCleery 1992). Sawmills played a key role in communities and laid the foundation for the state's economic growth. It was not uncommon for entire communities to work together in constructing new mills. Sawmills were powered by a variety of sources. Water was the principal source of power through the 1840s (Clark 1987), but in the 1850s steam power began to allow mills to venture away from flowing water into the woodlands. Cleared land was generally considered more valuable than land in forest cover, but a territorial law to protect unauthorized timber cutting appeared as early as 1799, when an $8 fine was imposed "for felling or boring any walnut, oak, whitewood, poplar, cherry, ash, locust, chestnut, coffee or sugar tree on another person's land without permission" (Pease 1925: 362). After acquiring statehood, Indiana's legislature passed similar regulatory laws adding further penalties for unsanctioned timber harvesting.

Prior to the Civil War, demand for Indiana hardwoods was mainly for construction and infrastructure. While these demands for durable products continued after the war, consumer interest also turned to products with both functional and aesthetic appeal (Clark 1987). Many species of hardwoods were used, but oak (*Quercus*), walnut (*Juglans*), maple (*Acer*), poplar (*Liriodendron tulipifera*), hickory (*Carya*), cherry (*Prunus serotina*), elm (*Ulmus*), and ash (*Fraxinus*) were of principal commercial value. The early Reconstruction period following the Civil War was a time of widespread change. Industries were busy shifting to peacetime production, subsistence farmers were being drawn to jobs in logging and sawmills, and Indiana's network of railroads was expanding (Hicks 1997). The popularity of the steam tractor in the 1870s (White 2010), coupled with genetically improved plants and the availability of fertilizers, led to greatly increased crop production per hectare. Some rugged lands that were tillable with a team of draft animals were left to revert to forest cover because they were too steep to navigate with a tractor and were marginally productive. Forest land was consequently regenerated in some areas, but those gains were offset by other factors. The building of more factories and homes not only increased the demand for construction lumber, but meant a greater demand for coal, oil, natural gas, and later gasoline to fuel the machines and vehicles. Forest land was lost to mining, pipelines, road construction, and other infrastructure (Hicks 1997).

Few people were concerned at the time that Indiana's forest resource might be in jeopardy. It wasn't until 1874 that the idea of a tax incentive related to forestry was first mentioned by M.B. Kerr in the Indiana Agricultural Report. Kerr noted that soil erosion and timber depletion were serious problems in many areas of southern Indiana and recommended that black locust (*Robinia pseudoacacia*) be planted in steep and eroded areas. He went further to recommend that tax exemptions be granted on lands that were planted to forest, at least for the first 10 years of the plantation. By 1880 Indiana's timbered area had shrunk to about 1.7 million ha (4.3 million ac), and by the end of that decade it would be depleted even further to about 607,000 ha (1.5 million ac).[2]

Indiana led the nation by the turn of the 20th century in the production of oak and walnut, with nearly 700 million board feet and 11 million board feet, respectively, produced annually. Due to the slow rate of secondary growth, this intense demand predictably led to a steady decline in hardwood availability. Government-funded research brought about more efficient milling and manufacturing techniques, which sustained the hardwood industry, but pressure on Indiana's forest resource took its toll. Indiana was able to maintain a sizable portion of the walnut market nationally, but production of other species declined rapidly. By 1900, Indiana's overall forest resource was reduced to a mere shadow of what it once was (Table 1).

[2] Data on file with the Indiana Department of Natural Resources-Division of Forestry, Indianapolis, IN.

Table 1.—Forest land area in Indiana, 1800-2008.

Year	Hectares	Acres
1800	7,891,650	19,500,000
1860	4,047,000	10,000,000
1900	607,050	1,500,000
1950	1,675,458	4,140,000
1967	1,604,352	3,964,300
1986	1,796,544	4,439,200
1998	1,821,676	4,501,300
2008	1,932,523	4,775,200

Near the end of the 19th century, several key developments would positively affect the early recovery of Indiana's forests. The Organic Administration Act of 1897 was federal legislation that made possible the formation of most of our National Forests. This law states, "No national forest shall be established, except to improve and protect the forest within the boundaries, or for the purpose of securing favorable conditions of water flows, and to furnish a continuous supply of timber for the use and necessities of citizens of the United States". These stipulations clearly set National Forests apart from National Parks, which were created to preserve the beauty of those properties and provide for recreational opportunities.

In 1898 a resolution was drafted by the Indiana Horticultural Society and forwarded to the Indiana General Assembly recommending a tax exemption for lands maintained strictly as forests, or at least tax adjustments to be made with appraisals for tax purposes based on the annual income accrued by the forest lands. Even though this resolution did not result in immediate legislative action, it undoubtedly had an impact on legislators as sentiment for support of forestry continued to mount.

The next piece of relevant state legislation came in 1899. At the urging of the Indiana Academy of Science and others, Indiana's state legislature passed "An Act for the Encouragement of Forestry" (Kriebel 1987), a more limited forerunner of today's Classified Forest program. Landowners were given a substantial tax reduction if they agreed to cut no more than 20 percent of their timber, plant a tree for every tree cut, and limit the grazing of livestock in their woods. This incentive could be applied to only one-eighth of the forested land they owned. This year also marked the formation of the Indiana Hardwood Lumbermen's Association, a group that would influence forest management and policy for decades to come.

With guidance from the National Committee on Wood Utilization and research from the U.S. Forest Service's Forest Products Laboratory, advances in milling

efficiency and forest management began to take shape. Cutting lumber to exact dimensions at the primary mill, for example, allowed the product to be received at the secondary mill ready to use, thus eliminating the inherent waste involved in remanufacture at the factory. This practice also enhanced utility of the material harvested and therefore increased the use of smaller logs. These efficiencies led to lower mill costs, less waste, and the development of new markets for lesser-quality lumber. It also demonstrated that the physical properties of second-growth timber were not inferior to those of virgin timber, refuting a commonly held belief.

INDIANA FOREST MANAGEMENT: 1901-2000

In 1901 the Indiana Board of Forestry was signed into law by Governor Winfield Durbin with the charge "to collect, digest and classify information on forestry and recommend plans and methods for the better practice of forestry and for the establishment of state forest reserves" (Clark 1987). At that point in time, there was not much forest land left to manage. Severely eroded lands needed to be reforested and habitat for wildlife restored. Early forest management focused on these efforts as well as public education. Private forest landowners were provided publications about tree planting, keeping livestock out of the woods, and growing trees for fence posts. On-site landowner assistance was also offered on a very limited basis.

Committing land to the public trust began in 1903 with passage of Indiana Code 14-23-4-1, allowing for the establishment and management of State Forests (Indiana Code 2011):

> It is the public policy of Indiana to protect and conserve the timber, water resources, wildlife, and topsoil in the forests owned and operated by the division of forestry for the equal enjoyment and guaranteed use of future generations. However, by the employment of good husbandry, timber that has a substantial commercial value may be removed in a manner that benefits the growth of saplings and other trees by thinnings, improvement cuttings, and harvest processes and at the same time provides a source of revenue to the state and counties and provides local markets with a further source of building material.

With this legislation came authorization for the state to purchase 809 ha (2,000 ac) of land in Clark County to serve as a forest laboratory, demonstration area, and tree seedling nursery. This property would later come to be known as Clark State Forest. Its condition upon purchase bore little resemblance to its beauty today. Having been heavily logged and scarred by fires that had occurred there almost yearly, Indiana's first State Forest had few large trees. Initial work focused on erosion control, tree planting, pruning, and establishment of roads and other infrastructure. A seedling nursery was established by 1907 to provide trees mainly for the property, with the intent of eventually making trees available to the public. This "Forest Reservation and Experiment Station," as it was known then, would remain Indiana's only State Forest for 26 years (Indiana Department of Natural Resources-Division of Forestry 2006).

Bluffton, IN, native Charles Deam was appointed as the first state forester in 1909. Deam was a druggist by trade but had a passion for studying plants. His achievements in botany are legendary. His botanical collection numbered some 78,000 specimens, of which 25 species were discovered by Deam. He also has 48 species that bear his name (Kriebel 1987). In the forestry arena, one of Deam's lasting legacies was writing a revised Forest Tax Classification Act, which the Indiana state legislature passed in 1921. Among other restrictions, this revised law eliminated the percentage limit on how much forest land could be classified, allowing for thousands more acres to be assessed at only $1 per acre in exchange for a written forestry management agreement. In 1925 a property tax increase of one-half mil on each $100 of taxable valuation was passed, which resulted in funding for

more forest management, including the employment of professional foresters. This legislation was known as the LaFuze Act, named after its sponsor, State Senator Oliver LaFuze. Its impact would prove to be enormous for the state Division of Forestry's operations (Bramble 1965).

Two important pieces of legislation at the federal level were passage of the Weeks Act of 1911 and the Clark-McNary Act of 1924. The Weeks Act authorized the Federal government's purchase of land in the eastern United States. Although the intent of these purchases was to protect the headwaters of navigable streams, much of the land purchased eventually became the core of what are now National Forests. The importance of the Clark-McNary Act of 1924 was to establish the groundwork for federal–state forestry cooperation, particularly in areas of fire control and working with private landowners. It also mandated the production of timber for National Forests (Hicks 1997).

A major effort was begun in the late 1920s by federal, state, and private forestry entities and by Purdue University to encourage sustained yield forest management, limit or eliminate livestock grazing in woodlands, and protect forest resources from fire. Improved forestry and logging practices were demonstrated to landowners and clearcutting practices were discouraged in favor of single-tree selection.

Increased funding from the mil tax allowed the state to hire additional professionally trained foresters. Financial hardships of the 1930s accentuated the depleted condition of Indiana's natural environment. Many farmers were unable to meet expenses, and those trying to make a living from the poorer land in hilly southern Indiana were not even able to pay taxes. Mortgaged lands were repossessed by banks, and other abandoned lands reverted to county ownership, much of it later to be converted to state and national forests (Troyer 1975). By 1932 the number of State Forests had increased to five (Table 2). Included in the State Forests' 10,120 ha (25,000 acres) was Morgan-Monroe State Forest, established in 1929. Much development of those properties took place at the hands of the federally funded Civilian Conservation Corps (CCC) during the 1930s. Land acquisition also began during this decade for the development of what would later become the Hoosier National Forest in 1951.

Any history of Indiana's forest resource would be incomplete without mention of the role of the CCC. Authorized by Congress in 1933 as one of President Franklin D. Roosevelt's New Deal programs, the CCC had two intended purposes: to provide employment for thousands who were out of work, and to improve and restore the country's natural resources, which were badly degraded. CCC projects included forest fire control, building construction, erosion control, dam construction, and trail building (Otis et al. 1986). Of

Table 2.—Establishment dates of Indiana State Forests (Booneville Enquirer 1933; Indiana Department of Conservation 1934, 1935, 1936, 1938).

Establishment Date	State Forest Name
1903	Clark
1929	Morgan-Monroe
1931	Jackson-Washington[a]
1932	Harrison-Crawford[b]
1932	Martin
1933	Scales Lake[c]
1934	Ferdinand[d]
1935	Pike
1935	Wells County[e]
1935	Salamonie River[f]
1936	Green-Sullivan
1939	Frances Slocum
1942	Selmier
1947	Yellowwood[g]
1948	Owen-Putnam

[a] Originally known as Jackson County State Forest.
[b] Originally known as Harrison County State Forest.
[c] Became a Warrick County park in 1967.
[d] Originally known as Dubois County State Forest.
[e] Became a State Recreation Area in 1962 and a State Park in 1983.
[f] Originally known as Wabash County State Forest.
[g] Previously leased from the U.S. Department of Agriculture since 1940.

particular note in Indiana, aside from the planting of millions of trees, was the timber stand improvement (TSI) and logging on State Forests. The CCC operated sawmills and planing mills that processed the lumber used to build the many barns, service buildings, and residences on state properties, many of which are still in use. When the CCC program came to an end in 1942, the United States was in the midst of World War II. The combination of a lack of manpower due to the war and the sudden loss of CCC labor resulted in a critical shortage of foresters, fire personnel, and property workers.

In 1944 a long-range forest management plan was developed for Indiana, and implementation of this plan received a boost a year later with an increase in the forestry tax to 5 mils. State Forester Ralph Wilcox expanded the number of forestry districts throughout the state to 10 and hired additional staff to meet the public demand for forest management assistance (Bramble 1965).

As soldiers from World War II returned home to a stimulated economy and began working in factories and raising families, a marked shift in forest landowner objectives occurred. Land that for decades had been "working forests," managed for production because it was considered essential for earning a living, now became places primarily held for recreation and aesthetic enjoyment (Hicks 1997). This characteristic of land ownership has carried through to the present day, as will be discussed later.

Moving into the 1950s, Indiana still suffered from a diminished supply of hardwood lumber. That, in combination with advances in metals and plastics technology, resulted in an overall depressed timber industry. More than 1,000 sawmills still operated in the mid-1950s, but only 30 had an annual production exceeding 1 million board feet. Ten years later, 68 percent of Indiana's mills were sawing less than 350,000 board feet of lumber per year (Clark 1987).

The forestry effort in Indiana was dealt a devastating blow in 1953, when the state administration suddenly ordered the firing of four district foresters and two nursery foresters. Working conditions deteriorated to the point that three additional foresters resigned, leaving only two foresters to provide services to a 42-county area in southern Indiana. Extension foresters from Purdue University offered assistance when possible, and even conservation law enforcement officers were assigned to conduct classified forest inspections. Conditions did not improve until 1957, when a new administration began hiring back some of the foresters previously dismissed and recruiting new foresters. Even then, improvements were very slow as it took time to rebuild positive relationships between professionally trained foresters and state government. When Robert Raisch became State Forester in 1962 under the supportive Department of Conservation leadership of Director Don Foltz, a new era of professionalism and growth began, expanding personnel and programs beyond what they had ever been (Bramble 1965).

Reduction in timber harvesting on federal lands, loss of forest land due to development, and increased demand for wood fiber created mounting pressure on non-industrial private forest land to produce timber. This pressure provided impetus for even greater support for forest management, both at the state and federal levels. In 1973 the Forest Incentives Program was initiated. This program covered up to 65 percent of the cost for planting trees or doing TSI, to a maximum of $10,000 per year. To participate in the program, landowners were required to maintain the practices for at least 10 years (Jacobson et al. 2006).

Patterned after the 1950s Federal Soil Bank program, the Conservation Reserve Program of 1985 encouraged farmers to plant trees on what was previously cropland (Hicks 1997). Land was typically enrolled in the program for 10 to 15 years, during which time the Commodities Credit Corporation made

annual payments based on the land's agricultural rental value. Landowners were also reimbursed up to 50 percent of their cost of planting trees or implementing other approved conservation practices on the land (U.S. Department of Agriculture 2007).

Downturns in the economy led to diminished federal funding of forestry programs in the late 1980s and throughout the 1990s. Most state forestry positions added when federal funding was plentiful were retained, but no new positions could be added and attrition led to the loss of some positions. "Streamlining" became the norm for doing business in both the private and public sectors.

On public forest land, there was greatly increased emphasis during the 1990s on forest values other than timber. Management decisions increasingly took into consideration wildlife populations, recreation, historic preservation, and other aspects of the forest. A more holistic approach to forest management evolved throughout that decade and continues today. The focus of federal forestry incentive funding was reflective of this change in approach. The Forest Stewardship Act of 1990 provided for funding of the Stewardship Incentive Program and the Forest Stewardship Program. Both of these long-term management programs valued not only timber production, but also the creation and maintenance of wildlife habitat, watershed protection, recreation, and aesthetics. The Stewardship Incentive Program was replaced in 2002 by the Forest Land Enhancement Program, which continues to be one of the most highly regarded incentive programs by both landowners and foresters for enhancing forest management and helping landowners meet their objectives (Jacobson et al. 2006).

MODERN FOREST MANAGEMENT

The greatest threats to both public and private forest land in Indiana through the 19th and early 20th centuries were unregulated over-harvesting and clearing for agriculture. Today, Indiana's forests face a number of challenges, perhaps the greatest of which are invasive species and forest fragmentation.

Garlic mustard (*Alliaria petiolata*), tree of heaven (*Ailanthus altissima*), bush honeysuckle (*Lonicera maackii*, *L. tatarica*, *L. morrowii*), and kudzu (*Pueraria lobata*) are but a few of the many invading plants that are displacing native species from their natural habitats. Insects such as the gypsy moth (*Lymantria dispar*) and emerald ash borer (*Agrilus planipennis*) are putting some of our most economically valuable tree species in jeopardy. Invasive species not only threaten the biodiversity of Indiana's hardwood forests but cost an estimated $120 billion in damages and control efforts (Pimentel et al. 2005), which does not take into account losses from tourism and recreation.

To put in perspective the potential impact that one of these alien invaders might have on Indiana's landscape and economy, we do not have to look far back into our past. *Cryphonectria parasitica*, the fungus that caused the chestnut blight, was first introduced into New York in 1904. Within 10 years it had spread to most states east of the Mississippi River. By the late 1930s, few mature American chestnut (*Castanea dentata*) trees were still living anywhere in the eastern United States. Within one generation, a tree whose lumber was worth millions of dollars and whose value as a wildlife food source was immeasurable was virtually wiped out (Hicks 1997). A similar urban story can be told of the impact of Dutch elm disease on the American elm (*U. americana*). Once a mainstay of the neighborhood landscape, the beautiful and abundant American elm was reduced to firewood.

The conversion of forest land to development has resulted in thousands of hectares of forest land being eliminated or seriously compromised. The fragmentation of forest land continues to create

problems for the management of both timber and wildlife. Forest fragmentation has been identified as a probable cause for the decline of several migratory bird species. Because fragmented forests greatly increase the amount of edge, Brown-headed Cowbirds (*Molothrus ater*) are able to more readily parasitize migrants' nests (Robinson et al. 1995). Aside from wildlife concerns, fragmentation also affects the forest structure itself, leading to changing tree associations and forest composition. The additional edge of a fragmented forest favors an increase in shade-intolerant tree species.

Another phenomenon different from fragmentation but often related is forest parcelization, or the division of ownership of a particular area of contiguous forest. Although parcelization may result in forest fragmentation, even parcelized land that remains as contiguous forest may well be impacted. Smaller parcels of land having different owners are likely to have homes and other structures built within them, or to have the forest managed differently based on differing landowner values and objectives.

Although the most recent Indiana forest inventory shows an increase in timberland throughout the state (Indiana Department of Natural Resources [IDNR]-Division of Forestry 2010), landowners' interest in marketing their timber appears to be declining. In a study conducted by the Center for Nonprofit Strategies, forest landowners in six Midwestern states including Indiana were surveyed and grouped into four categories according to their objectives for owning forest land. Only 26 percent of those private landowners fell into the "Working the Land" group, compared with 34 percent (the highest for any individual group) who were categorized as "Woodland Resort Owners".[3] As recreational and aesthetic values of private forest land supersede production values, the availability of marketable timber could decrease in the coming years.

[3] Data on file with the Indiana Department of Natural Resources-Division of Forestry, Indianapolis, IN.

In many respects, the history of active forest management in Indiana has been synonymous with the management of the oak species. Although walnut, maple, cherry, ash, and many other hardwood species certainly have economic and environmental values, the oaks are considered the most important aggregate of hardwoods found in North America (Harlow et al. 1996 from IDNR-Division of Forestry 2008a).

Oaks have a low tolerance of shade, so are generally early- to mid-successional species. Most oak seedlings can survive under low light conditions if food reserves stored in the cotyledon of acorns are adequate; but once those reserves are depleted, light becomes the limiting factor (IDNR-Division of Forestry 2008a). As oak mortality occurs, competing species such as sugar maple (*Acer saccharum*) and yellow-poplar take their place. If advance regeneration can occur under a closed canopy, followed by a disturbance that removes overstory to increase light to the forest floor, the chances of oak seedlings' successfully competing are enhanced. Without large-scale disturbance, the oaks have difficulty maintaining their dominance.

Native Americans used fire to convert forest land to an earlier successional stage, thus encouraging the growth of forbs and herbaceous cover and increasing wildlife habitat for hunting. Early settlers cleared forest land for agriculture, grazing, and settlement. All of these practices created disturbance that favored the regeneration of oak species. As the human population increased, however, large-scale fires were suppressed, giving the competitive advantage to more mesophytic tree species.

Today's Indiana woodlands have been shaped by disturbances far different from those of presettlement times. A great deal of data suggests that mature and over-mature oak-hickory stands are shifting to other forest types due to diminishing regeneration (IDNR-Division of Forestry 2008a). Because oaks generally grow more slowly than their competitors and are shade- and flood-intolerant, oaks are at a distinct

disadvantage. Aside from the lack of landscape-altering fire, management practices on both public and private land over the past several decades have not been particularly conducive to oak regeneration. High-grading on private forest land and relatively low levels of harvesting on public lands using principally single-tree selection have resulted in conditions unfavorable to oak regeneration. High-grading on private land can remove canopies before advance regeneration of oak seedlings occurs. Selective harvesting on public land allows little or no additional light through the canopy. Without advance regeneration, faster growing, more shade-tolerant species will quickly outcompete the oaks. Although oak seedlings are still present in Indiana forests, competitors outnumber them 4 to 1 statewide (IDNR-Division of Forestry 2008a). It is highly unlikely that natural regeneration will result in the oaks' return to dominance in an undisturbed stand. Intentional management will be required if Indiana's forests are to have a dominant presence of mature oaks in the future.

In recent years, prescribed fire has increasingly been used to reduce competition with oak seedlings. One adaptation that oaks have in their favor is rapid root growth. Although shoot growth of oak seedlings is very slow, oaks' roots grow faster than those of most of their competitors. Dormant buds near the root collar allow oak seedlings to resprout after top kill from fire or deer browse (IDNR-Division of Forestry 2008a). Each resprouting strengthens the root system and adds mass to it so that when disturbance increases the level of light, the shoot responds with vigorous growth. Mechanical removal of competing vegetation, soil scarification, and the application of herbicides are additional practices being used to eliminate competing tree species. Perhaps the most effective means of re-establishing forests with an oak dominance is through silvicultural practices that allow for enough light to reach oak seedlings. Combining selective harvesting with creating openings, small group selections and shelterwoods are being used to create conditions of more light without completely opening up the canopy.

Of course, not all forest management is focused on perpetuating the oaks. Public lands are managed for a multitude of purposes, including recreation and aesthetics. An uninterrupted tree canopy is important to visitors, providing a "natural" look to the forest. The varied tree species composition and forest structure of an uneven-aged stand also enhance wildlife habitat diversity and aesthetic appeal of the forest. Single-tree selection is used predominantly to maintain the uneven-aged stand, with group selection harvests employed when the goal is to open an area for shade-intolerant species. Group selection openings of at least 0.10 ha (0.25 acre) in size on south-facing slopes and 0.20 ha (0.50 acre) in size on north-facing slopes are generally used, with east and west slope openings of about 0.13 ha (0.33 acre) (IDNR-Division of Forestry 2008b). Group selections are generally not applied in areas where trees exhibit good health and vigor, but rather where a thinning to release trees of good quality is no longer possible. After approximately 10 years, TSI is done to the area to release crop trees, and prescribed fire may also be used if the principal crop trees in the area are oaks.

Creating a shelterwood situation is another option, removing the lower canopy of shade-tolerant trees by using prescribed fire or killing the lower canopy trees using TSI along with herbicides. This approach is generally used when there is an adequate number of seed trees or where it is practical to underplant seedlings in openings. Once the planted or regenerated seedlings have reached sufficient size, competition from overstory trees can be eliminated by removal or TSI.

When the goal is to improve development of existing trees without focusing on new seedling establishment, intermediate cuttings may be used. The practice may be a commercial harvest, TSI operation, or a combination of both. In any case, defective or otherwise undesirable trees are removed from the stand to improve overall stand quality. Whether management is even- or uneven-aged, thinning is an

intermediate cutting practice used to decrease stand density. Selection thinning removes dominant or co-dominant trees to release desirable trees in the lower canopy. Geometric thinning, most often used in a plantation, removes trees in a predetermined pattern. Free thinning is the combination of more than one thinning method (IDNR-Division of Forestry 2008b).

Deciding what combination of the above choices to use requires evaluating trees for vigor, canopy position, relationship of species with the site, best use of site space, and desired future development of the site. The limitations and potential of each species must be considered. Evaluation of the impact of surrounding trees must also be done to prevent possible damage to a crop tree during harvest or TSI operations. Aspect as related to the position of the trees that are to remain must also be taken into account. Removing or killing a tree on the south side of a crop tree, particularly on a north-facing slope, will yield greater benefit to the remaining crop tree than removing a tree on the north side of that crop tree.

Although management for factors other than timber production certainly occurs on public land, examples of these practices are quite common on private lands. Management to maximize mast production for wildlife, or to enhance cover or nesting sites, might lead a landowner to favor certain tree species over others regardless of timber value. Aesthetics and recreation are often cited as reasons for owning forest land, so trees with a form that might otherwise be considered undesirable might be retained for these values. In these situations landowners still employ selection thinning, simply basing their tree selections on different criteria.

Because of the greater emphasis today on the aesthetic and recreational potential of forest land, citizens have become a much more active voice in formulating forest management policies. The technical expertise of a professionally trained forester is still a valuable component in forest planning, but the forester now serves as just one member of a team in determining policy. To better facilitate this process, the Indiana Forest Stewardship Coordinating Committee was established. This committee brings together a wide range of individuals, organizations and agencies who share a common interest in the health and vigor of our forests, but who may differ on how that should best be achieved. The Forest Stewardship Coordinating Committee meets regularly to keep the lines of communication open and promote better understanding of forest management issues. The guiding principles of the group are spelled out in the Indiana Statewide Forest Strategy, a document that reflects a consensus of opinions gathered from forest stakeholders, both public and private (IDNR-Division of Forestry 2011).

The history of Indiana's forest resources has been one of drastic and relatively abrupt changes over the past 200 years. Forests covered more than 8.4 million ha (21 million acres) of what was to be Indiana when settlers arrived, but accounted for only 607,000 ha (1.5 million acres) as recently as 100 years ago. Previous Indiana inhabitants may have viewed the forest as an endless resource to be exploited, but we have come to appreciate the forest for its many benefits, including watershed protection, wildlife habitat, recreation, air purification, and fiber production. We also recognize the importance of managing our forests with the best science at hand so their many benefits will be enjoyed for generations to come. The fact that there is more than three times the amount of forest land today than there was 100 years ago is testament to the effectiveness of management.

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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.

INDIANA RESIDENTS' PERCEPTIONS OF WOODLAND MANAGEMENT

Daniel J. Witter, Shannon M. Amberg, David J. Case, and Phillip T. Seng[1]

Abstract.—A 2009 telephone survey of 1,402 Indiana adults was conducted to assess opinions regarding woodland management. Forty-eight percent said they were "very concerned" about the health and productivity of Indiana's woodlands, and 45 percent, "somewhat concerned." Almost half (47 percent) thought that the state's woodlands are held in about "equal government and private ownership," and 26 percent thought most woodlands are "government" owned (in fact, roughly 85 percent of the state's forest lands are privately owned). When informed that "Indiana's State Forest System makes up about 3 percent of Indiana woodlands," a majority (55 percent) thought that this amount was "not enough," and 39 percent, "about right." Forest practices receiving majority approval were: "removing some trees to protect Indiana woodlands from spread of disease and wildfire" (95 percent), "harvesting Indiana trees for woodland management if overseen by professional foresters" (85 percent), "harvesting Indiana trees to improve places for wildlife to live" (82 percent), "advising Indiana private landowners on how many and what kinds of trees they might harvest and sell" (70 percent), and "harvesting Indiana trees to make lumber or other wood products that we use" (61 percent). Majority agreement was given to the statement, "Indiana woodlands should be managed for a balance of wood products that we use, and other benefits like recreation, wildlife, and good water quality" (88 percent). Nearly three-fourths (72 percent) agreed with the statement, "The amount of forested land in Indiana is shrinking," and respondents identified "urban sprawl" as the highest-ranking threat to Indiana woodlands.

INTRODUCTION

Forestry professionals have long been interested in public opinions about woodland management in the United States. Insights into the "human dimensions" of forest conservation help managers integrate public perceptions into forestry programs and focus outreach strategies and messages about forest management. However, public opinions about how forests are managed are complex and lack homogeneity across stakeholders and geographic regions. Citizen sentiment on forest management ranges from indifference to zealotry and from staunch support for certain forest practices to strident opposition (Witter and Jahn 1998).

Several studies across the United States show that the public values non-commodity benefits of forests more than economic benefits. For instance, residents in Vermont ranked aesthetic, recreational, and ecological values as most important in forest management, and economics as one of the least important (Manning et al. 1999). In a study of southern states, Tarrant et al. (2003) found that people valued aesthetic qualities, such as clean air and scenic beauty, much more than wood production. Research also indicates that people are more supportive of various forest management techniques if the goals are focused on alleviating ecological concerns. For example, a national study of public opinions revealed that a majority of people favored management strategies that protect old growth

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forests and wildlife habitats over commodity-based strategies (Shindler et al. 1993).

A number of studies have attempted to better understand consumer attitudes towards certified forest products, though they did not investigate public perception of certified forest management (Ozanne and Smith 1998, Ozanne and Vlosky 2003). Hubbard and Bowe (2005) found that some forest firms perceived they gained increased credibility with the public after becoming certified. In addition, in a series of reports about sustainable forest management, Sheppard et al. (2004) stated that sustainable forest management should also be sustainable in a social sense, and thus incorporate the public's values into management.

Indiana supports an impressive forest resource that is valuable both for its ecological benefits and for the commodity-based opportunities it provides. "The Indiana forest products industry is the sixth largest manufacturing industry in the state, employing over 56,000 people. Forest-based manufacturing provides $3.5 billion in value-added and $7.9 billion in value of shipments to Indiana's economy each year. It ranks first nationwide in the production of wood office furniture, wood kitchen cabinets, and hardwood veneer, along with several other products. As small family-owned businesses, wood products companies average less then 50 employees and play an important role in rural communities" (Indiana Department of Natural Resources 2007).

Because of the social and economic importance of Indiana woodlands, an in-depth, contemporary assessment of Indiana residents' forest values and opinions about forest management was needed to explore such topics as residents' opinions about issues of sustainability, acceptable forest practices, and trust in state and federal government to manage woodlands. The Division of Forestry (DoF) in the Indiana Department of Natural Resources (DNR) contracted with Purdue University, Department of Forestry and Natural Resources (DFNR), to measure Indiana residents' perceptions of woodland management, specifically,

- opinions about active forest management on public and private lands,
- perceptions of what constitutes "sustainable" forest management,
- sentiments toward forest values in woodland management (e.g., aesthetics versus economics),
- awareness and perceptions of "certified sustainable forests" (e.g., are people more accepting of forest management if it is certified sustainable),
- trust in government regarding forest management,
- perceptions of selected forest management practices, and
- environmental values orientations.

Selected socioeconomic characteristics of respondents were to be assessed as well.

METHODS

The DFNR contracted with DJ Case & Associates (DJ Case) to help with survey development and analysis. The DNR, DFNR, and DJ Case survey team agreed that, because the population of interest was Indiana's general public, a telephone survey was the preferred methodology. The survey team developed a "Table of Specifications and Survey Item Pool" that listed survey objectives and specific question wording (or "items") that might illuminate each objective. Survey drafts were shared among the team in an iterative process. Seventeen different versions were considered through the refinement process, resulting in a survey instrument ready for pre-testing.

Though many question and wording nuances were considered in the survey development process, one deserving special mention was agreement within the survey team that the term "woodland" should accompany the word "forest," and in many cases,

supplant it. "Woodland" has proven its merit as a descriptive term in the U.S. Department of Agriculture, Forest Service's "National Woodland Owner Survey" (U.S. Forest Service 2011). The survey team agreed that "woodland" has intuitive meaning and appeal as a term to supplement "forest" and also serves as a descriptive stand-alone term.

Delve, a market research firm with call centers and focus group facilities in 10 major U.S. cities, administered the telephone pre-test by random-digit-dialed (RDD) selection to 25 English-speaking Indiana adults, 18 years and older. DJ Case listened to a sub-sample of these interviews and consulted with the survey team on survey modifications apparently needed, as well as on survey strengths (such as the pre-test respondents' clear understanding of the term "woodland"). Then the 25-question survey was finalized and interviewing began.

Completed telephone interviews (RDD) totaled 1,402 interviews (687 males and 715 females), with an average interview length of 15 minutes. There were 3,346 "refusals to participate," yielding a cooperation rate of 30 percent.

Every survey is subject to non-response bias, so there was the question of how well this sample represented the adult population of Indiana, 18 years and older. Respondents were compared to known Indiana population distributions for gender, residence by metropolitan/non-metro area, age, and education. Survey respondents tended to represent Indiana statewide gender and metro/non-metro distributions. However, younger adults were under-represented and older adults were over-represented; lower education levels were under-represented and higher education levels were over-represented. A weight was applied that combined the actual Indiana distributions for age and educational attainment to normalize the dataset for these variables.

The following narrative summarizes a detailed survey report prepared for the Indiana DoF [2] (Amberg 2010). Post-normalization results are presented for statewide results. Odds ratios are reported in those instances where the likelihood ratio chi-square statistic was significant at or beyond the 0.05 level. The odds ratio is the ratio of the odds of a condition occurring in one group to the odds of it occurring in another group.

RESULTS

When asked "How interested are you in Indiana's fish, forests, wildlife, and the out-of-doors?", 50 percent of Indiana adults said they are "very interested," 40 percent, "somewhat interested," and 10 percent, "not interested". Respondents were asked whether they or any members of their households participated in selected outdoor-related activities (Fig. 1).

Household outdoor activities mentioned by majorities of respondents were "visiting an Indiana state forest" (79 percent), "fishing or hunting in Indiana woodlands" (71 percent), and "gathering mushrooms, nuts, or berries from woodlands" (51 percent). Forty-three percent said that they or household members had "cut/collected trees for firewood," and about 1 in 10 respondents said s/he or a household member actually had "worked in the wood products industry."

Of special interest was the large number of respondents (nearly 80 percent) saying they had visited an Indiana State Forest. They were asked, "Could you tell me the name of the State Forest you visited?" A majority (60 percent) offered names of properties

[2] Amberg, S.M. 2010. Indiana residents' perceptions of woodland management: "Indiana woodland monitor 2009 (IWM-09)". A final report to Jack Seifert and the Indiana Division of Forestry. In cooperation with D.J. Case and Associates, Mishawaka, IN. 255 p.

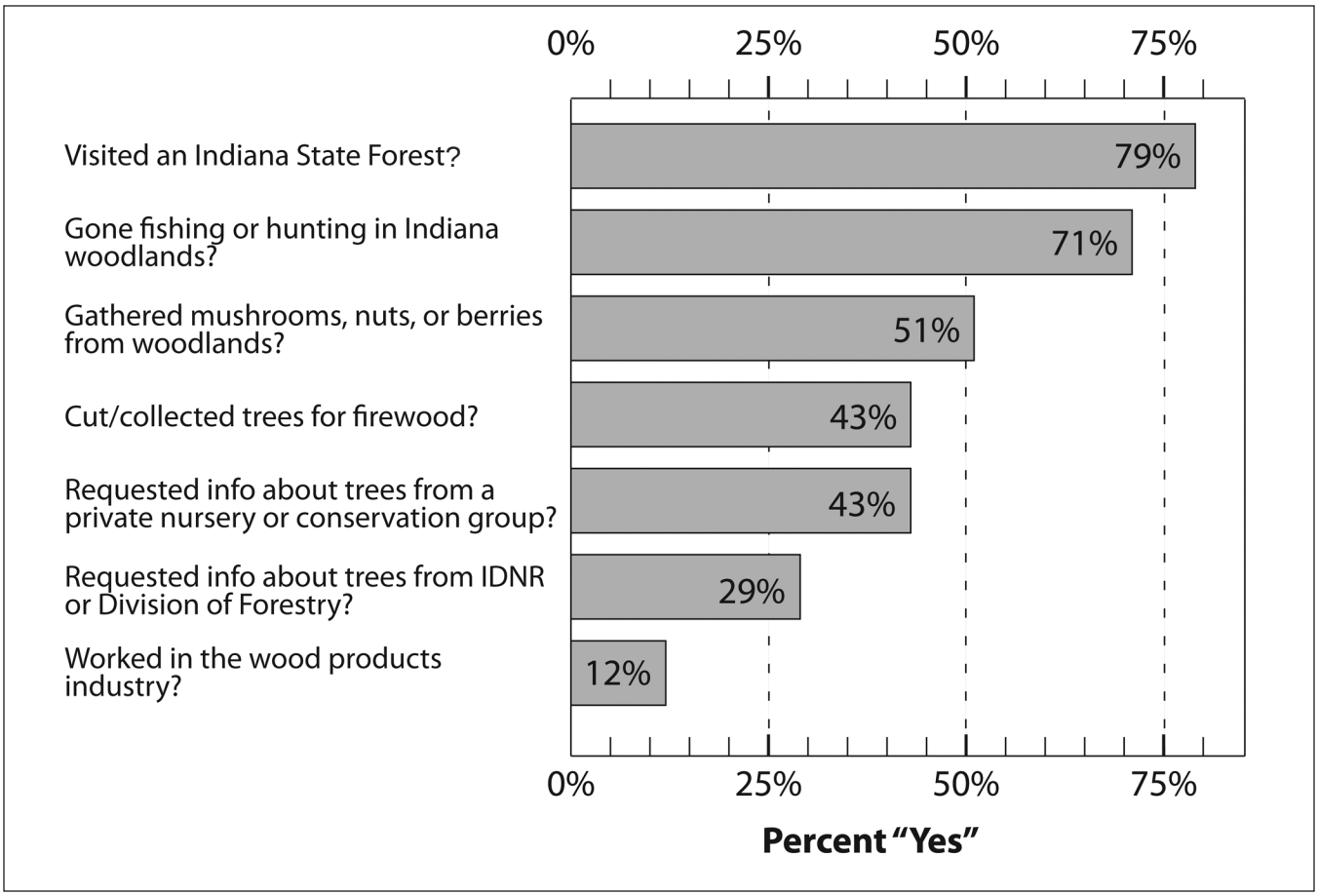


Figure 1.—Percentage of surveyed Indiana residents who answered yes to the question, "Have you or any member of your household…" (n~1,360).

not managed by the DoF, though many were administered by the DNR, such as Indiana State Parks. Fourteen percent correctly offered the name of at least one DoF property. Nine percent named the "Hoosier" National Forest, managed by the U.S. Forest Service, and 17 percent could not remember the name of the area visited.

Respondents were asked how familiar they were with the Indiana DoF. A majority of respondents (58 percent) said they were "not familiar" with the DoF, while 37 percent were "somewhat familiar" and only 5 percent were "very familiar." When asked, "What first comes to mind when you think of the Indiana Division of Forestry" (without any prompt), 80 percent of respondents offered an answer. The most frequent response was "forest management," yet only 29 percent of respondents offered this seemingly obvious answer.

Respondents also were asked to rate the job that the Indiana DoF is doing in providing services for Indiana. In total, 57 percent responded either "excellent" (11 percent) or "good" (46 percent), with 23 percent rating performance as "fair," 4 percent "poor," and 15 percent "don't know." A large majority (74 percent) of the small group (5 percent) that said they were "very familiar" with the DoF rated the Division's performance as "excellent" or "good."

Of interest is whether Indiana residents know who owns the state's woodlands. A plurality (47 percent)

thought most of Indiana's woodlands are held in about equal ownership by private/corporate owners and government; 26 percent responded "government," 18 percent, "private," and 9 percent, "don't know." In fact, roughly 85 percent of Indiana's forest lands are privately owned, with the balance in federal (12 percent) and state (3 percent) ownership (Sierra Business Council 2011). Similarly interesting is whether Hoosiers think that the state owns enough forested land. When informed that "Indiana's State Forest System makes up about 3 percent of Indiana's woodlands," a majority (55 percent) said they thought that the amount was "not enough," with 39 percent indicating the amount was "about right."

Respondents were asked how concerned they are about the long-term health and productivity of Indiana's woodlands. Nearly half (48 percent) indicated they were "very concerned," almost as many (45 percent) said "somewhat concerned," and only 7 percent said "not concerned."

The degree of Hoosiers' concern for the long-term health of Indiana's woodlands varied by selected background variables. The odds of respondents' saying they were "very concerned" (versus "somewhat/not concerned") were:

- larger by a factor of 1.7 for those owning ≥1 wooded acre (0.4 ha) than those who do not,
- larger by a factor of 1.5 for that group who purchased lumber or wood products for home improvement in the last 5 years than those who did not,
- larger by a factor of 1.3 for residents of southern Indiana than residents of northern Indiana,
- larger by a factor of 1.3 for Hoosiers older than 44 years versus those 44 years and younger, and
- larger by a factor of 1.3 for Indiana residents who say they are liberal-leaning than those who are conservative/conservative-leaning.

Survey participants were presented a list of ways that woodlands might be managed, and asked to indicate their approval or disapproval. Near-unanimity (95 percent) existed for removal of some trees to protect Indiana woodlands from disease and wildfire. Majority support also was given to:

- harvesting Indiana trees if overseen by professional foresters (85 percent),
- harvesting trees to improve wildlife habitat (82 percent),
- advising private landowners on how many and what kinds of trees they might harvest and sell (70 percent), and
- harvesting trees to make lumber or other wood products we use (61 percent).

Respondents seemed ambivalent about the idea of "importing wood products from other states and countries instead of harvesting Indiana trees," with 48 percent approval. There was very low approval (18 percent) of "cutting Indiana trees to make room for new homes."

Arguably, one of the most important questions in the survey, at least from the standpoint of contemporary forest management and wood products in Indiana, was the public's sentiment toward "harvesting Indiana trees to make lumber or other wood products that we use" (61 percent approval). The odds of respondents' saying they "approved" of harvesting Indiana trees to make lumber or other wood products that we use (versus "disapproved") were:

- larger by a factor of 3.1 for those who purchased lumber or wood products for home improvement in the last 5 years than those who did not,
- larger by a factor of 1.9 for Indiana households that cut or collected firewood than those that did not,
- larger by a factor of 1.7 for households with a member who worked in the wood products industry,

- larger by a factor of 1.7 for men than women,
- larger by a factor of 1.5 for Hoosiers residing outside of metropolitan counties than those residing in metro counties,
- larger by a factor of 1.4 for Hoosiers describing themselves as "conservative/conservative-leaning" than those describing themselves as "liberal/liberal-leaning,"
- larger by a factor of 1.4 for respondents describing their residences as "rural" than "urban" residents,
- larger by a factor of 1.4 for Hoosiers owning ≥1 wooded acre (0.4 ha) than those who do not, and
- larger by a factor of 1.3 for residents of southern Indiana than northern Indiana.

Respondents were asked to express agreement or disagreement with each of several statements about the importance of Indiana woodlands and management of these woodlands. Hoosiers agreed that:

- "Indiana woodlands should be managed for a balance of wood products that we use, and other benefits like recreation, wildlife, and good water quality" (88 percent),
- "If I hear an Indiana forest is being managed as a certified green forest, I get the idea it's being wisely managed for future generations" (76 percent), and
- "The amount of forested land in Indiana is shrinking" (72 percent).

Respondents disagreed that:

- "Indiana woodlands are valuable only if they produce jobs and income for people" (74 percent).

As a group, respondents generally were ambivalent about the statements:

- "Indiana woodlands should be left untouched for humans" (49 percent disagreed, 33 percent agreed, 18 percent were neutral/don't know),
- "The ways we manage Indiana woodlands today usually hurt the environment in the long-run" (33 percent disagreed, 36 percent agreed, 31 percent were neutral/don't know), and
- "In Indiana, our future wood needs will be met because we're growing more than enough trees to replace the ones we're harvesting" (42 percent disagreed, 27 percent agreed, 31 percent were neutral/don't know).

These findings also can be portrayed using "mean score analysis" (Fig. 2), which produces a "word anchor" that can be assigned to each statement based on the rounded mean score of responses. In this case, mean score analysis confirmed that respondents:

- agreed that
  - Indiana woodlands should be managed for a balance,
  - "certified green forest" gives the idea of forests for the future,
  - the amount of Indiana forested land is shrinking;
- were neutral that
  - woodland management today usually hurts the environment,
  - Indiana woodlands should be untouched by humans,
  - Indiana is growing enough trees to replace those harvested;
- disagreed that
  - Indiana woodlands are valuable only for jobs and incomes.

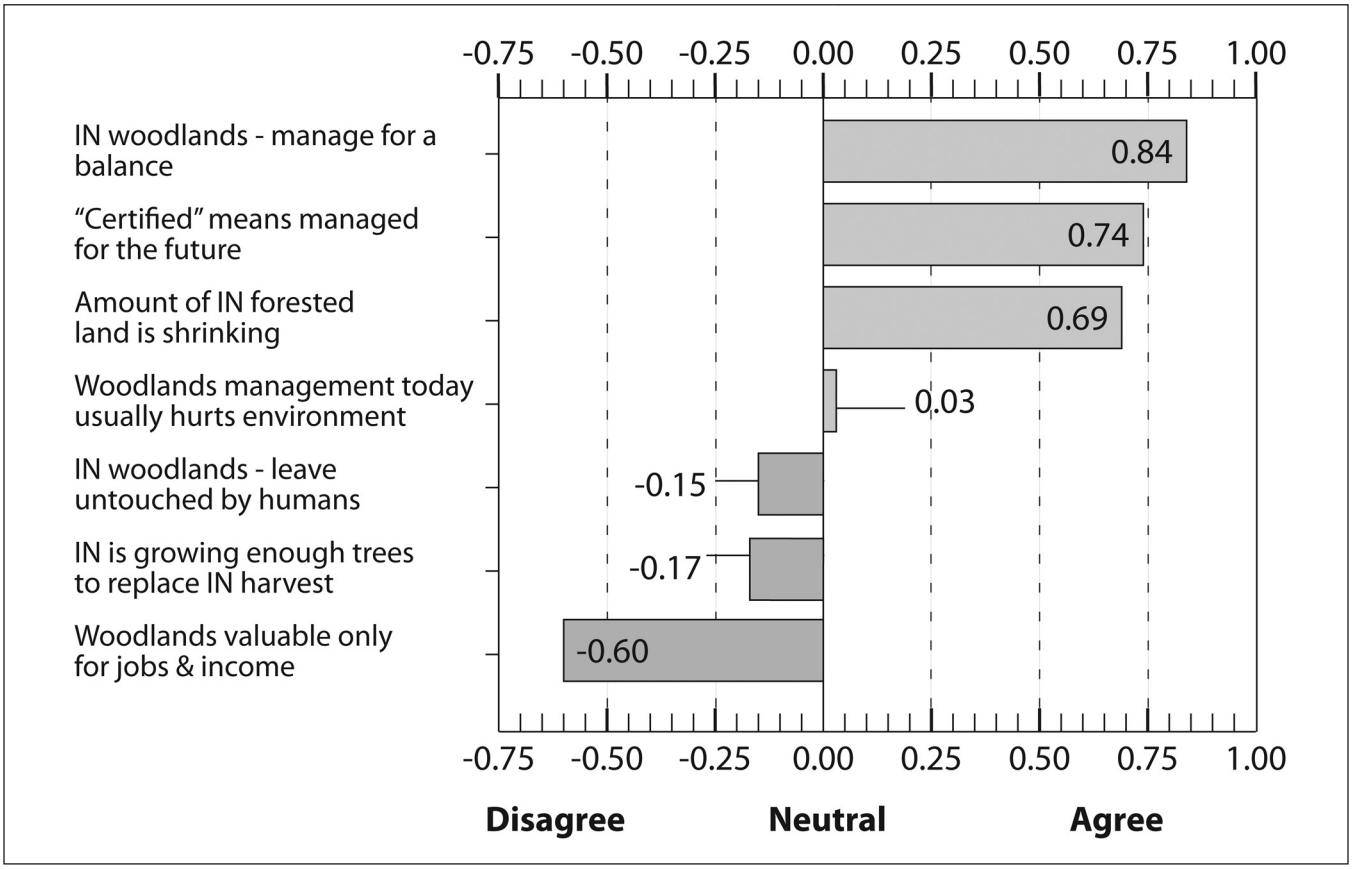


Figure 2.— Surveyed Indiana residents' degree of agreement with seven statements about Indiana woodlands: ". . . please tell me if you agree, disagree, or are neutral," where -1 = "disagree," 0 = "neutral," and +1 = "agree." Agree range = 0.50 to 1.00; Neutral range = -0.49 to +0.49; Disagree range = -0.50 to -1.00 (n~1,320).

The topic of "certified green forests" was pursued at some length with respondents. They were asked, "If you've heard the term 'certified green forests,' in a few words, could you please tell me what the term means?" Though the term "certified green forests" appealed to respondents, most (68 percent) were unable to offer a meaning of the phrase. Of the few able to provide an answer, most frequently mentioned was the idea that such forests are managed for "green" or environmentally friendly products (14 percent), or more simply, that the forests are wisely or well managed (7 percent).

Respondents were asked their opinions of the severity of each of nine possible threats to Indiana woodlands. Mean score analysis revealed that "urban sprawl" and "air and water pollution" were the two highest-rated threats, followed closely by "insects or plant diseases." On average, however, none of these qualified in the minds of respondents as a "high threat" to Indiana woodlands, but rather as "medium threats" (Fig. 3). All nine possible threats fell in the "medium threat" range, on average. The bottom of the list was shared by "unplanned forest fires" and "planned or prescribed fires." "Climate change" was near the bottom of the threat list, indicating that the issue has not registered as a serious matter in the opinion of the Indiana adult population.

Among the most revealing questions in the survey was one asking respondents to identify the person or experience that most influenced their sentiments about woodland management and timber harvest. Among the 1,402 responses provided was "can't answer".

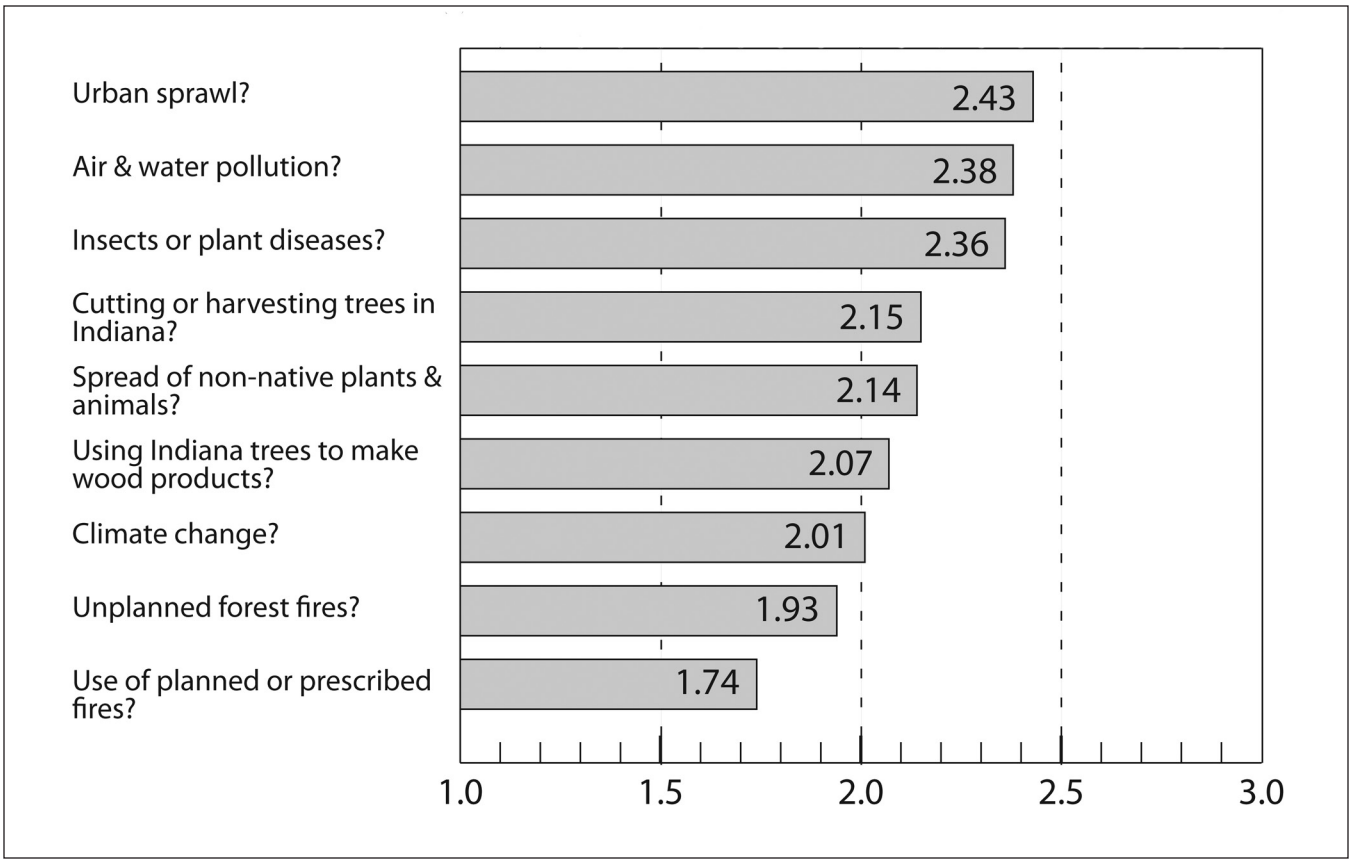


Figure 3.— Surveyed Indiana residents' responses to questions, "Do you think each of the following represents a high, medium, or low threat to today's Indiana woodlands," where 1 = "low threat," 2 = "medium threat," 3 = "high threat," or "don't know". The answer "don't know" was eliminated for purposes of this analysis. High threat range = 2.50 to 3.00, Medium threat range = 1.50 to 2.49, Low threat range = ≤1.49 (n~1,330).

These data presented an analytical challenge because the open-ended statements were diverse in context, and reduction or categorization of the responses might result in loss of depth and richness of answers. For purposes of data reduction and summarization, however, each response was read individually and grouped into one of ten general categories (Table 1).

Thirty-one percent of respondents implied or stated that their woodland values were rooted in their lifestyles (Table 1), essentially, values virtually inseparable from their personal identities and how they think as individuals. These values generally grew from early life experiences and training—family activities, memories at a specific outdoor place, hiking,

Table 1.—Categorization of open-ended responses to: "Briefly, what person or experience would you say most influenced your sentiments about woodland management and timber harvest?"

Influence	Number	Percent
My lifestyle	432	31
"Resource professionals"	238	17
Family member	210	15
News media	152	11
Can't answer	151	11
Travel/witnessing good and bad	81	6
Self-taught	66	5
Conservation-environ. groups	37	3
Friend/neighbor	32	2
Classified forests	3	trace
Total	1,402	100

camping, Scouting, fishing, collecting and burning fuel wood—though some respondents referenced more recent epiphanies about the importance of nature. Seventeen percent were influenced by a resource professional; that is, they were employed in natural resource management, influenced by a relative or acquaintance in park or resource disciplines, schooled in natural resources, mentored in resource studies by teachers or professors, or knew farmers or loggers/ mill operators who made their living on the land. Additionally, they reported being exposed to formal professional outreach, such as university extension, DNR publications, soil and water district contacts, political contacts, National Guard training, or Smokey Bear.

Other pivotal people or experiences included family members (15 percent), news media (11 percent), exposure through travel to good and bad examples of forest management (6 percent), conservation and environmental groups (3 percent), and friends and neighbors (2 percent); a few respondents mentioned their participation in the DNR's Indiana Classified Forest and Wildlands Program (Table 1).

Objectives of the Indiana Woodland Monitor 2009 are italicized below and were to assess Hoosiers':

- *Opinions on active forest management on public and private lands, and trust in government regarding forest management.* Indiana residents are generally unknowledgeable about who owns the state's woodlands, with almost half thinking that Indiana forests are in equal public and private ownership. However, roughly half say they are "very concerned" (and most others, "somewhat concerned") about the long-term health and productivity of Indiana's woodlands. A large majority would support efforts to advise private landowners on how many and what kinds of trees they might harvest and sell. A majority think that the 3 percent of Indiana's woodlands in State Forest management is not enough, even though most say they are "not familiar" with the Indiana DoF, and when asked what first comes to mind when they think of the DoF, few indicate that the agency has something to do with looking after trees.
- *Perceptions of what constitutes "sustainable" forest management, and importance of different forest values in forest management (e.g., aesthetics versus economics).* "Sustainable" forest management in the minds of Hoosiers appears to mean balancing forest benefits. A large majority supports harvesting Indiana trees for woodland management if overseen by professional foresters, but fully one-third would prefer that the state's woodlands be untouched by human hands. According to our survey, Hoosiers are essentially imploring, "Do what you need to do, including harvesting Indiana trees for lumber or other wood products that we use, but do it professionally with an eye toward forests for the future."

  There is strong indication that the Indiana citizenry's acceptance of the full gamut of forest values reflects the sentiment of Indiana's forest landowners. In a study of family forest owners in north-central Indiana, Ross-Davis (2006) identified three distinct landowner types: (1) "Forest Managers," who attributed importance to diverse values with regard to owning their forests and were most likely to have harvested trees from their land in the past, (2) "New Forest Owners," who owned their properties for the least amount of time and attributed importance to diverse ownership motivations with the exception of producing timber, and (3) "Passive Forest Owners," who owned the smallest forested areas and attributed importance only to enjoying the scenery of their woodlands.
- *Awareness and perceptions of "certified sustainable forests" (e.g., Are people more accepting of forest management if it is certified sustainable?).* The phrase "certified green forest" gives Hoosiers the idea that the forest

is being managed for future generations, so whatever "certified green forest" is, they are reassured. However, other than this vague impression, Hoosiers generally do not have a clear understanding of the phrase. Moreover, Indiana residents indicate that they think the amount of forested land in the state is shrinking, and many perceive that not enough trees are being grown to replace trees harvested.

- *Types of forest management which are socially acceptable.* A majority of Indiana residents support tree harvest to protect Indiana woodlands from spread of disease and wildfire, to improve places for wildlife to live, and to produce lumber and wood products. The citizenry is divided on the idea of importing wood products from other states and countries instead of harvesting Indiana trees, but Hoosiers roundly reject the thought of cutting Indiana trees to make room for new homes, describing the greatest threat to Indiana woodlands as "urban sprawl." Near the bottom of Hoosiers' list of woodland threats is "climate change."
- *Environmental values orientations.* As a citizenry, Hoosiers acknowledge the utilitarian values of the state's woodlands, while expecting to see the all-important balance between wood products and other benefits, such as recreation, wildlife, and good water quality. Without doubt, the Indiana public considers what the DoF does to be important and relevant, but the citizenry lacks a well-defined idea of who does it.

ACKNOWLEDGMENTS

The authors extend thanks to the Indiana adults who made this project possible by taking time to participate in the telephone survey, and to the Indiana Department of Natural Resources for initiating and funding the project.

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INTRODUCTION TO THE HARDWOOD ECOSYSTEM EXPERIMENT

THE HARDWOOD ECOSYSTEM EXPERIMENT: GOALS, DESIGN, AND IMPLEMENTATION

Rebecca A. Kalb and Cortney J. Mycroft[1]

Abstract.—The Hardwood Ecosystem Experiment (HEE) is a long-term, landscape-level field experiment initiated in 2006 by the Indiana Department of Natural Resources-Division of Forestry. The HEE is a multi-disciplinary, collaborative research project involving researchers from Purdue University, Indiana State University, Ball State University, Indiana University of Pennsylvania, and Drake University. It is designed to study the ecological and social impacts of forest management through a replicated series of study areas established across 730 hectares in two Indiana State Forests. Each of the nine study areas is approximately 80 ha in size. Over the next 100 years, the impacts of even-aged management, uneven-aged management, and no timber harvesting (control) will be assessed. Across these areas, long-term responses to forest management of various plant and animal taxa, including threatened and endangered species, will be monitored. Pre-treatment data collection began in 2006. Treatments were applied beginning in the summer of 2008.

INTRODUCTION

Disturbance patterns of fire, grazing, farming, and unregulated timber harvesting throughout the Central Hardwood Region has led to the ecological, economic, and social interdependencies found in the region's forests today (Carman, this publication). Both historically and currently, oak (*Quercus*) and hickory (*Carya*) are dominant throughout much of Indiana's forestland. As disturbance regimes have changed, however, oaks have become outnumbered in the seedling and sapling layers by more shade-tolerant species, principally maples (*Acer*) and American beech (*Fagus grandifolia*). This shift from oak-hickory to beech-maple poses a challenge for both the wildlife and people that depend on oak-dominated mid-successional species for their livelihoods (McShea et al. 2007, Shifley and Woodall 2007).

With the loss of American chestnut (*Castanea dentata*) in the early 1900s, oak acorns have become an even more important food source for many wildlife species throughout the Central Hardwood Region (Dalgleish and Swihart 2012). Acorns are readily available and rich in energy. With such a valuable food source, oak-dominated ecosystems support diverse wildlife communities (McShea et al. 2007, Rodewald and Abrams 2002). Beyond changes in overstory structure and composition, a reduction in mast-producing species may alter the representation of birds, small mammals, and other species (McShea et al. 2007).

Indiana's hardwoods are essential for sustaining forest-dependent wildlife, and they also provide an economic livelihood for citizens of the state. Indiana's forest products sector is the sixth largest industry in the state and directly employs more than 35,000 people

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(Indiana Department of Natural Resources [IDNR] - Division of Forestry [DoF] 2011). Indiana has a rich history of high-quality hardwood production (Carman, this publication). There may be economic implications if the forests are allowed to shift to their climax state (Hoover, this publication).

Many people enjoy Indiana forests for recreation and aesthetic value. Bird watching is a particularly popular activity, with 1.4 million people observing birds around the state in 2001. More than 400,000 people visited Indiana's public woodlands in 2001 (U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, Bureau of the Census 2003). Perception and acceptance of forest management differ considerably among individuals and can affect their enjoyment of these areas for recreation (Rogers et al., this publication; Witter et al., this publication).

Since the creation of Indiana's State Forests, their management has been primarily single-tree and small group selection (Carman, this publication; Parker and Ruffner 2004). The small canopy openings that result have been shown by previous researchers to favor the regeneration of shade-tolerant species (Johnson et al. 2002). Although even-aged silvicultural systems have been recommended for oak forests of the Central Hardwood Region (Johnson et al. 2002, McShea and Healy 2002), the public is often vehemently against the initial appearance these systems, especially clearcuts, leave on the landscape (Bliss 2000; Rogers et al., this publication). The social implications of creating larger openings to promote oak regeneration need to be weighed against the need for a diverse matrix of early- and late-successional stages in Indiana's remaining forestland.

The interface of ecological change and societal acceptance of forest management will present a growing challenge for land managers in the coming decades. Science-based management will be essential if the current oak-dominated landscape and its associated diversity are to be maintained. Moreover, educating the public about research findings and integrating the public's views into management objectives will enhance the likelihood of success (Hanley 1994). The questions surrounding oak regeneration are complex. The best hope of answering them is in the use of multi-disciplinary studies designed with long-term objectives. One such study currently underway in Indiana is the Hardwood Ecosystem Experiment (HEE). Although the HEE is in its initial stages, it is designed as a 100-year project. The HEE includes research on improving oak regeneration and seeks to create a better understanding of the relationships between the forest, its wildlife, and humans. The ultimate intent of the HEE is to provide information that will improve management prescriptions and thereby enhance sustainability of forest resources and improve the quality of life for Indiana residents.

THE HARDWOOD ECOSYSTEM EXPERIMENT

The HEE is a multi-disciplinary project initiated by the IDNR-DoF in collaboration with scientists from Purdue University, Indiana State University, Ball State University, Indiana University of Pennsylvania, and Drake University. The HEE is designed as a long-term, landscape-level field experiment to study forest management and its impacts. In collaboration with DoF, a replicated series of study areas was created at Morgan-Monroe and Yellowwood State Forests (MMYSF) in southern Indiana, on which alternative approaches to forest management will be implemented and monitored for 100 years. This framework of study sites will be used to quantify the long-term response of selected flora and fauna in both the treated areas and the surrounding forest. Assessing the attitudes of landowners and the general public toward active forest management is an equally important component.

Through the HEE, we seek to understand how harvest-induced disturbance affects individuals, populations, and communities of plants and animals. Some treatments are aimed at improving oak regeneration, whereas others likely will encourage conversion to beech-maple forests. Researchers will explore how different species use the landscape in response to the harvests. Research results will lead to a comprehensive guide for forest management with clear outcomes described for alternative forest management techniques.

Additionally, social scientists are studying the attitudes and values held by various stakeholders toward forest management. This research is aimed at understanding stakeholder attitudes toward forest management practices and policies, engaging stakeholders and decisionmakers in dialog, and designing outreach programs that address the attitudes and behavior of landowners, residents, and decisionmakers living adjacent to or near MMYSF. Stakeholders include landowners and non-landowners as well as recreational user groups.

HEE field work began in May 2006 with research efforts focused on collecting baseline data to determine the current state of forest resources in MMYSF. Silvicultural treatments were applied between 15 July 2008 and 6 January 2009. Baseline data collection was ongoing until the harvest period (Table 1); post-harvest data collection focused on the response of target species to the various harvest treatments.

STUDY OBJECTIVES

The goal of the HEE is to understand the ecological and social impacts of even-aged (EA), uneven-aged (UEA), and no harvest (NH) management on public and private lands in Indiana and the Central Hardwood Region, with the following primary research objectives:

1. Develop EA and UEA silvicultural systems that maintain oak-dominated forest communities and landscapes.
2. Determine the impacts of these systems on populations of herbaceous, avian, and terrestrial amphibian species groups.
3. Determine the social and economic ramifications of these systems for both the local and regional communities.
4. Provide demonstration sites and develop novel educational materials and techniques to engage the public concerning forest management.

The complete HEE objective tree is found in Appendix 1.

Table 1.—Complete listing of all active research projects during the pre-treatment period of the HEE. Also provided are the start dates, years of data collection, and field season for each.

Pre-treatment research projects:	Start date	Number of inventories	Primary field season
Breeding birds (Malloy and Dunning, this publ.)	2006	3	May-June
Oak mast production and removal (Kellner et al., this publ.)	2006	3	August-December
Wood-boring and longhorned beetles (Holland et al., this publ.)	2006	3	May-August
Bats: mist netting (Sheets, Whitaker, et al., this publ.)	2006	3	May-August
Bats: acoustic sampling (Sheets, Duchamp, et al., this publ.)	2007	2	May-August
Terrestrial salamanders (MacNeil and Williams, this publ.)	2007	2	February-May and August-November
Small mammals (Urban and Swihart, this publ.)	2007	2	June-August
Cerulean warblers (Islam et al. 2012)	2007	2	May-August
Timber rattlesnakes (MacGowan and Walker 2012)	2007	2	April-October
Eastern box turtles (Currylow et al., this publ.)	2007	2	May-October
Lepidoptera (Summerville et al., this publ.)	2007	1	May-August
Overstory/sapling/snag inventory (Saunders and Arseneault, this publ.)	2008	1	Year-round

STUDY AREA

MMYSF is located within the unglaciated Brown County Hills Section of Indiana's Highland Rim Natural Region (Homoya et al. 1985). The area was heavily forested before settlement by Europeans. Much of the land was burned, cleared, heavily grazed, and farmed by European settlers during the 1800s (Parker and Ruffner 2004). This degraded land was acquired by the State in the early- to mid-1900s and allowed to return to forest (Carman, this publication). The area is characterized by dry ridges with often steep ravines underlain by siltstone, shale, and sandstone bedrock. There are few natural water bodies other than steep ephemeral streams which drain to medium- to low-gradient creeks (Homoya et al. 1985). The silt-loam soils of the Wellston-Berks-Gillpin complex are found throughout this area (Jenkins, this publication).

The overstory is predominantly oak-hickory with black oak (*Q. velutina*), white oak (*Q. alba*), and shagbark hickory (*C. ovata*) dominating ridges and south-facing slopes. Almost pure stands of chestnut oak (*Q. prinus*) can be found on the driest, highest ridge tops. Northern red oak (*Q. rubra*), tulip poplar (*Liriodendron tulipifera*), American beech, sugar maple (*A. saccharum*), and hickories occur on the mesic north-facing slopes. The species composition of the understory is largely beech-maple; tulip poplar and sassafras (*Sassafras albidum*) are also common. Sedges (*Carex*) are prevalent on dry ridges, whereas ferns and other perennials occur on the mesic slopes and lowlands. One noteworthy species is the state-threatened yellowwood tree (*Cladrastis kentukea*), which occurs in Indiana only in Brown County. Yellowwood State Forest also has areas of pine (*Pinus*) plantations from the Civilian Conservation Corps era of the 1930s. Carman (this publication) and Jenkins (this publication) provide further history and description of the area.

Nine HEE management units were established in MMYSF in south-central Indiana (Fig. 1).[2] Morgan-Monroe State Forest contains >9,712 ha in Morgan and Monroe Counties, and was established in 1929. Yellowwood State Forest, created in 1947, encompasses >9,439 ha of Brown County. The HEE management units stretch across the landscape, covering 30 linear kilometers. The total area encompassed by the HEE is 3,603 ha, which is nearly 20 percent of the entire MMYSF complex.

Each management unit consists of a central research core area surrounded by a buffer area (Fig. 1) with the research core serving as the experimental unit. The management units were created using existing DoF management tracts with boundaries typically following ridges and valleys. Each research core comprises two to three DoF management tracts, totaling 78-110 ha (Table 2). Tracts immediately adjacent to these cores were designated as buffers to isolate research cores from state forest management activities not associated with the HEE. The sizes of the management units, including research cores and buffer areas (219-393 ha), range from 303 to 483 ha (Table 2). Each of the nine cores was randomly assigned a treatment: EA management using clearcut and three-stage shelterwood methods, UEA management using single-tree selection with patch cutting, and NH (experimental control), with three replicates of each. Most of the HEE inventory and monitoring systems are contained within the research cores.

The core areas of the nine HEE management units are to be managed on a 100-year rotation, regenerating 20 percent of each core area every 20 years. The management regime will remain constant, but the size and number of openings will be informed by the previous harvest results and current forest management practices within an adaptive framework (Walters and Holling 1990).

[2] All spatial information is recorded and presented in North American Datum 1983, Universal Transverse Mercator Zone 16N.

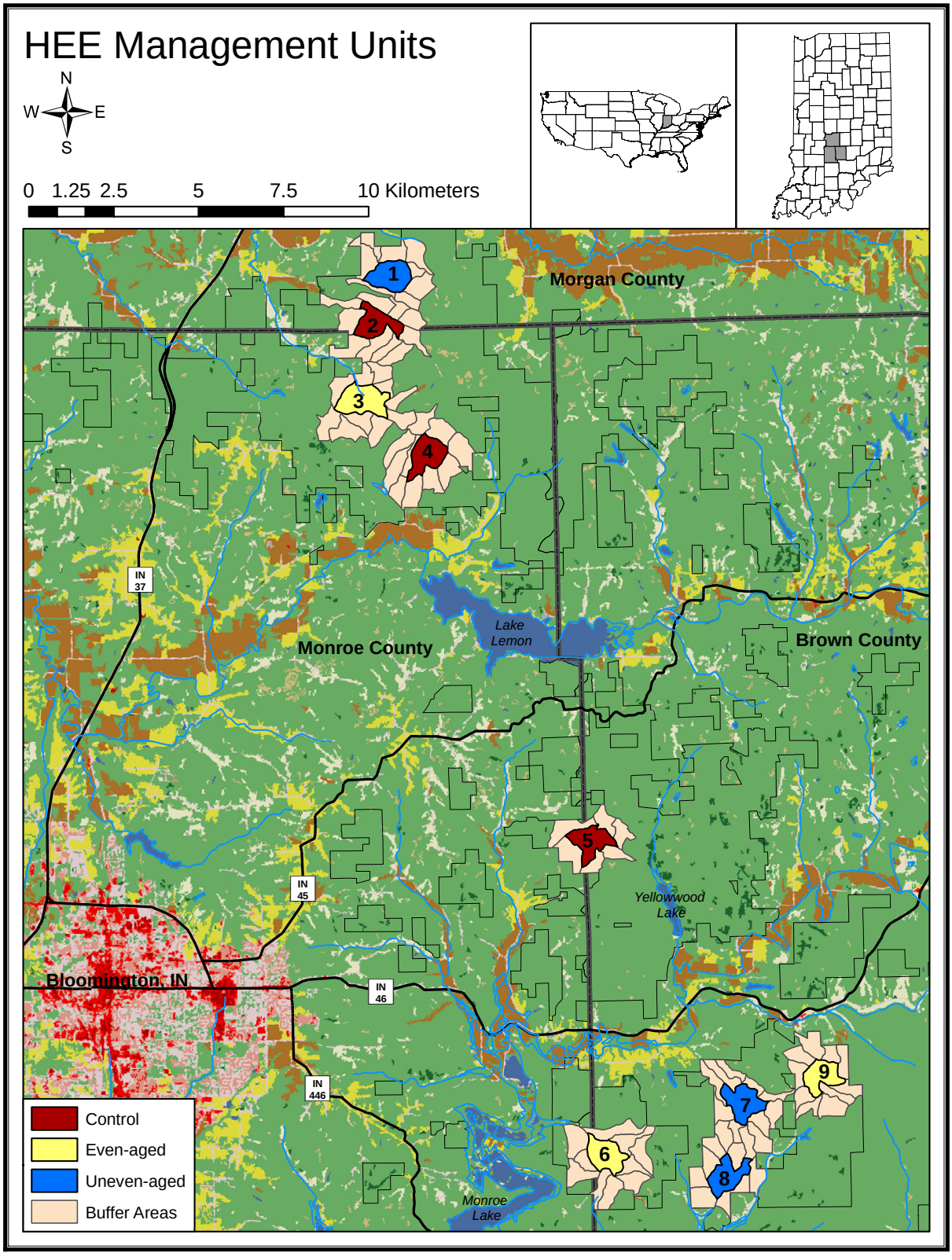


Figure 1.—Map of the entire HEE study area within Indiana. Management units are identified by management type and are shown with surrounding buffer areas. Straight lines are outlines of Morgan-Monroe and Yellowwood State Forest property boundaries. Green areas are deciduous forest as classified by the 2006 National Land Cover Dataset (see Multi-Resolution Land Characteristics Consortium 2011 for the complete land cover classification legend).

Table 2.—Area (in hectares) for each HEE Management Unit, including the research cores and buffer areas (see Fig. 1). All areas were calculated in ArcGIS version 9.3.1.

Unit #	Management type	Research core (ha)	Buffer area (ha)	Management unit (ha)
1	uneven-aged	91.46	253.90	345.37
2	control	83.52	350.83	434.34
3	even-aged	110.44	332.69	443.13
4	control	89.92	392.71	482.63
5	control	84.74	219.24	303.98
6	even-aged	87.15	383.34	470.48
7	uneven-aged	92.42	269.11	361.54
8	uneven-aged	84.59	296.61	381.20
9	even-aged	78.31	302.28	380.59

MANAGEMENT UNIT AND RESEARCH CORE AREAS

Even-Aged Management

To determine harvest locations in EA research cores, six points were randomly placed on the digital elevation map (DEM) using Hawth's Tools in ArcGIS (ESRI, Redlands, CA). The points were stratified by aspect with three on north or east aspects and three on south or west aspects. Two sites from each aspect were chosen by examining topographic maps for space availability and correct aspect. The topographic maps provided more accurate estimations than the DEM. Slopes with the smallest area were discarded. Other points were discarded based on composition or recent disturbance events using knowledge and data from MMYSF foresters. Harvest areas of 4.05 ha in size were drawn around these points to maximize the amount of upper and lower slope positions. Ridges and valleys were used as boundaries when possible. Of the four harvest areas in each unit, one northeast aspect and one southwest aspect were randomly assigned to shelterwood treatments. The remaining two harvest areas were assigned to clearcut treatments. The total area designated for harvest in the first year was 16 ha per core (a total of 48 ha for the three EA cores). EA management units are designated as Unit 3 (Fig. 2), Unit 6 (Fig. 3) and Unit 9 (Fig. 4).

Clearcut

All woody stems >30.48 cm diameter at breast height (d.b.h.) were harvested by a contracted logger. The completion of post-harvest cleanup resulted in a silvicultural clearcut with all woody stems removed. All remaining standing oak, hickory, ash (*Fraxinus*), tulip poplar, and black walnut (*Juglans nigra*) between 2.54 cm and 35.56 cm d.b.h. were coppiced to within 15.24 cm from the ground. All other tree species and larger stems were either felled and chemically treated (with glyphosate or glyphosate with picloram for maple) or girdled and chemically treated. All vines were cut with each end chemically treated. All tree of heaven (*Ailanthus altissima*) received a chemical basal bark treatment of triclopyr with imazypyr.

Shelterwood

The shelterwood treatment is a three-stage system (Smith et al. 1997). The 2008/2009 preparatory cut removed midstory and understory layers with a timber stand improvement (TSI) treatment, which removed most non-oak stems from the stand that were ≤25.4 cm d.b.h. A few small overstory culls were removed, neither lowering basal areas below 13.8 m^2/ha nor creating large canopy gaps. In 5-10 years an establishment cut will reduce stocking to a basal area of 13.8–16.1 m^2/ha. Removal will focus on poorly formed canopy and subcanopy trees, leaving an

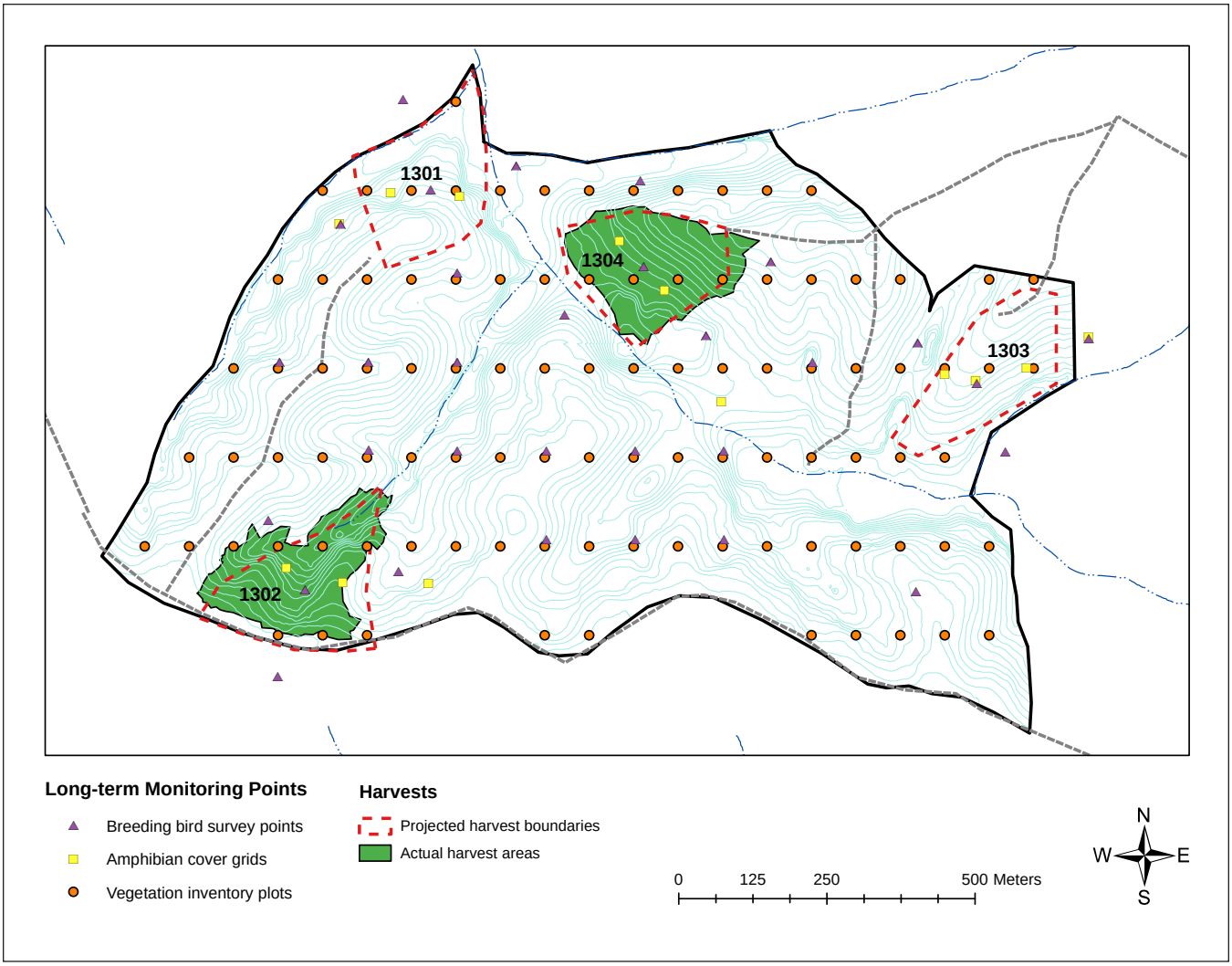


Figure 2.—Map of HEE Research Core 3 (even-aged management) shown with 3-m contour lines, access roads (gray dashed line), and streams (blue dot/dash line). Harvest types include clearcut and shelterwood. Projected harvest areas refer to the harvest areas that were digitally created in ArcGIS. Actual harvest areas were digitized following on-the-ground mapping of the harvest boundaries in 2009, post-harvest. The shelterwoods will be mapped following the mid-story removal stage, so actual harvest areas are not currently available. Long-term monitoring points for breeding birds, amphibians, and vegetation are also shown.

evenly dispersed overstory. Trees favored for retention should have vigorous crowns that will provide ample seed source. Species favored for retention will include oak and hickory. The final overstory removal cut will remove the remaining overstory trees 5-10 years after the establishment cut, or whenever ample oak regeneration reaches breast height (1.35 m), not to exceed 20 years from the initial preparatory cut.

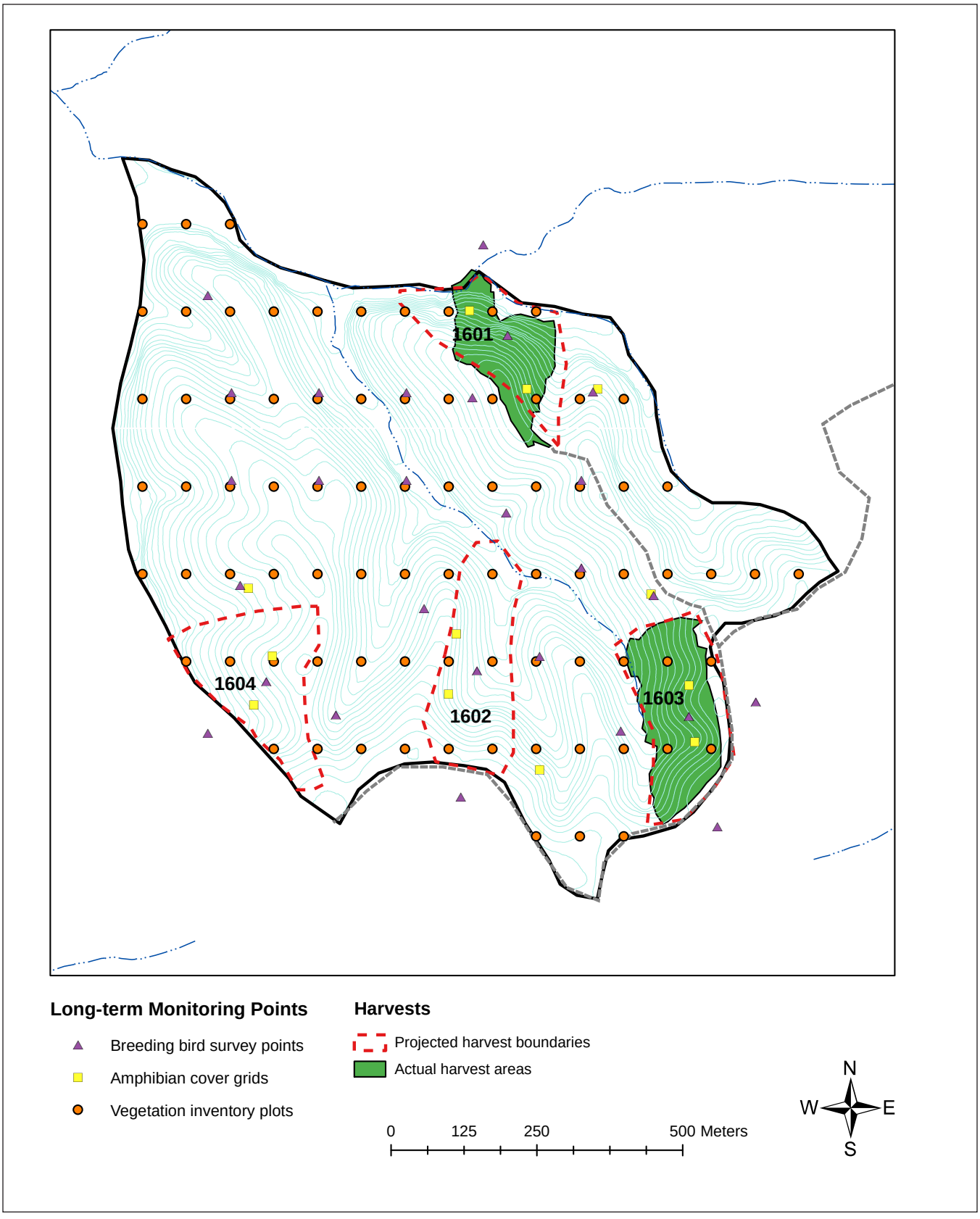


Figure 3.—Map of HEE Research Core 6 (even-aged management). Information is summarized as in Figure 2.

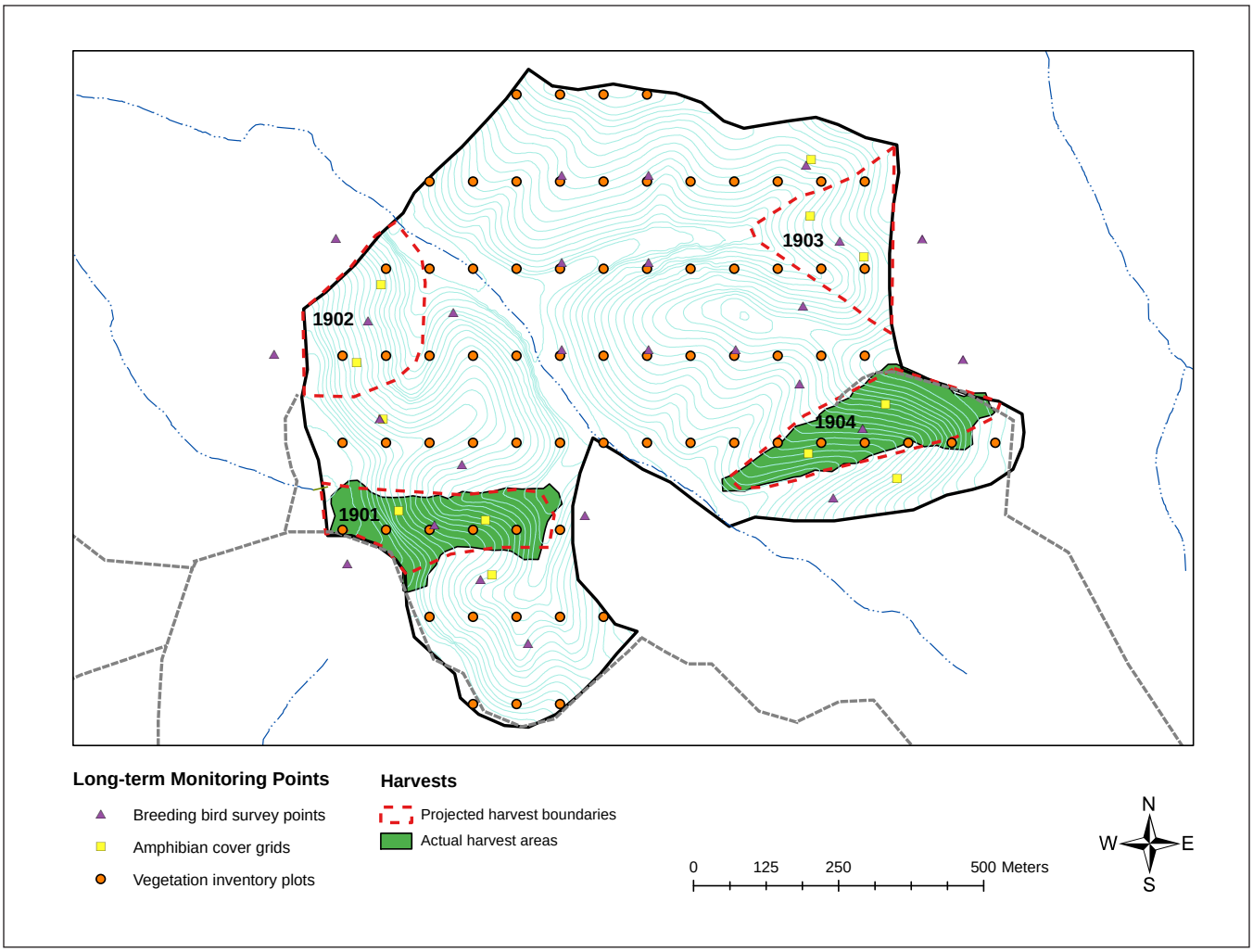


Figure 4.—Map of HEE Research Core 9 (even-aged management). Information is summarized as in Figure 2.

Uneven-Aged Management

The UEA management cores received patch cutting with single-tree selection. To determine patch locations in UEA research cores, eight randomly selected points (four northeast aspects and four southwest aspects) were established following the same method as in the EA cores. A harvest boundary was created around each point in three different patch sizes: two 2.02 ha, two 1.21 ha, and four 0.40 ha. Patch cuts were generally defined by four-sided polygons, centered on each point, with as much of the upper and lower slope positions captured as possible. Patch cuts may not be implemented with every 20-year entry. Single-tree selection was marked with a basal-area goal of 16.1-22.9 m^2/ha. Single-tree selection will recur every 20 years. Single trees will not be marked within 15.2 m of the patches to avoid expanding their size. UEA management units are designated as Unit 1 (Fig. 5), Unit 7 (Fig. 6), and Unit 8 (Fig. 7). This management type most closely resembles the current silvicultural practices used by DoF at MMYSF and throughout Indiana.

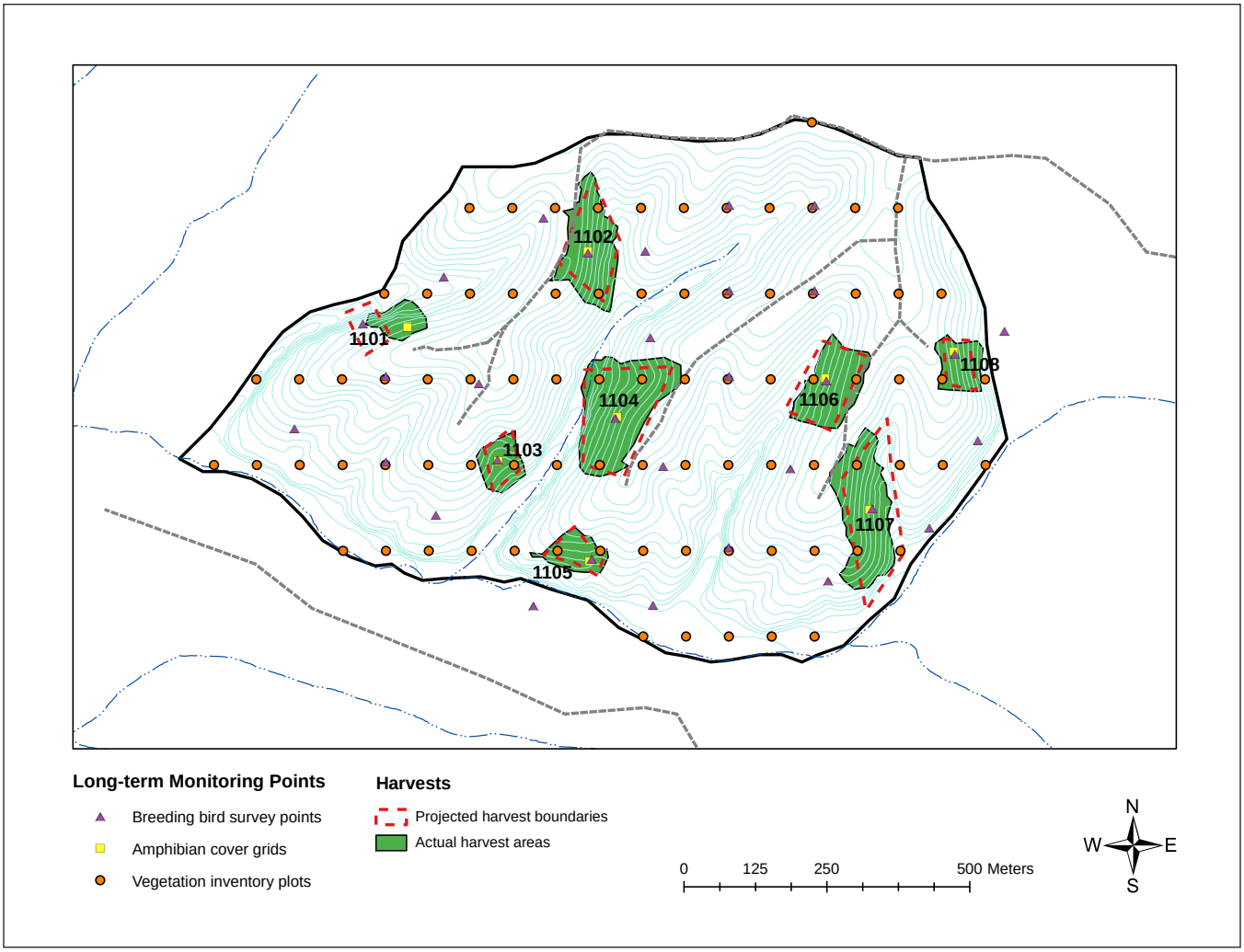


Figure 5.—Map of HEE Research Core 1 (uneven-aged management) shown with 3-m contour lines, access roads (gray dashed line), and streams (blue dot/dash line). All harvest types are patch cuts only. Single-tree selection harvests occur throughout the research core and are not depicted in this map. Projected harvest areas refer to the harvest areas that were digitally created in ArcGIS. Actual harvest areas were digitized following on-the-ground mapping of the harvest boundaries in 2009, post-harvest. Long-term monitoring points for breeding birds, amphibians, and vegetation are also shown.

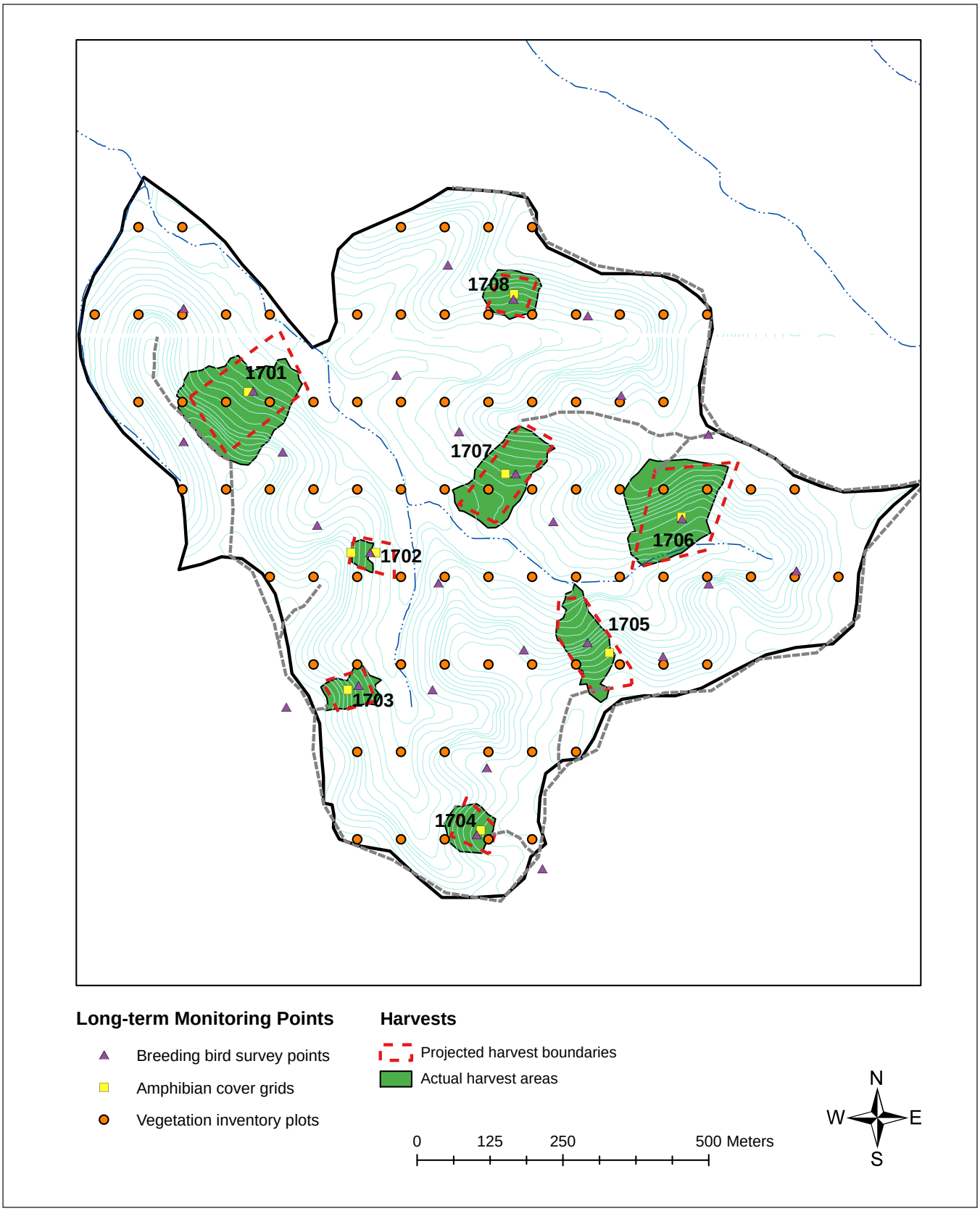


Figure 6.—Map of HEE Research Core 7 (uneven-aged management). Information is summarized as in Figure 5.

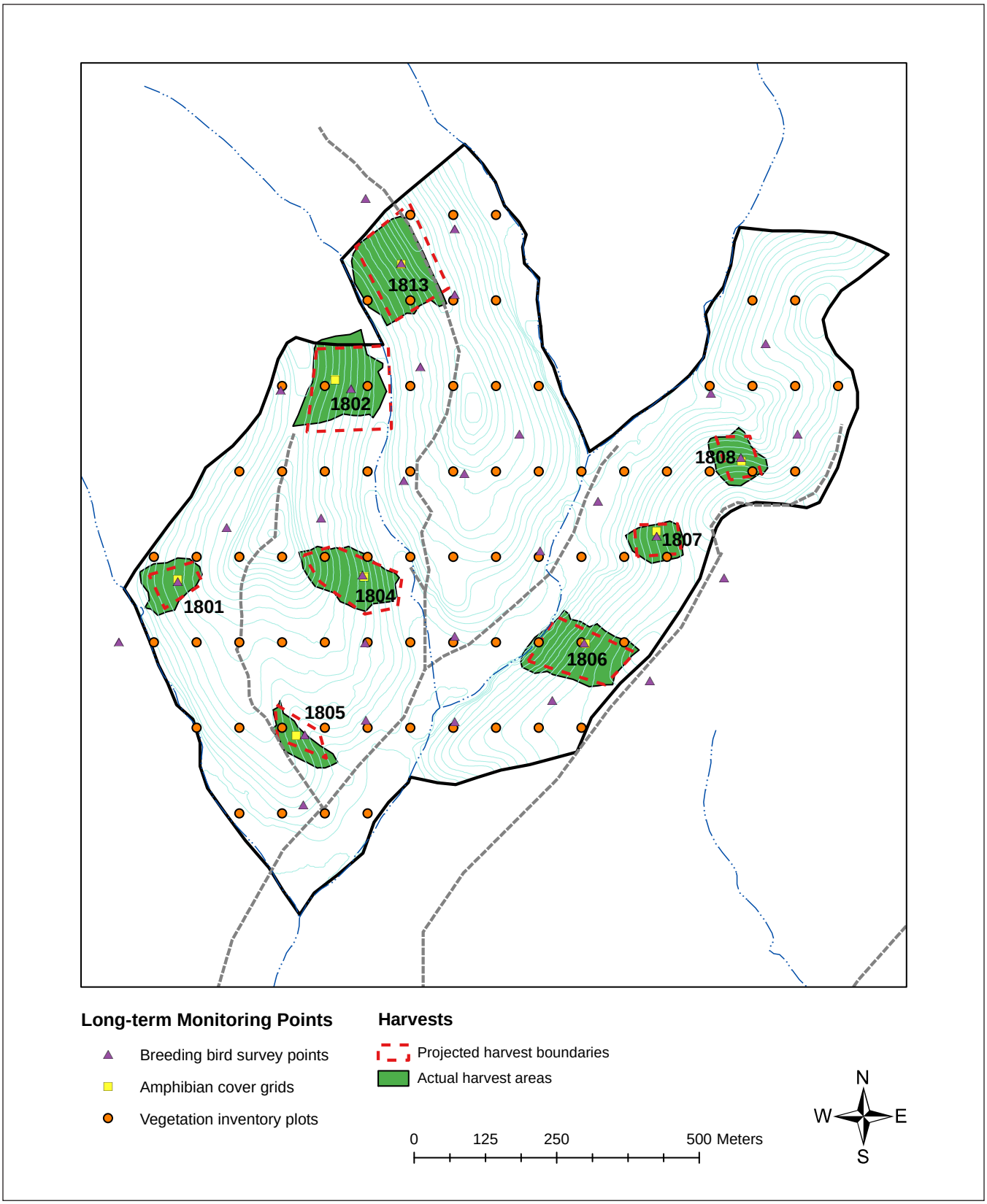


Figure 7.—Map of HEE Research Core 8 (uneven-aged management). Information is summarized as in Figure 5.

No Timber Harvesting

No harvests will be implemented on the control research cores. The three management units assigned to control are Unit 2 (Fig. 8), Unit 4 (Fig. 9), and Unit 5 (Fig. 10), which will receive no active harvesting in the research core for the duration of the study. The control units are used to monitor variation in plant and animal populations and communities in the absence of timber harvesting.



Figure 8.—Map of HEE Research Core 2 (control-no harvest) showing the locations of long-term monitoring points for breeding birds, amphibians, and vegetation with 3-m contour lines, access roads (gray dashed line), and streams (blue dot/dash line).

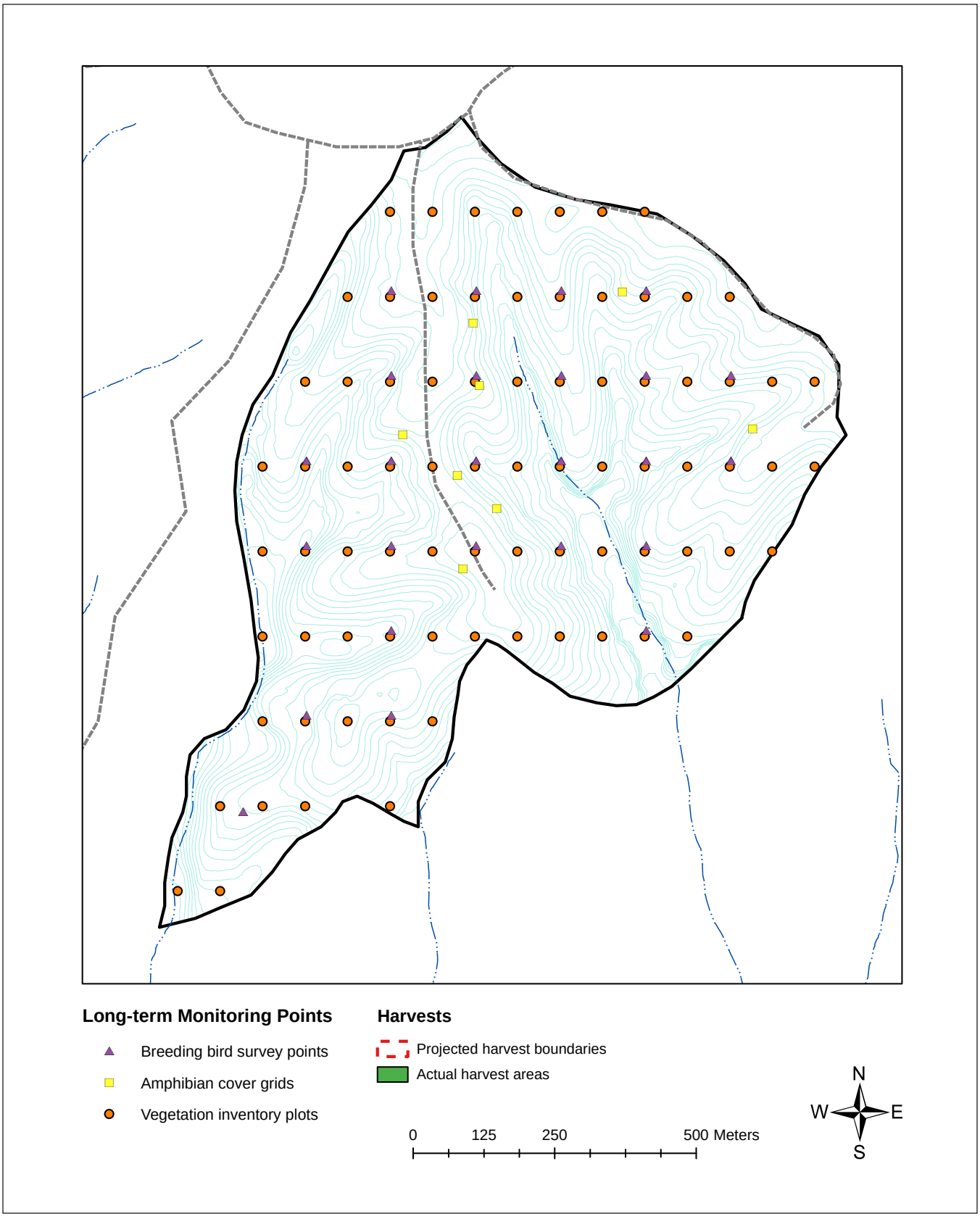


Figure 9.—Map of HEE Research Core 4 (control-no harvest). Information is summarized as in Figure 8.

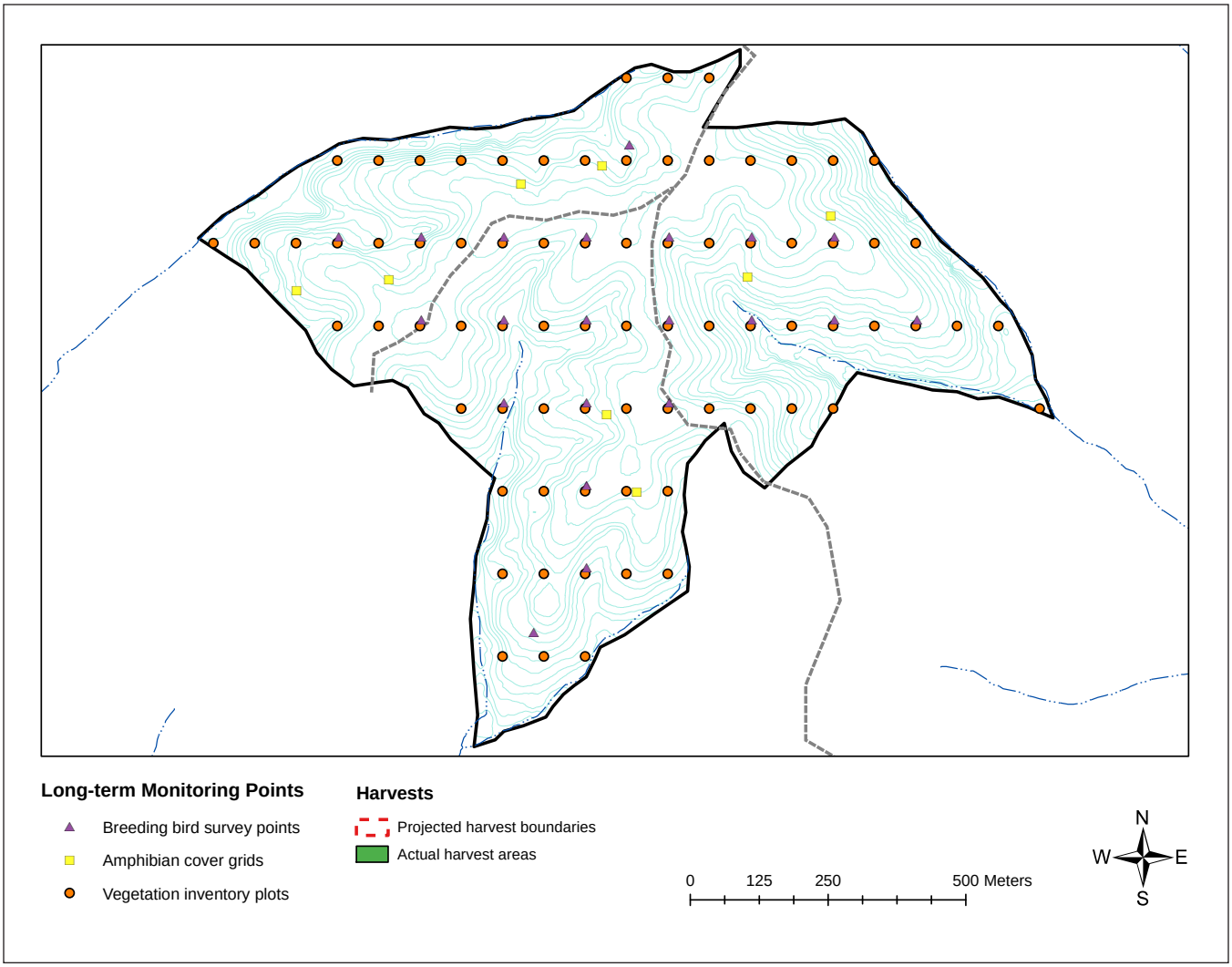


Figure 10.—Map of HEE Research Core 5 (control-no harvest). Information is summarized as in Figure 8.

Buffer Areas

The buffer areas around each of the nine research cores are managed by DoF according to its existing management practices. Management is standard among all buffers regardless of the management in the research core. Single-tree selection should bring stocking levels to between 70 and 75 percent. If it is deemed silviculturally appropriate for a group opening, an opening should be between 0.4 ha and 1.2 ha and the total area cut in group selections within a DoF management tract should not exceed 5 percent of the management tract area. Buffer area group openings may not be created within 100 m of any research core.

TIMBER HARVESTS

Timber was posted for closed bid on 17 April 2008 at Morgan-Monroe State Forest. To account for the differences in logging techniques, the units were divided into three blocks (Morgan-Monroe, Yellowwood East, Yellowwood West), each including a NH, UEA, and EA unit (Table 3). Timber was sold in blocks, and the buyer agreed to use the same crew on both the UEA and EA units within a block. UEA units were opened beginning 15 July 2008; buyers were asked to begin harvesting as soon as possible. UEA harvests had to be completed and closed before harvesting could begin in the block's EA unit. All harvesting was to be completed and closed by 28 February 2009. The six 4.05-ha shelterwood harvest areas were omitted from this sale because they received a preparatory cut that focused on understory trees and lacked sufficient merchantable timber to warrant harvesting.

All harvests followed Indiana best management practices for water quality, including water bar installation and proper skid trail and landing layouts (IDNR-DoF 2005). At all harvest sites, small-diameter slash and tree tops were left on site. Snags were not felled unless necessary for safety concerns. Post-harvest firewood cutting could not be regulated and was therefore prohibited at all sites.

Harvest sizes for the EA and UEA regeneration openings are listed in Tables 4 and 5, respectively. Areas of actual openings were calculated in ArcGIS v. 9.3.1 following on-the-ground mapping of the harvest boundaries in 2009. Single-tree removals were not mapped.

Table 3.—HEE management unit assignments and harvest dates for each timber sale block.

Block name	Total board feet marked for sale[a]	UEA unit	Start date	Close date	EA unit	Start date	Close date
Morgan-Monroe	742,149	1	15 July 2008	8 Sept. 2008	3	15 Nov. 2008	4 Dec. 2008
Yellowwood West	425,978	8	6 Oct. 2008	20 Nov. 2008	6	1 Dec. 2008	6 Jan. 2009
Yellowwood East	442,334	7	9 Sept. 2008	3 Oct. 2008	9	4 Oct. 2008	17 Oct. 2008

[a] The six 4.05-ha shelterwood areas were omitted from this sale because they lacked enough merchantable timber to warrant a logging company's entering the site.

Table 4.—Details for each harvest area in the three HEE even-aged management units. Types include clearcut and shelterwood harvests. Projected harvest size refers to the harvest areas that were digitally created in ArcGIS. Actual harvest size was calculated following on-the-ground mapping of the harvest boundaries in 2009, post-harvest (see Figs. 2 through 4). Shelterwood harvest areas will be mapped following the mid-story removal stage, so actual harvest sizes are not currently available. All areas were calculated in ArcGIS version 9.3.1.

Unit	Harvest #	Harvest type	Aspect	Projected harvest size (ha)	Actual harvest size (ha)
3	1301	shelterwood	NE	4.05	n/a
	1302	clearcut	NE	4.05	4.35
	1303	shelterwood	SW	4.05	n/a
	1304	clearcut	SW	4.05	4.43
6	1601	clearcut	NE	4.05	2.72
	1602	shelterwood	NE	4.05	n/a
	1603	clearcut	SW	4.05	3.70
	1604	shelterwood	SW	4.05	n/a
9	1901	clearcut	NE	4.05	4.05
	1902	shelterwood	NE	4.05	n/a
	1903	shelterwood	SW	4.05	n/a
	1904	clearcut	SW	4.05	4.25

Table 5.—Details for each harvest area in the three HEE uneven-aged management units. All harvest types are patch cuts only; single-tree selection harvests are not included in this table. Projected harvest size refers to the harvest areas that were digitally created in ArcGIS. Actual harvest size was calculated following on-the-ground mapping of the harvest boundaries in 2009, post-harvest (see Figures 5 through 7). All areas were calculated in ArcGIS version 9.3.1.

Unit	Harvest #	Aspect	Projected harvest size (ha)	Actual harvest size (ha)
1	1101	NE	0.40	0.46
	1102	NE	1.21	1.57
	1103	SW	0.40	0.59
	1104	SW	2.02	2.39
	1105	SW	0.40	0.58
	1106	SW	1.21	1.29
	1107	NE	2.02	1.97
	1108	NE	0.40	0.64
7	1701	NE	2.02	2.56
	1702	NE	0.40	0.15
	1703	NE	0.40	0.49
	1704	SW	0.40	0.54
	1705	NE	1.21	1.03
	1706	SW	2.02	2.28
	1707	SW	1.21	1.49
	1708	SW	0.40	0.62
8	1801	SW	0.40	0.70
	1802	NE	2.02	1.72
	1804	NE	1.21	1.25
	1805	NE	0.40	0.51
	1806	SW	1.21	1.60
	1807	SW	0.40	0.54
	1808	NE	0.40	0.65
	1813	SW	2.02	1.88

PHOTOGRAPHIC DOCUMENTATION

Photographic documentation points were first established in 2008 in Units 1, 2, and 3. The photographs are representative of each type of harvest method: clearcut (Fig. 11), shelterwood (Fig. 12), patch cutting (Fig. 13), single-tree selection (Fig. 14), and control (Fig. 15). Nearly all of the photographic points are associated with long-term vegetation plots that will quantitatively monitor the changes documented in the photographs over time. The points are permanently marked in the field, allowing a photographer to return to the exact location every year. Photographs are taken using a tripod with level indicators centered over the point marker. Using a compass, the photographer centers the initial image on north. Photographs then proceed clockwise (eastward) for six images, each with enough overlap to allow the images to be aligned. Points are generally located on level ground along the edge of the harvest areas looking downslope into the harvests. The first photographs were taken in May 2008, pre-harvest, by John Maxwell of the IDNR. Maxwell returned in November 2008 to take the immediate post-harvest pictures. This routine will be repeated each spring prior to full leaf-out. The individual photos are digitally stitched together to create a 360° panorama. These images can then be aligned to compare the changes within a forest opening over time. Due to the nature of this process, small areas of distortion within the images are possible. Although every effort is made to exactly replicate the conditions for taking each photograph, some deviation is possible; therefore images from year to year may not line up exactly.

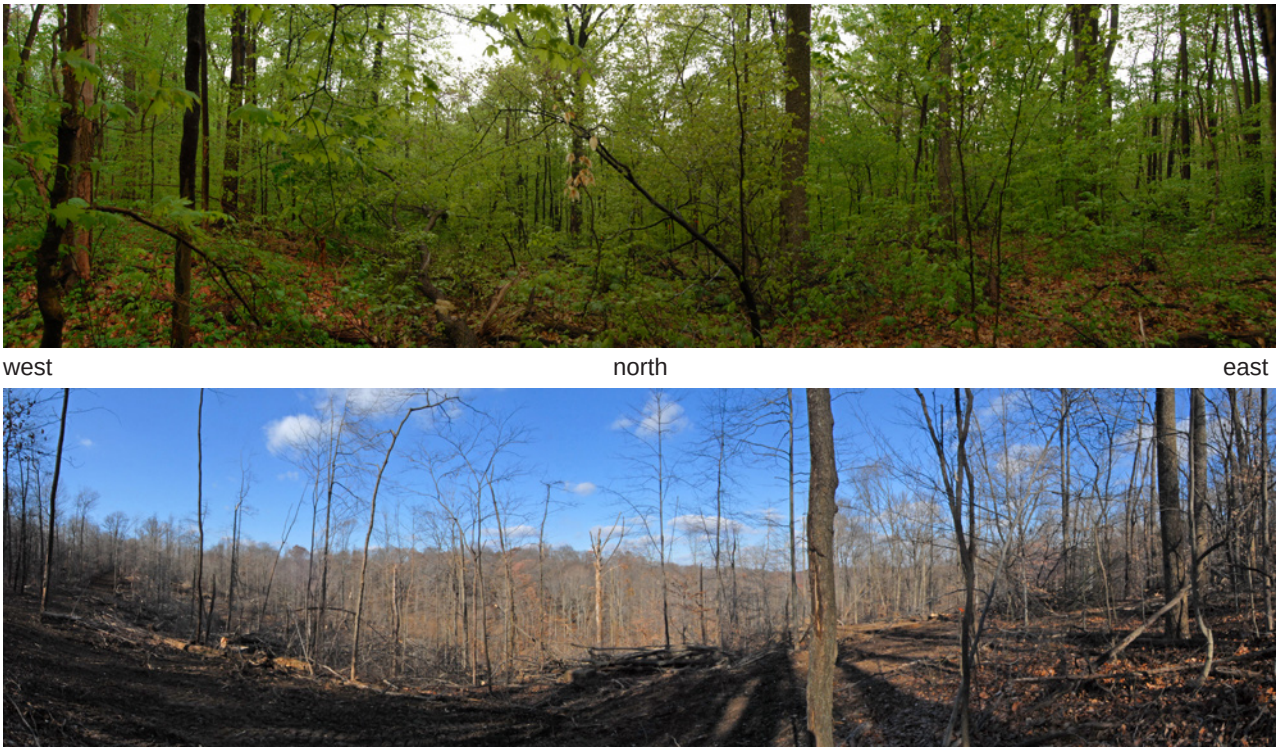


Figure 11.—Panorama images from HEE Unit 3 (even-aged management), Harvest 1302 (4.35-ha clearcut). Top image is pre-harvest, May 2008. Bottom image is immediately post-harvest, November 2008. These images are panoramas of the harvest area only and do not include the uncut forest to the south. The true 360° panorama does not accurately reflect the large size of this particular harvest area. (Photo credit: John Maxwell, Indiana Department of Natural Resources)

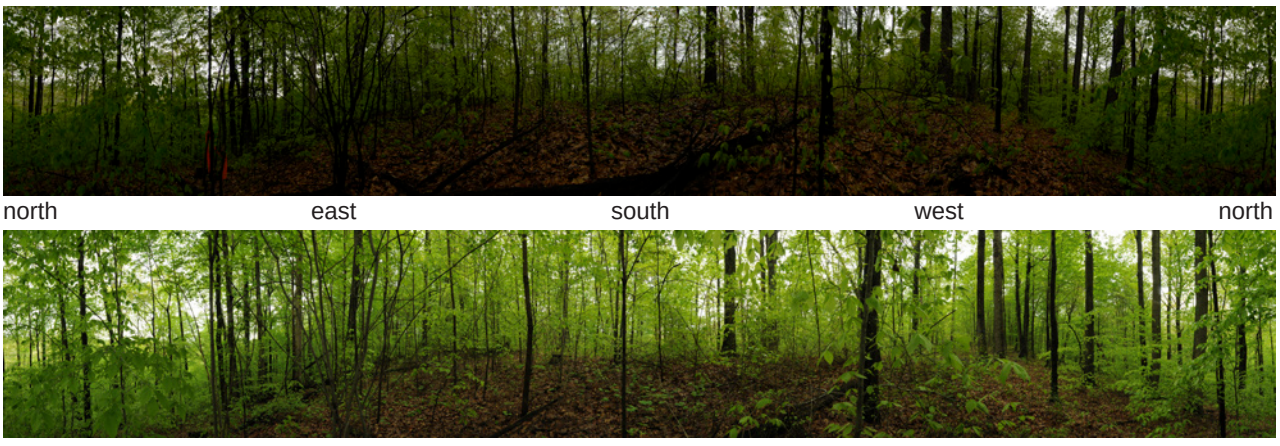


Figure 12.—360° panorama images from HEE Unit 3 (even-aged management), Harvest 1301 (shelterwood). Top image is May 2008, pre-harvest. Bottom image is immediately post-harvest, May 2009. (Photo credit: John Maxwell, Indiana Department of Natural Resources)



Figure 13.—360° panorama images from HEE Unit 1 (uneven-aged management), Harvest 1106 (1.29-ha patch cut). Top image is May 2008, pre-harvest. Bottom image is immediately post-harvest, November 2008. (Photo credit: John Maxwell, Indiana Department of Natural Resources)

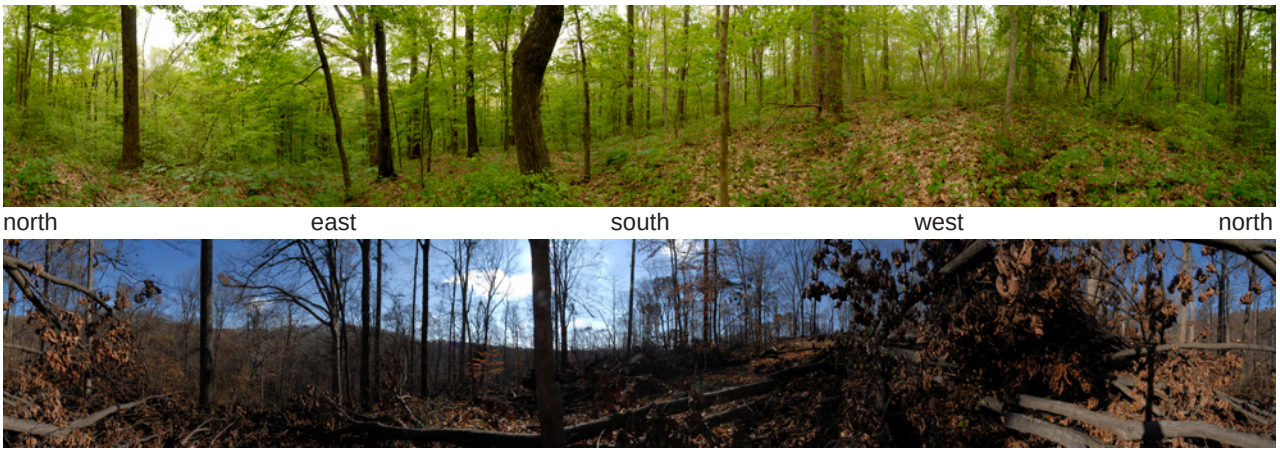


Figure 14.—360° panorama images from HEE Unit 1 (uneven-aged management), single-tree selection. Top image is May 2008, pre-harvest. Bottom image is immediately post-harvest, November 2008. (Photo credit: John Maxwell, Indiana Department of Natural Resources)



Figure 15.—360° panorama images from HEE Unit 2 (control – no harvest). Top image is May 2008, pre-harvest. Bottom image is immediately post-harvest, November 2008. (Photo credit: John Maxwell, Indiana Department of Natural Resources)

RESEARCH PROJECTS

With the inception of the HEE in 2006, research projects designed for the long-term monitoring of several species and taxa were implemented. Other research projects were added in 2007, and more continue to be added every year. Baseline data collection began immediately with each project and continued until the implementation of the harvests. Post-harvest data collection began immediately after harvests. Table 1 provides a comprehensive listing of all field research projects associated with the pre-treatment period of the HEE.

Select forest vertebrates and invertebrates will continue to be monitored after silvicultural treatments to determine faunal responses at individual, population, and community levels. Survey techniques will be taxon-specific. Species selected for study include a range of taxa that represent both species of management concern and species indicative of microhabitat conditions and successional stages. Target species of management concern include Indiana bat (*Myotis sodalis*) (Sheets, Duchamp, et al., this publication; Sheets, Whitaker, et al., this publication), cerulean warbler (Islam et al., this publication), eastern box turtle (*Terrapene carolina carolina*) (Currylow et al., this publication), and timber rattlesnake (*Crotalus horridus*) (MacGowan and Walker, this publication). Important habitat indicators include wood thrush (*Hylocichla mustelina*), worm-eating warbler (*Helmitheros vermivorum*), hooded warbler (*Setophaga citrina*), red-backed salamander (*Plethodon cinereus*), shrews (*Sorex*, *Blarina brevicauda*), and native rodent species (MacNeil and Williams, this publication; Malloy and Dunning, this publication; Urban and Swihart, this publication). Survey plots for all taxa are distributed throughout the research cores to include the treatments, the adjacent forest edge, and the untreated forest.

Also essential to the HEE is long-term vegetation monitoring. To monitor the landscape-level changes in forest canopy structure and overstory recruitment by oak and other highly valued tree species in response to forest management over time, 683 vegetation inventory plots were installed across the 9 research cores using a 75m x 150m grid. Complete inventories of all plots are designed to follow a 4-year schedule (Saunders and Arseneault, this publication).

The second primary objective of the HEE is to determine the impacts of these systems on populations of herbaceous plants and avian and terrestrial amphibian species groups. Maps showing the locations of long-term monitoring points for these target groups are provided in Figures 2 through 10.

Through these many projects, the HEE provides educational and employment opportunities for students from across the United States and Canada. Nine graduate students from three universities were involved with the HEE during the pre-treatment period. Also during this time, the HEE employed 43 students to conduct summer field work; several students returned for multiple years. The HEE provides a unique opportunity for students to gain practical experience in a broad array of field-based research activities.

SUMMARY

The mission of the HEE is to examine the ecological, economic, and social ramifications of forest management systems that include EA, UEA, and NH. Compositional changes can be seen only over time. The long-term design of the HEE will allow for comparing these different treatments to address the role of forest management in the succession from oak-hickory to beech-maple forests while quantifying the responses of plants and animals to timber harvesting (Saunders and Swihart, this publication). It is equally important to understand the economic and social implications of these ecological changes. Our project will utilize outreach techniques to disseminate research findings to state agencies, non-profit groups, and the general public (MacGowan et al., this publication).

It is not one vector, but the combination of vectors—ecological, economic, and social—that will dictate the direction of forest management in the future (Haulton, this publication).

ACKNOWLEDGMENTS

This paper is a contribution of the Hardwood Ecosystem Experiment, a partnership of the Indiana Department of Natural Resources, Purdue University, Ball Sate University, Indiana State University, Drake University, Indiana University of Pennsylvania, and The Nature Conservancy. Funding for the project is provided by the Indiana Division of Forestry and Purdue University, Department of Forestry and Natural Resources. The authors thank past HEE project coordinator Ben Dolan for his dutiful record-keeping. R. Swihart, S. Haulton, R. Jensen, and an anonymous reviewer provided helpful comments for improving this manuscript. M. Saunders provided additional silvicultural advice. J. Arseneault assisted with map creation.

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APPENDIX 1. HARDWOOD ECOSYSTEM EXPERIMENT OBJECTIVE TREE.

The objectives and hypotheses for the HEE are categorized as primary, secondary, and tertiary. Primary objectives are those that are considered fundamental to the study, the investigation of which will run the duration of the study and will be given top priority for funding. Secondary hypotheses or objectives are of long-term significance but are not necessarily funded as a permanent part of the project. Tertiary hypotheses or objectives are short-term in scope and are tested around the needs of the primary and secondary hypotheses or objectives. Secondary and tertiary hypotheses or objectives can be nested within or unrelated to a primary objective.

Primary Objectives:

1. Develop even-aged and uneven-aged silvicultural systems that maintain oak-dominated forest communities and landscapes.

   Related Secondary/Tertiary Hypotheses or Objectives:

   1.1. The spatial distribution of advanced oak regeneration is related to the spatial distribution and abundance of overstory species and edaphic factors.

   1.2. Silvicultural systems that promote development of advanced oak regeneration will lead to higher oak stocking in the successive stand than those systems relying on seedlings established after canopy removal.

   1.3. Acorn production will increase for trees with canopies adjacent to harvest areas due to increased access to resources caused by competitive release.

   1.4. Small mammal dispersal of acorns from harvest edges into openings will decline with distance from the harvest edge but increase with age of the opening. Dispersal will be more likely for black oak (*Quercus velutina*) than white oak (*Quercus alba*) due to differences in germination schedules.

   1.5. Wood-boring beetles will respond to timber harvest and have direct influences on tree mortality and stand succession.

   1.6. Seed and seedling predators (e.g., deer and small mammals) will have a significant impact on the abundance, growth, and spatial distribution of oak in the successive stand.

      1.6.1. Small mammals will have a greater impact on oak regeneration in larger canopy openings than in small openings or the intact matrix.

      1.6.2. Deer herbivory will lead to lower stocking of oak in the post-treatment stand, dependent on local deer density.

      1.6.3. Interiors of large clearcuts will suffer less damage from deer than areas along edges or small openings.

2. Determine both the positive and negative impacts of these systems on populations of herbaceous, avian, and terrestrial amphibian species groups.

   Related Secondary/Tertiary Hypotheses or Objectives:

   2.1. Relative abundance and species richness of terrestrial salamanders will remain the same or decrease in post- compared to pre-harvest sampling periods in treatment areas, and will be lower in clearcut and group selection openings compared to treatments with canopy remaining.

2.2. Abundance and species richness of terrestrial salamanders will remain the same or decrease along a gradient from the forest interior to a recent clearcut interior.

2.3. Cerulean warbler (*Setophaga cerulea*) populations will be less affected by group selection cuts than by even-aged treatments.

2.4. One objective is to locate successful breeding habitat of the cerulean warbler in relation to canopy openings created by even-aged or uneven-aged management techniques.

2.5. Shrews (*Blarina brevicauda* and *Sorex* spp.) will be less affected by group selection cuts than by even-aged treatments.

2.6. Lepidopteran community resistance to disassembly will be proportional to level of timber harvest; community resilience will be related to speed of oak and hickory regeneration.

2.6.1. Alpha diversity will recover within harvest sites faster than beta-diversity; shifts in the rank order of the most abundant taxa will generate heterogeneity in species dominance compared to pre-harvest.

2.7. Changes in lepidopteran species assemblages will generate broader shifts in trophic structure within even-aged management units. Bottom-up trophic control of bird guild structure and bat foraging activity will be proportional to magnitude of reduction in moth species diversity.

2.8. Ecological assembly rules for butterfly and moth metacommunity structure will be more significantly related to niche parameters (e.g., shifts in host plant availability post-harvest) than dispersal parameters.

2.9. Wood-boring beetle assemblages will change with disturbance, and the dominance of species attacking living trees or recently dead trees will increase post-management. Thereafter, assemblages will shift over time and approach controls.

2.10. Bat species specialized for interior forest will be negatively affected by the even-aged treatments, but other bat species will be positively affected by the openings created by the even-aged treatments. All bat communities will be less affected by group selection cuts than even-aged treatments.

2.10.1. Bat diets will not be affected by either type of treatment despite the effect on the communities because bats choose from specific prey species and do not simply choose whatever is available.

3. Determine the social and economic ramifications of these systems in both the local and regional communities.

Related Secondary/Tertiary Hypotheses or Objectives:

3.1. Public acceptance of uneven-aged management will be greater than that of even-aged management.

3.2. Economic benefits of ecosystem services (e.g., carbon sequestration) will more fully inform forest management policy and improve how harvest concessions are allocated across the state of Indiana and in the Midwest.

3.3. Attitudes regarding silviculture will vary with informational intervention and duration of exposure to treatments.

4. Provide demonstration sites and develop novel educational materials and techniques to engage the public concerning forest management.

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.

TARGETED WILDLIFE SPECIES

THE CERULEAN WARBLER IN MORGAN-MONROE AND YELLOWWOOD STATE FORESTS, INDIANA: PRE-TREATMENT DATA ON ABUNDANCE AND SPATIAL CHARACTERISTICS OF TERRITORIES

Kamal Islam, Kyle J. Kaminski, Margaret M. MacNeil, and Lila Prichard Young[1]

Abstract.—The cerulean warbler (*Setophaga cerulea*) is a small Neotropical migrant songbird that is currently experiencing population declines. We monitored the number of cerulean warblers, as well as size and spatial attributes of territories, before two methods of silvicultural treatments took place in six of nine study units involved in the Hardwood Ecosystem Experiment within the Morgan-Monroe and Yellowwood State Forests in southern Indiana. Data were collected during the 2007 and 2008 breeding seasons. Relative abundance was apparently greater in 2008, but this increase was not consistent at each individual site. Territory sizes also increased from 2007 to 2008 and this trend was consistent across most study sites. Territories were spatially clustered in both years. These findings are similar to other research involving cerulean warblers in southern Indiana. There were no significant differences in the number of cerulean warbler detections or the size of territories among the three study groups (control, even-aged, and uneven-aged) for the two combined pre-treatment years. Future studies will examine how cerulean warblers respond to silvicultural prescriptions.

INTRODUCTION

Cerulean warbler (*Setophaga cerulea*[2]) is a small migratory songbird with a breeding range covering much of the Midwest, northeastern portions of the United States, and parts of Ontario and Quebec (Hamel 2000a, Robbins et al. 1992). It winters primarily on the slopes of the Andes in South America (Hamel 2000a). Cerulean warbler has experienced such long-term population declines at both the local and range-wide levels as to become a species of concern (Hamel 2000a, Robbins et al. 1992). For the past 40 years, this species has experienced annual population declines estimated at 4.1 percent (Sauer et al. 2008) based on Breeding Bird Surveys. This decline represents the greatest annual rate of decline of any North American wood warbler and songbird (Jones et al. 2008, Robbins et al. 1992). As a result, it is listed as a top priority species for conservation action by Partners in Flight (Hamel et al. 2004). It has been listed as "Vulnerable" by the International Union for Conservation of Nature (Birdlife International 2004) and was petitioned for listing as a threatened species under the Endangered Species Act (U.S. Fish and Wildlife Service 2002). However, the petition to list was denied (U.S. Fish and Wildlife Service 2006).

In Indiana, cerulean warblers declined by 2.7 percent over the last 40 years (Hands et al. 1989). From 1980 through 1998, Breeding Bird Surveys conducted in Indiana indicated a relative abundance estimate of 0.29 birds/route for the entire survey period (Hamel 2000b). This abundance value was the lowest for any state in which an estimate could be calculated (Hamel 2000b). Cerulean warblers, once considered common

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[2] Formerly *Dendroica cerulea*

summer breeders in Indiana, were listed as a "species of special concern" (Bruner 1998) but are now listed as endangered in the state (Indiana Department of Natural Resources-Division of Nature Preserves 2010).

The cerulean warbler is an area-sensitive species (Hamel 2000a, Robbins et al. 1992) with a preference for large contiguous tracts of mature deciduous forest. Robbins et al. (1992) concluded that 700 ha was the minimum tract size of forest for cerulean warblers in the Middle Atlantic states. In Wisconsin, however, cerulean warblers could be located in tracts ranging from as small as 16 ha to larger than 32 ha (Hands et al. 1989), suggesting that minimum tract size requirements may differ between regions. In Indiana, cerulean warblers are most commonly associated with forests that exhibit heterogeneous canopy structure with a preference for large, dominant trees (Jones and Islam 2006). Although the geographic limits of the breeding range for cerulean warblers have changed little over time, the abundance of the species in its breeding range has declined (Hamel 2000a). Much of the deciduous forest that makes up the breeding range throughout the Midwest has been converted for agricultural use (Hands et al. 1989), resulting in loss of habitat and further fragmentation of remaining habitat. Northern Indiana has been severely fragmented and now has less than 10 percent forest cover, while southern Indiana has nearly 80 percent forest cover (Fauth 1999).

Despite long-term declines in cerulean warblers' overall relative abundance, some authors have documented that the species is expanding to the northeast (Oliarnyk and Robertson 1996, Robbins et al. 1992, Sauer et al. 2003), possibly due to changes in climate and land use. In this region, large areas of agricultural land have been allowed to succeed to mature forests (Hamel 1992), which suggests that forest management practices have the potential to benefit cerulean warblers.

Few studies have examined cerulean warblers' response to silvicultural treatments[3] (Hamel et al. 2006, 2010). However, some evidence suggests that large-scale habitat alterations can be detrimental to this species. In Arkansas and Ontario, ice storms negatively affected cerulean warbler relative abundance (Hamel et al. 1998, Jones et al. 2001). In West Virginia, cerulean warblers occurred at a lower abundance and frequency within 15- to 18-year-old regenerating clearcut stands compared to 70- to 80-year-old mature second-growth forests (Wood et al. 2006). Hamel et al. (2004) indicated that studying cerulean warblers' response to experimental habitat manipulation is an important way to determine which characteristics of vegetation structure are preferred by the birds and how to produce these characteristics on the landscape.

The objectives of our study were to explore the abundance and spatial characteristics of cerulean warbler territories in the Hardwood Ecosystem Experiment (HEE) before silvicultural treatments. Our specific objectives were to determine:

1. If cerulean warbler relative abundance differed among the three groups (control, uneven-aged, and even-aged),
2. If cerulean warbler territory sizes differed among the three groups (control, uneven-aged, and even-aged), and
3. The spatial distribution of cerulean warbler territories across the landscape of the HEE.

STUDY AREA

Nine management units, ranging from 303 to 483 ha in size, were established in southern Indiana, with four units in Morgan-Monroe State Forest and five units

[3] Cerulean Warbler Conservation Initiative, Joint AFO/COS/WOS conference, Kearney, NE, 9-13 March 2011, personal communication.

in Yellowwood State Forest (Kalb and Mycroft, this publication: Fig.1). Within each management unit, a 1.96-km^2 (196-ha) study site was overlain over a 200-ha core area. Pre-treatment data at each of the nine study sites were collected from 1 May - 8 August in 2007 and 2008.

MATERIALS AND METHODS

Point Count Surveys

Each study unit was overlain with a 7 point x 7 point grid study site with points 200 m apart for a total of 49 survey points (Fig. 1). We conducted point counts at each study site once per year using a 100-m radius at each survey point to locate male warblers. Survey points were spaced 200 m apart to minimize detection of the same bird twice. According to Jones et al. (2000) and Hamel et al. (2009), male cerulean warblers were found to broadcast their songs up to 100 m. The edge of each study site was surrounded by a 50-m buffer to minimize edge effects. Surveys were carried out between 05:30 and 10:30 EST. Surveys were not conducted during periods of rain, wind, or any other phenomenon that could have prevented vocalizations from being heard.

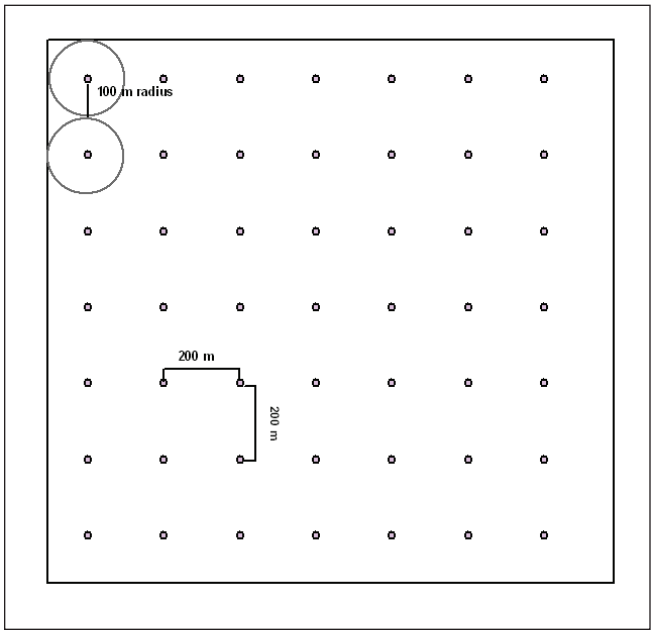


Figure 1.—Protocol established for cerulean warbler (*Setophaga cerulea*) surveys within nine study units located in Morgan-Monroe and Yellowwood State Forests in southern Indiana during pre-treatment years of 2007 and 2008. Seven transects, 200 m apart, were surveyed. Each transect contained seven survey points.

Observers listened for vocalizations for 2 minutes after arriving at the point, used playback of cerulean warbler songs for 1 minute, and then listened for an additional 2 minutes (Falls 1981). Compass direction and an estimate of the singer's distance were recorded for every vocalization to distinguish between multiple individuals. At each survey point, males were detected aurally and/or visually. The number of males detected at each study site was divided by the total area of each study site (196 ha) to determine number of males/km^2 for each unit. Point counts were conducted between 1 May - 8 June 2007 and 2008. To eliminate potential bias between time (day of month) of bird survey at a particular study site and the number of cerulean warbler detections, the order of survey of study sites was determined by drawing from a random numbers table for each year.

Territory Mapping

To demarcate territories, survey points where cerulean warblers were detected for a given year were revisited from 1 June - 15 July in 2007 and 2008. Trees where male birds were observed singing during the territory demarcation period (1 June - 15 July) were flagged with tape. Not all territories were demarcated in 2008 because of the early initiation of silvicultural treatments. Global Positioning System (GPS) coordinates of the trees were recorded, as well as tree species. GPS points were recorded as Universal Transverse Mercator (UTM) units in a Garmin GPS 76 handheld unit (Garmin International, Inc., Olathe, KS). Territories were considered to be completely mapped when 4-12 trees were flagged, depending on territory size.

Whenever possible, we revisited known territories multiple times before final demarcation of the territory's perimeter was determined. Depending on the individual bird, territory marking took several hours to several days. The coordinates of all points collected while demarcating territories were imported into ArcMap (ESRI, Redlands, CA). These data were used to create territory polygons using the minimum convex polygon method. Distances between the centroids of each territory and the nearest neighboring territory were used as part of a nearest neighbor analysis (Krebs 1989), which was performed using the average nearest neighbor tool in ArcMap.

Data Analysis

Normal probability plots of cerulean warbler abundances for the two pre-treatment combined years (2007 and 2008), generated in Minitab16© (Minitab, Inc., State College, PA) for the control sites and each of the treatment sites (even-aged and uneven-aged study sites), were normal. However, territory size data for each of the three groups were negatively skewed and these data were transformed to approximate a normal distribution. One-way analysis of variance (ANOVA) was performed to determine if there were differences in the number of cerulean warbler detections and size of territories among even-aged, uneven-aged, and control sites for the pre-treatment years. ANOVAs were performed with Minitab16© at $\alpha = 0.05$ level to test for significance.

RESULTS

Two hundred eighteen male cerulean warblers were detected in 2007 (94) and in 2008 (124) (Table 1) in the nine sites surveyed. Cerulean warblers were located in all nine units for the combined pre-treatment years with the number of individual birds identified per unit ranging from 0 to 47. Relative abundance estimates ranged from 0 to 23.98 males/km^2, averaging 6.18 males/km^2 (Table 1). Although the number of detections increased overall from 2007 to 2008, this trend was not consistent at each individual site. Sites 3 and 8 had the most detections and sites 1 and 2 had the fewest detections over the 2-year period (Table 1). However, there was no significant difference ($F = 2.57$, $df = 215$, $P = 0.11$) in the number of detections among the three groups for the two combined years.

Table 1.—Yearly cerulean warbler relative abundance in Yellowwood and Morgan-Monroe State Forests in southern Indiana according to site and harvest type for the pre-treatment years of 2007 and 2008. n = Number of males detected and R.A. = Relative abundance (males/km^2).

	2007		2008	
Site/harvest	n	R.A.	n	R.A.
Uneven-aged				
1	0	0.00	5	2.55
7	15	7.65	15	7.65
8	40	20.41	47	23.98
Even-aged				
3	9	4.59	20	10.20
6	8	4.08	4	2.04
9	9	4.59	13	6.63
Control				
2	1	0.51	4	2.04
4	4	2.04	9	4.59
5	8	4.08	7	3.57
Total	94	5.33	124	7.03

In 2007 and 2008, 120 cerulean warbler territories were demarcated (Table 2, Appendix 1). The number of territories demarcated at each study site ranged from 0 to 24. Overall territory sizes averaged from 0.11 ha to 0.86 ha (Table 2). We were unable to sample all territories in 2008 due to the initiation of the silvicultural treatments, which explains the apparent decrease in the number of territories for that year (see Materials and Methods: Territory Mapping above). There was no significant difference ($F = 0.29$, $df = 217$, $P = 0.75$) in the size of territories among the three groups for the two combined years. Mean distance of territories to the nearest neighboring territory in 2007 and 2008 was 126.5 m and 188.4 m, respectively. Based on the results of a nearest neighbor analysis, 83 percent of the territories were clustered in 2007 and 52 percent were clustered in 2008. These results suggest that most territories were spatially clumped and not randomly or uniformly distributed across the landscape.

Table 2.—Yearly cerulean warbler territory sizes in Yellowwood and Morgan-Monroe State Forests in southern Indiana according to site and harvest type for the pre-treatment years of 2007 and 2008. Values reported as mean ± 1 S.D.

	2007		2008	
Site/harvest	# of territories	Area (ha)	# of territories	Area (ha)
Uneven-aged				
1	0	0	1	0.44 ± 0
7	7	0.18 ± 0.08	7	0.86 ± 0.59
8	24	0.18 ± 0.17	21	0.19 ± 0.10
Even-aged				
3	8	0.18 ± 0.14	13	0.28 ± 0.16
6	6	0.16 ± 0.15	0	0
9	9	0.13 ± 0.13	5	0.31 ± 0.27
Control				
2	1	0.11 ± 0	3	0.40 ± 0.11
4	4	0.40 ± 0.46	2	0.44 ± 0.04
5	6	0.11 ± 0.07	3	0.69 ± 0.24
Total	65	0.18 ± 0.19	55	0.36 ± 0.34

DISCUSSION

The average estimate of 6.18 males/km^2 for the two pre-treatment years, although slightly higher, is comparable to previous density estimates in southern Indiana in the Hoosier National Forest, and Morgan-Monroe and Yellowwood State Forests. In 2000 and 2001, relative abundance estimates for these forested areas varied from 0 birds/km^2 to 3.9 birds/km^2 and the 2-year average for 2004 and 2005 was 3.58 males/km^2 (Islam and Basile 2002, Islam et al. 2006). A study at Big Oaks National Wildlife Refuge indicated an average relative abundance estimate of 1.78 birds/km^2 for 2002 and 2003 (Islam and Roth 2004). Despite the overall decline in cerulean warbler populations, density estimates in southern Indiana for the past 8 years have been similar. These results are a good indication that the population is continuing to persevere in some of Indiana's southern forests.

The number of territories and size of each territory differed among sites, with an average size of 0.28 ha. These results are similar to and within the range of territory sizes from previous studies (Islam and Basile 2002). In the Hoosier National Forest, and Morgan-Monroe and Yellowwood State Forests, mean territory size (n=59) was 0.34 ha (Islam and Basile 2002). At Big Oaks National Wildlife Refuge (n=51), mean territory size was estimated at 0.21 ha (Islam and Roth 2004). In contrast, the mean for territories mapped (n=18) in Ontario was 1.04 ha (Oliarnyk and Robertson 1996), which is much larger than the mean territory size for this study or previous studies of territory size in southern Indiana.

According to Barg et al. (2006), territory size can be influenced by many factors, which can lead to variation in territory sizes. Factors that may affect territory size include habitat quality and resource distribution. These factors may explain why territories in this part of the cerulean warblers' breeding range are smaller than territories in Ontario.

Territories' occurring in clumps can also be an explanation for territory size. Cerulean warblers are known to exhibit clustered territoriality (Roth and Islam 2007). We found reduced clustering in 2008 as compared to 2007, but our territory sampling effort was reduced in 2008 due to the beginning of the silvicultural treatments.

There were no significant differences in the number of cerulean warbler detections or the sizes of territories among the three groups (control, even-aged, and uneven-aged treatments). These results are encouraging as any changes that occur in detection rates or the size and spatial arrangements of territories after silvicultural treatment will likely be a response to that treatment.

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APPENDIX 1.

The following sets of UTM coordinates demarcate cerulean warbler territories in Morgan-Monroe State Forest (units 1-4) and Yellowwood State Forest (units 5-9) in southern Indiana for 2007 and 2008. Projection is North American Datum 1983 UTM zone 16N.

Year	Unit	Bird name	Easting	Northing
2007	2	LP2E4	548427	4354817
			548420	4354815
			548410	4354822
			548446	4354793
			548465	4354785
			548478	4354771
			548476	4354778
			548477	4354755
			548479	4354749
2007	3	MH3E4	547978	4352263
			547995	4352332
			548020	4352270
			547921	4352230
			547913	4352253
			547903	4352230
			547946	4352249
2007	3	LP3D	547648	4351993
			547640	4351998
			547658	4351987
			547643	4351986
			547625	4351962
			547556	4351968
2007	3	MH3G2A	548279	4351991
			548298	4351984
			548293	4351996
			548273	4351998
			548303	4351997
			548331	4352000
			548320	4352028
			548319	4352022
2007	3	MH3G2B	548331	4352029
			548326	4352035
			548332	4352056
			548345	4352047
			548324	4352062
			548334	4352042
			548342	4352090
2007	3	MH3G3	545350	4352164
			548367	4352131
			548393	4352063
			548397	4352084
			548397	4352078
			548388	4352154
			548385	4352149
			548361	4352180
2007	3	MH3F4	548160	4352366
			548111	4352356
			548104	4352333
			548091	4352324
			548108	4352326
			548084	4352336
			548058	4352344
2007	3	LP3EA	547866	4352147
			547862	4352164
			547881	4352203
			547864	4352198
			547872	4352200
			547869	4352192
			547894	4352168
			547841	4352183
2007	3	MH3E3B	547958	4352152
			547963	4352141
			547955	4352138
			547954	4352150
			547967	4352154
			547010	4352162
			547013	4352164
			548005	4352152
2007	4	LP4C7	549482	4351227
			549437	4351291
			549450	4351300
			549454	4351304
			549443	4351258
			549345	4351239
			549324	4351257
			549329	4351309
			549325	4351335
2007	4	N4C7	549377	4351462
			549408	4351529
			549430	4351530
			549412	4351558
			549439	4351499
			549420	4351506
2007	4	MHF6	550196	4351137
			550280	4351157
			550251	4351155
			550266	4351165
			550304	4351169
			550317	4351158
			550307	4351173

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2007	4	RBMB4G5	550372	4350873
			550357	4350878
			550360	4350882
			550362	4359878
			550361	4350857
2007	5	LP5C6	554289	4339828
			554310	4339802
			554302	4339802
			554333	4339861
			554305	4339822
			554322	4339819
			554314	4339793
2007	5	CP5D6	554396	4339728
			554397	4339724
			554402	4339713
			554413	4339731
			554410	4339731
			554397	4339725
2007	5	RS5E6	554672	4339747
			554720	4339776
			554729	4339756
			554716	4339749
			554729	4339752
			554717	4339760
			554714	4339740
2007	5	RS5G3A	555153	4339209
			555087	4339177
			555093	4339187
			555121	4339176
			555076	4339204
			555140	4339202
			555149	4339188
2007	5	RS5G3B	555096	4339175
			555063	4339170
			555038	4339150
			555086	4339173
			555035	4339148
			555096	4339147
			555059	4339118
2007	5	RS5G3C	554992	4339188
			555006	4339189
			555010	4339187
			555020	4339193
			554990	4339177
			555013	4339198
			555010	4339185
2007	6	MH6B6A	554606	4330539
			554626	4330576
			554637	4330577
			554648	4330545
			554655	4330502
			554618	4330519
			554633	4330566
2007	6	MH6B6B	554574	4330631
			554577	4330639
			554551	4330644
			554531	4330646
			554579	4330618
			554536	4330647
			554567	4330646
2007	6	MH6B6C	554533	4330685
			554501	4330674
			554516	4330689
			554527	4330683
			554517	4330675
			554537	4330673
			554528	4330672
2007	6	RS6C7A	554795	4330659
			554781	4330603
			554795	4330605
			554796	4330642
			554827	4330666
			554793	4330673
			554827	4330679
2007	6	RS6C7B	554780	4330727
			554783	4330728
			554768	4330750
			554769	4330730
			554774	4330725
			554770	4330720
			554772	4330730
2007	6	RS6D3	554956	4329828
			554957	4329897
			554903	4329853
			554945	4329841
			554907	4329856
			554974	4329850
			554926	4329945
2007	7	LP7G3B	559556	4331524
			559574	4331515
			559566	4331497
			559578	4331505
			559550	4331564
			559574	4331495
			559590	4331446
2007	7	MH7B6	558455	4332053
			558438	4332053
			558459	4332055
			558432	4332043
			558401	4332030
			558407	4332003
			558392	4331998
			558455	4332077

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2007	7	RS7B5A	558613	4331831
			558603	4331812
			558589	4331821
			558603	4331876
			558596	4331897
			558633	4331835
			558606	4331823
			558610	4431819
			558615	4331809
2007	7	RS7B5B	558430	4331756
			558429	4331764
			558410	4331774
			558448	4331722
			558436	4331725
			558468	4331612
			558464	4331618
			558422	4331762
2007	7	R57B4	558453	4331640
			558452	4331634
			558437	4331586
			558514	4331534
			558501	4331555
			558440	4331610
			558462	4331605
			558423	4331605
2007	7	LP7G3A	559635	4331588
			559661	4331607
			559625	4331590
			559630	4331586
			559656	4331557
			559660	4331541
			559656	4331538
			559685	4331655
2007	7	LP7G3C	559607	4331417
			559615	4331441
			559641	4331445
			559643	4331432
			559636	4331470
			559632	4331485
			559624	4331498
			559631	4331437
2007	8	MH8G4	559071	4329676
			559109	4329734
			559126	4329752
			559128	4329743
			559111	4329720
			559112	4329725
			559105	4329745
2007	8	MH8G5	559085	4329856
			559077	4329845
			559074	4329829
			559074	4329850
			559050	4329880
			559021	4329847
			559100	4329850
			559111	4329786
2007	8	MH8E5B	558515	4329806
			558530	4329819
			558532	4329810
			558543	4329818
			558546	4329828
			558539	4329831
			558533	4329846
			558506	4329810
2007	8	MH8E5A	558638	4329738
			558630	4329731
			558644	4329728
			558626	4329782
			558635	4329763
			558618	4329766
			558650	4329765
			558632	4329762
2007	8	MH8E6	558758	4329971
			558784	4329989
			558744	4329980
			558744	4329989
			558806	4330005
			558777	4329970
			558823	4329981
2007	8	MH8G2Z	559151	4329263
			559151	4329254
			559151	4329258
			559151	4329234
2007	8	RS8B5S	557894	4329678
			557914	4329720
			557891	4329721
			557847	4329645
			557847	4329629
2007	8	RS8B5B	557974	4329778
			558002	4329702
			557977	4329786
			558136	4329694
			558030	4329748
2007	8	MH8F7C	558859	4330401
			558850	4330372
			558868	4330378
			558859	4330372
			558896	4330320
			558923	4330338
			558905	4330349

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2007	8	MH8F7B	558835	4330458
			558838	4330456
			558847	4330434
			558824	4330439
			558842	4330464
			558847	4330468
			558850	4330477
2007	8	MH8F6A	558830	4330018
			558829	4330043
			558801	4330027
			558820	4329956
			558814	4329920
			558784	4329914
			558763	4329916
			558837	4329921
2007	8	MH8G2A	559196	4329380
			559222	4329268
			559217	4329268
			559213	4329261
			559206	4329276
			559222	4329254
			559223	4329263
2007	8	LP8D3	558358	4329369
			558390	4329371
			558411	4329364
			558385	4329374
			558356	4329389
			558348	4329385
			558348	4329375
2007	8	LP8C3A	558186	4329529
			558177	4329538
			558175	4329565
			558171	4329558
			558167	4329534
			558179	4329527
			558129	4329544
2007	8	LP8C3B	558177	4329592
			558152	4329550
			558179	4329564
			558180	4329590
			558189	4329597
			558160	4329602
			558161	4329616
			558158	4329627
2007	8	RS8A12	557958	4329174
			557959	4329171
			557934	4329174
			557925	4329180
			557920	4329167
			557913	4329168
			557942	4329196
			557938	4329170
			557935	4329173
			557958	4329174
2007	8	RSA2A	557833	4329347
			557854	4329338
			557845	4329305
			557843	4329317
			557840	4329336
			557824	4329310
			557841	4329353
			557820	4329346
2007	8	RS8B2C	558160	4329270
			558152	4329224
			558139	4329219
			558140	4329203
			558140	4329212
			558150	4329190
			558127	4329232
2007	8	RS8B1B	558001	4329148
			557938	4329232
			557935	4329247
			557998	4329187
			558008	4329220
			558010	4329228
			558016	4329238
			558054	4329233
2007	8	LPC1A	558226	4329042
			558241	4329066
			558246	4329062
			558220	4329061
			558234	4329024
			558208	4329010
			558222	4329164
			558197	4329003
			558192	4329000
2007	8	RS8B1A	558022	4329059
			558056	4329016
			558053	4329027
			558075	4322902
			558058	4329027
			558043	4329038
			558044	4329018
2007	8	LP8B1W	558025	4328987
			558020	4328997
			558026	4328982
			558019	4328993
			558026	4328977
			558012	4328971
			558018	4328986
2007	8	LP8B1Q	558000	4329030
			557993	4329021
			558004	4329024
			557994	4329050
			558003	4329020
			558003	4329054
			558008	4329061
			557988	4329061

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2007	8	RS8B1P	557978	4329126
			557968	4329132
			557949	4329131
			557955	4329105
			557967	4329077
			557971	4329085
			557972	4329090
2007	9	MH9B23	560777	4332355
			560797	4332364
			560830	4332326
			560831	4332324
			560817	4332360
			560898	4332240
			560875	4332253
			560886	4332238
2007	9	LP9A5A	560718	4332804
			560712	4332806
			560733	4332790
			760743	4332789
			760714	7332773
2007	9	LP9A5B	560729	4332688
			560728	4332705
			560732	4332705
			560730	4332716
			560723	4332704
			560726	4332680
			560731	4332681
			560756	4332729
			560711	4332721
			560715	4332730
2007	9	LP9A5C	560567	4332857
			560568	4332843
			560584	4332840
			560571	4332862
			560579	4332871
			560577	4332873
			560534	4332857
2007	9	LP9A5D	560595	4332769
			560590	4332786
			560586	4332783
			560576	4332784
			560596	4332795
			560593	4332793
			560593	4332798
2007	9	LP9B3A	560816	4332457
			560793	4332477
			560786	4332467
			560791	4332478
			560789	4332473
			560814	4332446
			560791	4332446
2007	9	MH9B3C	560785	4332404
			560763	4332394
			560767	4332398
			560777	4332384
			560804	4332389
			560413	4332435
			560784	4332385
			560463	4332378
2007	9	LP9A5Z	560616	4332747
			560618	4332750
			560639	4332738
			560638	4332761
			560624	4332751
			560635	4332760
			560646	4332755
2007	9	MH9B3B	560720	4332492
			560726	4332489
			560701	4332510
			560753	4332476
			560741	4332455
			560735	4332400
			560728	4332407
			560758	4332402
			560746	4332388

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2008	1	102	547888	4355833
			547903	4355853
			547863	4355852
			547809	4355898
			547910	4355914
			547884	4355906
2008	2	233	548353	4354874
			548472	4354845
			548453	4354873
			548417	4354882
			548491	4354783
			548489	4354808
2008	2	239	548580	4354690
			548557	4354739
			548546	4354697
			548587	4354719
			548606	4354711
			548608	4354675
			548627	4354680
2008	2	246	548786	4354730
			548804	4354742
			548731	4354723
			548800	4354731
			548838	4354720
			548871	4354712
			548777	4354752
			548791	4354695
2008	3	247	547416	4352483
			547409	4352524
			547389	4352517
			547421	4352490
			547401	4352465
			547383	4352484
			547361	4352477
			547356	4352482
			547355	4352453
2008	3	329	547960	4351698
			547944	4351749
			547960	4351783
			547971	4351819
2008	3	331	547883	4352174
			547886	4352170
			547877	4352170
			547935	4352186
			547928	4352214
			547862	4352276
			547879	4352342
			547862	4352276
			547866	4352173
			547870	4352201
2008	3	335	547897	4352881
			547921	4352859
			547957	4352855
			547892	4352863
			547934	4352878
			547949	4352884
2008	3	336	547826	4352875
			547754	4352895
			547837	4352896
			547770	4352841
2008	3	343	548414	4351706
			548452	4351744
			548428	4351645
			548431	4351595
			548456	4351593
			548422	4351603
			548441	4351680
2008	3	344	548363	4351983
			548414	4351946
			548396	4351920
			548388	4351932
			548363	4351975
			548353	4351958
			548381	4351982
			548323	4351981
2008	3	345	548323	4352184
			548382	4352131
			548377	4352178
			548360	4352174
			548373	4352160
			548388	4352162
			548364	4352140
2008	3	3455	548291	4352127
			548330	4352108
			548314	4352086
			548322	4352097
			548315	4352083
			548359	4352101
			548360	4352119
			548323	4352112
2008	3	346	548313	4352314
			548252	4352283
			548264	4352334
			548269	4352270
			548239	4352316
			548325	4352277
			548302	4352311
			548302	4352293
2008	3	3466	548406	4352359
			548373	4352378
			548366	4352336
			548376	4352371
			548351	4352367

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2008	3	338	548214	4352500
			548252	4352494
			548269	4352546
			548275	4352525
2008	3	347	548423	4352441
			548426	4352446
			548505	4352467
			548515	4352467
			548504	4352423
			548465	4352459
			548449	4352439
			548387	4352485
2008	4	414	549324	4350015
			549345	4350020
			549363	4349970
			549339	4349938
			549361	4349923
			549321	4349897
			549373	4349998
			549338	4350029
2008	4	443	550221	4350205
			550218	4350217
			550183	4350199
			550177	4350207
			550246	4350185
			550222	4350196
			550212	4350213
			550225	4350172
			550229	4350153
			550311	4350151
2008	5	532	554606	4339350
			554594	4339328
			554554	4339324
			554627	4339300
			554719	4339323
			554673	4339306
2008	5	539	554848	4339286
			554852	4339299
			554805	4339271
			554872	4339274
			554786	4339250
			554733	4339207
			554858	4339241
			554871	4339226
2008	5	545	555045	4339063
			555080	4339123
			555092	4339213
			555051	4339178
			555084	4339182
			555065	4339207
			555111	4339205
			554984	4339199
2008	7	7233	558607	4331792
			558608	4331824
			558610	4331763
			558613	4331720
			558544	4331820
			558570	4331840
			558521	4331793
2008	7	730	558479	4331954
			558561	4331980
			558480	4332073
			558352	4332103
			558387	4331994
2008	7	719	559143	4331397
			559148	4331389
			559100	4331500
			559057	4331368
			559107	4331352
			559182	4331351
			559197	4331524
2008	7	706	559447	4330998
			559434	4331040
			559367	4331116
			559373	4331123
			559405	4331112
			559467	4330997
			559536	4331022
2008	7	714	559538	4331265
			559557	4331295
			559571	4331340
			559535	4331376
			559520	4331419
			559475	4331407
			559515	4331289
2008	7	715	559537	4331406
			559548	4331400
			559557	4331372
			559592	4331375
			559599	4331384
			559598	4331373
			559606	4331400
			559565	4331401
2008	7	721	559504	4331450
			559485	4331437
			559477	4331417
			559572	4331465
			559581	4331480
			559504	4331441
			559505	4331460

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2008	8	801	557930	4329041
			557930	4329012
			557905	4329026
			557926	4329023
			557959	4329051
			557974	4329045
			557960	4329036
			557969	4329040
2008	8	8011	557975	4329028
			557984	4329052
			557990	4329049
			557975	4329028
			557991	4329026
			557987	4329025
			558018	4329060
			557977	4329075
2008	8	802	558046	4329016
			558051	4329037
			558091	4329012
			558074	4329018
			558113	4329042
2008	8	813	558016	4329177
			558029	4329167
			558052	4329174
			558060	4329148
			558031	4329125
			558055	4329095
			557982	4329148
2008	8	8266	558149	4329621
			558145	4329632
			558170	4329650
			558145	4329624
2008	8	818	558566	4329444
			558532	4329450
			558584	4329414
			558616	4329420
			558624	4329446
			558626	4329420
			558642	4329375
2008	8	826	558202	4329590
			558211	4329597
			558167	4329595
			558172	4329581
			558220	4329576
			558150	4329582
			558175	4329561
			558133	4329578
2008	8	8265	558168	4329526
			558193	4329520
			558175	4329517
			558174	4329529
			558196	4329542
			558181	4329519

Year	Unit	Bird name	Easting	Northing
2008	8	832	558479	4329823
			558490	4329815
			558471	4329846
			558512	4329829
			558506	4329867
			558524	4329815
			558553	4329822
			558482	4329857
2008	8	833	558460	4329853
			558424	4329877
			558401	4329886
			558465	4329876
			558451	4329882
			558410	4329866
			558486	4329903
2008	8	839	558487	4330019
			558440	4330025
			558442	4330016
			558432	4330092
			558446	4330037
			558503	4330585
			558510	4330019
2008	8	805	558647	4329049
			558636	4329091
			558638	4329150
			558647	4329096
			558640	4329111
			558626	4329056
			558673	4329072
			558623	4329117
2008	8	810	558605	4329125
			558598	4329143
			558593	4329166
			558572	4329143
			558579	4329116
			558605	4329094
			558638	4329150
			558598	4329132
			558590	4329080
			558605	4329130
2008	8	825	558542	4329653
			558583	4329655
			558563	4329612
			558542	4329661
			558556	4329668
			558551	4329633
			558560	4329631
2008	8	833	558620	4329807
			558609	4329814
			558600	4329761
			558630	4329762
			558638	4329766
			558626	4329789
			558601	4329740
			558553	4329822

APPENDIX 1. (continued)

Year	Unit	Bird name	Easting	Northing
2008	8	809	558813	4329206
			558838	4329208
			558792	4329188
			558815	4329157
			558833	4329234
			558803	4329222
			558802	4329180
			558821	4329190
2008	8	819	558797	4329418
			558820	4329443
			558779	4329482
			558813	4329411
			558787	4329403
			558818	4329400
			558810	4329411
2008	8	820	558964	4329365
			558984	4329352
			558953	4329314
			558926	4329390
			558931	4329365
2008	8	821	559124	4329424
			559082	4329428
			559109	4329433
			559130	4329416
			559027	4329489
			559058	4329477
			559004	4329456
2008	8	8211	559147	4329424
			559149	4329422
			559162	4329412
			559163	4329387
			559156	4329375
			559179	4329356
			559173	4329369
2008	8	8212	559130	4329416
			559140	4329444
			559155	4329438
			559156	4329431
			559151	4329453
2008	9	903	560651	4332468
			560613	4332491
			560600	4332493
			560654	4332513
			560677	4332512
			560635	4332530
			560589	4332538
2008	9	905	560629	4332807
			560626	4332819
			560641	4332793
			560668	4332810
			560652	4332826
			560647	4332824
2008	9	906	560559	4332849
			560606	4332791
			560570	4332816
			560557	4332836
			560555	4332813
			560533	4332859
			560518	4332850
2008	9	911	560813	4332543
			560801	4332553
			560814	4332540
			560795	4332545
			560787	4332533
			560795	4332558
			560753	4332542
			560782	4332521
2008	9	918	561016	4332660
			560939	4332630
			561021	4332642
			561045	4332657
			561056	4332683
			561000	4332600
			560914	4332695

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.

SPATIAL ECOLOGY AND BEHAVIOR OF EASTERN BOX TURTLES ON THE HARDWOOD ECOSYSTEM EXPERIMENT: PRE-TREATMENT RESULTS

Andrea F. Currylow, Brian J. MacGowan, and Rod N. Williams[1]

Abstract.—To understand better how eastern box turtles (*Terrapene carolina carolina*) are affected by forest management practices, we monitored movements of box turtles prior to silvicultural treatments within the Hardwood Ecosystem Experiment (HEE) in Indiana. During 2007 and 2008, we tracked 23-28 turtles on six units of the HEE. Estimated minimum convex polygon home-range sizes (range 0.8-187.7 ha) varied among units and across years but were similar among slated treatment types (control, even-aged, and uneven-aged). These data will allow us to test for short-term responses of box turtles to silvicultural practices aimed at oak (*Quercus*) regeneration.

INTRODUCTION

Timber harvests and other management practices may change the vegetation structure and microhabitat climate within forests and can affect a wide variety of forest wildlife (Goldstein et al. 2005, Gram et al. 2003, Renken et al. 2004, Sullivan and Sullivan 2001). Although some data suggest that logged areas are associated with moderate increases in bird and reptile diversity (Fredericksen et al. 2000), it is not clear whether this observation can be considered a general trend. Recent attempts to assess the effects of timber harvests on ectothermic species often suffer from a lack of replication or comparable pre-harvest data (e.g., Goldstein et al. 2005, McLeod and Gates 1998). The majority of these herpetofaunal studies focused on harvest effects on amphibian populations (e.g., Hocking and Semlitsch 2008, Knapp et al. 2003, Rittenhouse and Semlitsch 2009, Semlitsch et al. 2009), whereas relatively little is known about the impacts on reptile populations.

Moreover, existing data suggest not only that reptiles are sensitive to habitat perturbations, but that the impacts are more pervasive and severe than for amphibians (Gibbons et al. 2000). Behavioral and population changes of reptiles relative to habitat alterations are often difficult to assess because of their stochastic patterns of activity, large home ranges, and low densities. The Hardwood Ecosystem Experiment (HEE) site is rich in reptilian diversity, with more than 20 species known to occur within its boundaries. The eastern box turtle (*Terrapene carolina carolina*) offers potential in studying responses to timber harvesting given its distribution within and use of forested habitats.

The eastern box turtle is a relatively long-lived reptile (50+ years) (Stickel 1978) native to forested regions across the eastern United States. Box turtles have experienced substantial population declines over the past several decades, particularly in areas dominated by agriculture (Hall et al. 1999, Stickel 1978, Williams and Parker 1987). Causes for these declines likely include habitat loss and degradation,

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but the response of box turtles to forest management is largely unknown. There is some evidence that breaks in forest canopy cover may be beneficial for thermoregulation (Hallgren-Scaffidi 1986), and other open areas such as meadows or agricultural fields are often used as nesting sites (Dodd 2001, Nazdrowicz et al. 2008). Areas harvested for timber may also increase local food sources and cover items such as herbaceous biomass relative to unharvested sites (Perison et al. 1997).

Other evidence suggests that box turtles are sensitive to environmental disturbances that affect local habitat features (Currylow 2011, Dodd 2001). Therefore, annual losses of a relatively small proportion of adults may result in a gradual decline toward local extirpation (Belzer 2002, Doroff and Keith 1990). The eastern box turtle, however, is the longest-lived reptile to occur throughout the eastern forests. Further investigation is needed to understand how timber harvests affect this K-selected species of conservation concern during both active and inactive seasons.

We examined the effects of even-aged and uneven-aged forest management on eastern box turtles as part of the HEE. Our overall objectives of the project were to:

1. Determine box turtle home-range size and movement patterns prior to harvests;
2. Examine the short-term effects of timber harvests on box turtle home range, movements, habitat use, and thermal ecology during the active season;
3. Characterize the hibernal thermal behavior of box turtles within a managed forest landscape; and
4. Estimate the adult annual survival rate of eastern box turtles using radio-telemetry.

This paper describes the methods and results of pre-harvest data collection (objective 1), which occurred in 2007 and 2008. For detailed descriptions of objectives 2 through 4, refer to Currylow (2011) and Currylow et al. (2012), respectively.

STUDY AREA

The study area was located within Morgan-Monroe State Forest (MMSF) and Yellowwood State Forest (YSF) in Morgan, Monroe, and Brown Counties in south-central Indiana. The area is characterized by steep ridges and valleys covered with a mixture of oak-hickory (*Quercus-Carya*) and American beech-maple (*Fagus grandifolia-Acer*) forests. The study area and HEE study design are described in detail by Kalb and Mycroft (this publication). Box turtle pre-treatment data collection took place on units 4-9 during 2007 and 2008.

MATERIALS AND METHODS

Radio-Telemetry and Home Range

We initially located adult box turtles by visual encounter surveys and opportunistically while conducting other field work. Upon initial capture, we recorded sex, weight (to nearest 10 g), and carapace length and width (to nearest 0.1 cm). We assigned a unique ID number and marked each turtle using a triangular file along marginal scutes following a modified Cagle scheme (Cagle 1939, Ernst et al. 1974, Ferner 2007). For the initial location site and all subsequent locations, we recorded the date, observer, slope aspect, ambient temperature (°C) and humidity, elevation, ground temperature (°C) and humidity, Universal Transverse Mercator coordinates (North American Datum 1983, Zone 16N), general weather condition (cloud cover and precipitation), and general condition of the turtle.

To obtain estimates of home-range size, we affixed a radio transmitter (model RI-2B Holohil Systems, Ltd., Carp, ON) to the carapace of 4-6 turtles in each unit. Sex ratios and numbers of turtles were equally divided among sites and management classes when possible. We radio-tracked (by homing) 23-28 turtles 2-3 times per week during at least one active season (May through October). For each tracked location, we recorded the location data as described above. We calculated 100-percent Minimum Convex Polygons

(MCP) to estimate the home-range area of each turtle (Hawth's Analysis Tools, ArcGIS 9.0, ESRI, Redlands, CA). Turtles with <20 locations were omitted from home-range analyses for that active season (Row and Blouin-Demers 2006).

Habitat

From 5 June to 15 August 2007, we measured habitat characteristics for every other turtle location and a random location (see below). We estimated percentage cover for vegetation ≤1.0 m tall, coarse woody debris (≥10 cm diameter), water, rock, and bare ground within a 2-m-radius circular plot using a sight tube (James and Shugart 1970). The sight tube consisted of a polyvinyl chloride pipe 15 cm long with two black thread crosshairs on one end. An observer held the sight tube 1 m above and perpendicular to ground level and walked a series of four transects 4 m in length across the plot. Along the N-S and E-W transects, the observer recorded the number of hits (plant material at the center of the crosshairs) in the sight tube beginning at the start and every 0.5 m along both transects, skipping the plot center after the first transect. Along the remaining NW-SE and NE-SW transects, the observer stopped every 1 m along each transect for 25 observations per plot. Numbers of hits in the crosshairs of the tube were summed for each habitat variable and multiplied by 4 to estimate percentage cover.

To assess how habitat at turtle locations differed from available habitat, we collected the same data as above for random points. Distance (30-100 m) and azimuth (1-360°) from a given turtle location were randomly determined a priori. Although these plots are not truly random because distance was restricted, they reasonably reflected the habitat available to each turtle for that day based on typical daily movements of box turtles (Iglay et al. 2007) rather than habitat available across the entire study site.

RESULTS

We located and processed 236 turtles during the pre-harvest period and tracked the movements of 28 turtles for an average of 36 locations each (range = 14-45) during at least 1 active season (Table 1). Estimated home-range sizes were variable (range 0.8-187.7 ha; Table 1) and some turtles made atypical movements in one or both years. Turtle #406 exhibited transient behavior (Dodd 2001) in both years and was excluded from further comparisons. During an extended hot and dry period in 2007, turtle #603 moved 1.2 km from unit 6 and remained along a lake margin from 25 July to 6 September. We did not observe this behavior during 2008 or subsequent years (Currylow 2011). The resultant MCP home range (54.6 ha) was thus inflated and not included in subsequent comparisons. The removal of turtles #406 and #603 reduced the variation in home-range sizes considerably (range 0.8-22.1 ha).

We pooled home ranges by year for each proposed treatment type to determine if box turtle spatial use differed among control, even-aged, and uneven-aged units prior to timber harvests. Home-range size differed little between sex or proposed treatments (Fig. 1), but differed with respect to year and unit when pooled by sex (Fig. 2). Except for units 7 and 9, home-range size was larger during 2008. Mean home-range size within unit 8 was more than double the mean size for all units considered together.

SUMMARY AND FUTURE WORK

Mean home-range sizes were much larger than earlier studies and nearly double the largest published estimates (for a complete review, see Currylow 2011). Home ranges were not consistent between years (Fig. 2), but were fairly comparable among treatment types pooled across years (Fig. 1) and among units, with the exception of unit 8 (Currylow 2011: Fig. 2). Prolonged hot and dry conditions during summer 2007 may have contributed to annual differences but do not explain differences among units within the same year.

Table 1.—Box turtle home-range estimates[a] on HEE units 4-9 during the pre-harvest period (2007 and 2008).

Turtle ID#	Unit	Sex	Weight[b] (g)	2007		2008	
				Number of locations	Home range (ha)	Number of locations	Home range (ha)
400	4	F	470	38	1.6	lost	
402		M	320	35	1.8	38	5.6
404		F	390	34	2.3	39	4.8
406		M	435	31	130.9	32	187.7
500	5	M	455	28	2.0	42	2.8
503		F	410	33	1.6	42	10.0
504		M	510	31	1.4	43	2.0
506		F	420	32	1.0	40	1.5
602[c]	6	F	465	43	1.6	24	-
610[c]		F	300			11	-
603		F	400	40	54.6	42	3.1
605		M	490	45	1.8	40	9.2
607		M	490	43	3.6	42	4.9
615		M	600			34	2.0
616		F	420			41	2.7
700	7	M	365	35	2.7	35	1.0
704		M	395	33	4.3	39	1.4
706		F	535	29	2.0	38	2.1
708		M	475	39	2.9	36	1.2
20	8	M	415	23	7.1	33	22.1
800		F	405	38	2.6	39	16.5
806		M	455	14	-	40	2.6
814		M	440			20	0.8
848		M	460			37	1.2
900	9	F	470	41	4.1	33	3.0
904		M	240	39	1.4	36	1.5
906		F	530	39	5.4	36	5.0
908		M	455	40	1.3	34	1.7
Mean[d]					2.6		4.6

[a] 100-percent Minimum Convex Polygon (Hawth's Analysis Tools, ArcGIS 9.0)

[b] Weight at initial capture in grams

[c] Turtle #602 died on July 11, 2008 from injury of unknown source and was replaced by #610.

[d] Calculations of means do not include turtle #406 or #603.

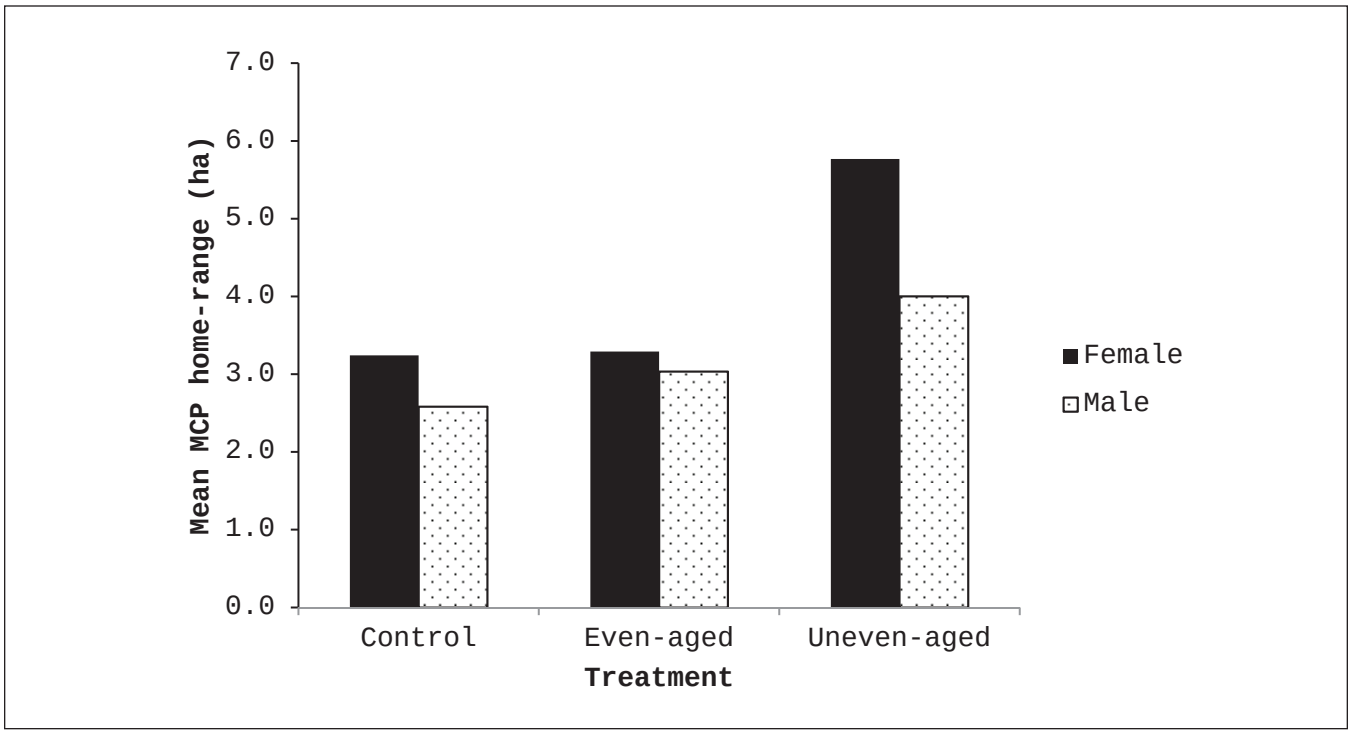


Figure 1.—Mean home-range sizes (100-percent Minimum Convex Polygon) for box turtles in control (N=8), even-aged (N=11), and uneven-aged (N=8) HEE management units during the pre-harvest period (2007 and 2008).

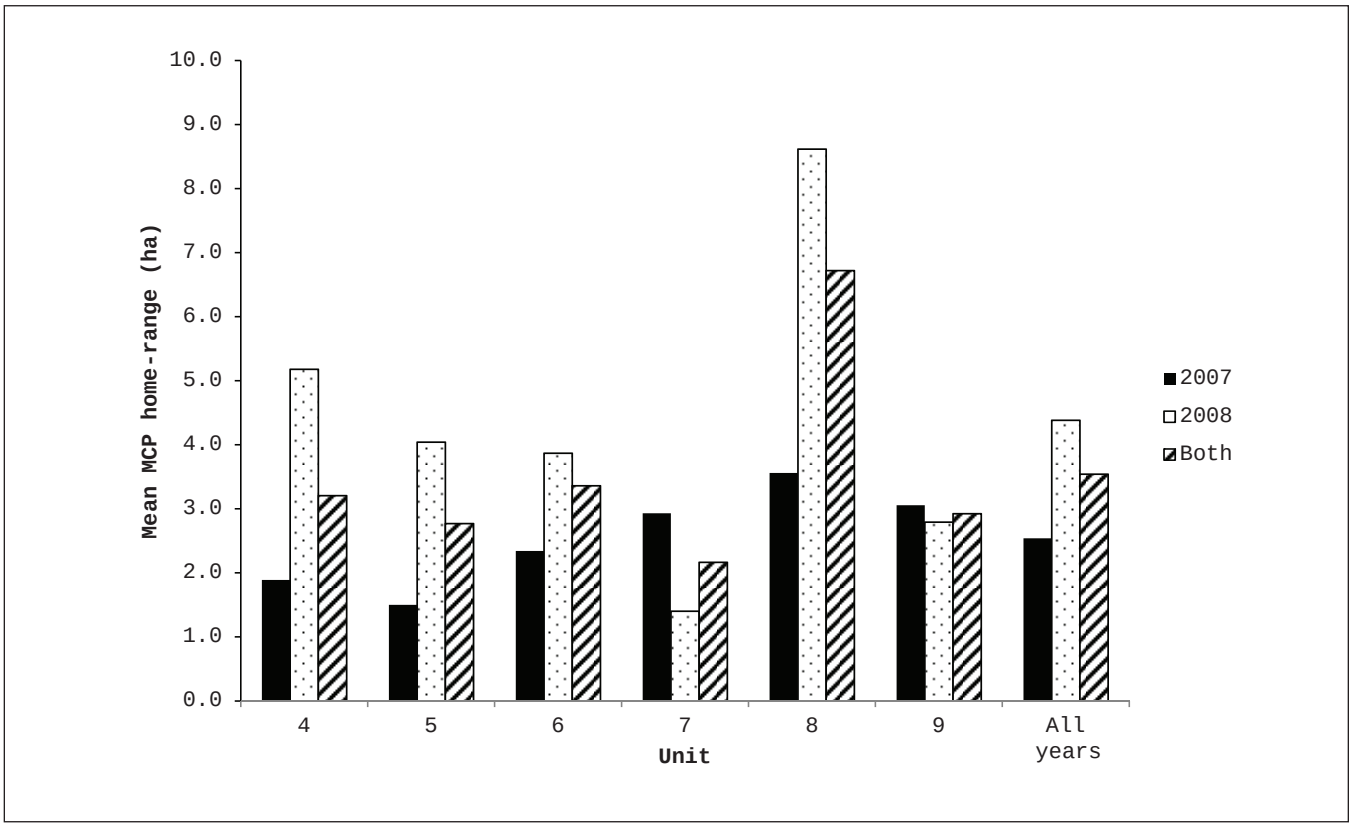


Figure 2.—Mean home-range sizes (100-percent Minimum Convex Polygon) for box turtles in HEE units 4-9 by year during the pre-harvest period. Data for sexes are pooled.

During the pre-harvest period, one female on unit 4 was lost during winter 2007-08 and another female on unit 6 died from injuries caused by an unknown predator. Currylow et al. (2011) estimated the annual survival of box turtles in the HEE study area during hibernal and active periods (2007-09). Their overall estimate of annual survival was 0.964 and was only slightly higher during the active period.

A primary goal of our work was to examine the short-term effects of timber harvests on box turtles. We continued to collect data during the post-harvest period until October 2010 as we tracked 50 turtles in at least one active season across all nine HEE units. We grouped sites by management class (clearcut, patch cut, and control) and evaluated treatment effects in the pre- and post-harvest data using a full factorial generalized linear mixed model with year, sex, management class, and their interactions as fixed effects and animal ID nested in year as a random effect to find any differences in annual MCP home ranges with relation to harvests (Currylow 2011).

Temperature plays an important part in ectotherm ecology; thus, we also studied thermoregulatory behavior of turtles during the post-harvest period (Currylow 2011; Currylow et al. 2012). We affixed iButton temperature dataloggers (model DS1921G-F5#, Maxim Integrated Products, Inc., Sunnyvale, CA) to the carapace of tracked turtles. Dataloggers recorded temperatures every 45 minutes during the active season (May-October) in 2009 and 2010. We randomly placed dataloggers at four locations within each HEE unit, resulting in 18 dataloggers in harvest openings and uncut areas combined. In a similar manner, we placed datalogger profiles at box turtle hibernacula and random sites within and outside of harvest openings on even-aged units during the hibernal season. We assessed the impact of clearcuts on hibernation depth and the availability of potential hibernation sites in clearcuts based on temperature profiles at various depths and slope aspects (Currylow et al. 2012).

This study increases our understanding of box turtles' responses to timber harvesting as well as the general ecology of box turtles. Sampling across the relatively large geographic area encompassed by the HEE, the collection of pre- as well as post-harvest data, the inclusion of thermal behavior during both active and inactive seasons, and the relatively long study duration (4 years) are key strengths to the study. Our results will be a valuable contribution to our understanding of how different silvicultural treatments impact box turtles.

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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.

SPATIAL ECOLOGY OF TIMBER RATTLESNAKES ON THE HARDWOOD ECOSYSTEM EXPERIMENT: PRE-TREATMENT RESULTS

Brian J. MacGowan and Zachary J. Walker[1]

Abstract.—The timber rattlesnake (*Crotalus horridus*) is a species of conservation concern throughout much of its geographic range and may serve as a sentinel species in investigations of the effects of timber harvesting on forest reptiles. Our objective was to determine the effect of even-aged timber management regimes on timber rattlesnake home range and movements. During pre-treatment data collection in 2007 and 2008, we tracked 23 rattlesnakes on 4 units of the Hardwood Ecosystem Experiment in Indiana. Home-range sizes of male rattlesnakes (65.7 ha) were greater than non-gravid (20.6 ha) and gravid (17.6 ha) females. Home ranges were generally consistent among units and between treatments. These data will allow us to test for immediate responses of timber rattlesnakes to even-aged timber management.

INTRODUCTION

Forest management agencies are under increasing pressure to justify land management practices, especially those that involve traditional silvicultural methods. Moreover, third-party certification of public and private forests has increased the need to identify negative impacts of timber harvesting on wildlife populations and adjust management plans accordingly. Forest wildlife of particular interest are area-sensitive species and species of conservation concern. The long-term sustainability of these species largely relies on public forest lands because of the fragmented ownership and turnover of private forest lands across the eastern United States (see Birch 1996).

The timber rattlesnake (*Crotalus horridus*) may serve as a sentinel species for long-term studies that examine the impacts of silvicultural practices, largely because of its behavioral and demographic characteristics. Behaviorally, timber rattlesnakes maintain strong fidelity to hibernacula, and they remain near these den sites during ingress and egress (Brown et al. 1982, Gibson 2003, Walker 2000). Gravid females tend to stay near hibernacula at birthing rookeries during the entire active season (Brown 1991, Martin 1993). Demographically, females do not reach sexual maturity until at least 7 to 11 years of age and reproduce every 3 to 5 years (Brown 1991, Martin 1993). Thus, timber harvesting occurring near den sites could have long-lasting ramifications for rattlesnake populations if it results in higher mortality or alterations of den sites.

Timber rattlesnakes are endangered throughout much of their range (Martin 1982), including Indiana, where they are restricted primarily to the Highland Rim Natural Region (Homoya et al. 1985) in forested areas with suitable den sites. Previous research on timber rattlesnakes in Indiana has provided insights into spatial ecology, den site selection, and habitat

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use (Gibson 2003, Gibson et al. 2008, Walker 2000) in forests not managed for timber. Although researchers have gained a better understanding of the ecology of this species in Indiana and other parts of its range, information regarding responses to human activities is limited. Reinert et al. (2011) observed some logging-related mortality (<2 percent) of timber rattlesnakes, but resultant habitat changes did not alter snake behavior and movement patterns 1 year post-harvest. In the long term, understanding how forest management alters timber rattlesnake habitat use and survival is important for sustaining their populations within Indiana and the Central Hardwoods Region.

We examined the effect of even-aged silviculture on timber rattlesnakes as part of the Hardwood Ecosystem Experiment (HEE). Our objectives were to:

1. Estimate timber rattlesnake home-range size and movement patterns prior to harvests;
2. Determine the immediate effects of timber harvests on timber rattlesnake home range, movements, and habitat use;
3. Estimate timber rattlesnake adult annual survivorship during the active and hibernal seasons;
4. Determine the genetic variation and structure of timber rattlesnakes; and
5. Assess den characteristics during the hibernal period.

This paper describes the methods and results of pre-harvest conditions of timber rattlesnakes on the HEE sites (objective 1) during 2007 and 2008.

STUDY AREA

The study area was located within Morgan-Monroe State Forest (MMSF) and Yellowwood State Forest (YSF) in Morgan, Monroe, and Brown Counties in south-central Indiana. The area is characterized by steep ridges and valleys composed of a mixture of oak-hickory (*Quercus-Carya*) and American beech-maple (*Fagus grandifolia-Acer*) forests. The study area and HEE study design are described in detail by Kalb and Mycroft (this publication). Timber rattlesnake pre-treatment data collection took place on units 2, 5, 6, and 9 during 2007 and 2008.

MATERIALS AND METHODS

Snake Sampling

During parts of the active season (20 April to 15 August) in 2007 and 2008, we located snakes using three methods. First, we actively searched for timber rattlesnakes in appropriate habitat on and adjacent to HEE units. Specific locations were based on 2006 sightings by HEE personnel and recommendations by State Forest staff. Second, rattlesnakes were located opportunistically during this period by our field crew and other HEE field technicians. Finally, from 21 April to 16 May 2008, we trapped selected den sites on or near HEE units 2, 5, and 6 using fiberglass window screening covering den site entrances. All sides were secured to the ground using a combination of garden staples, logs, and rocks. At a single opening we secured a funnel trap constructed of 1.3-cm hardware cloth with a canvas funnel at one end. Traps were checked daily until their removal after egress from the dens.

All snakes were handled using snake hooks and handling bags. Upon initial capture of each snake, we measured total body length (TBL), snout to vent length (SVL) (both to the nearest 1 cm), and tail length (TL) (to the nearest 0.1 cm), and determined sex (via TL and/or cloacal probe), age (neonate TBL <20 cm; juvenile 20 cm ≤ TBL ≤ 80 cm; adult TBL >80 cm), and weight to the nearest 10 g (1 g for neonates). We implanted a Passive Integrated Transponder tag (AVID Identification Systems, Inc., Norco, CA) subcutaneously under the 7th ventral scute anterior from the vent within each snake and painted a portion of the base rattle section white to facilitate identification in the field. A small ventral scale clip

was collected from each snake for subsequent genetic analysis. For each snake location, we recorded the date, slope aspect, Universal Transverse Mercator coordinates, elevation, general weather, and snake behavior upon location (feeding, basking, hiding, or traveling). We also measured ambient and ground temperature and humidity using a handheld weather meter (Kestrel 3000, Nielsen Kellerman, Boothwyn, PA).

Radio-Telemetry and Home Range

We tracked movements of a subset of timber rattlesnakes using radio-telemetry. We marked the location of capture with flagging and transported snakes from the field to a local Indiana Department of Natural Resources office. Purdue University veterinarians implanted radio transmitters (type AI-2T 25g; SI-2T 13.5g and 9.0g, Holohil Systems, Ltd., Carp, ON) using a method modified from Reinert and Cundall (1982). Transmitter selection was based on weight and girth of each snake. Following a 48- to 96-hour recovery time, snakes were returned to their exact point of capture and released.

Rattlesnakes were tracked by typical homing methods usually three times per week during the active season (18 April to 25 October). Coordinates were taken at the location of the snake or, when the snake was not visible, at its hiding location (e.g., log pile, hollow log). Data for all snake locations were recorded as described above.

We calculated 95-percent Minimum Convex Polygon (MCP) to estimate the home-range area of each snake (Hawth's Analysis Tools, ArcGIS 9.0, ESRI, Redlands, CA). We chose this method because MCPs allow comparison to previous research. Two snakes captured late during the 2008 season (≤5 locations) were omitted from home-range analyses.

Habitat

In 2007 (10 May to 25 July) we measured habitat characteristics for a subset of timber rattlesnake locations. Habitat data were collected after each snake had moved but within 2 weeks of the location date to prevent disturbance to the snake and to minimize vegetative differences between the dates of actual location and vegetation sampling. We focused our sampling on habitat elements most important for timber rattlesnakes (Gibson 2003, Walker 2000). We estimated percentage cover of vegetation ≤1.0 m tall, coarse woody debris (≥10 cm diameter), water, rock, and bare ground in a 2-m-radius circular plot using a sighting tube (James and Shugart 1970). The sighting tube consisted of a polyvinyl chloride pipe 15 cm long with two black thread crosshairs on one end. An observer held the sighting tube 1 m above and perpendicular to ground level and walked a series of four transects 4 m in length across the plot. Along the N-S and E-W transects, the observer recorded the number of hits in the sighting tube beginning at the start and every 0.5 m along both transects, skipping the plot center after the first transect. Along the remaining NW-SE and NE-SW transects, the observer stopped every 1 m along each transect for 25 observations per plot. Numbers of hits in the crosshairs of the tube were summed for each habitat variable and multiplied by 4 to estimate percentage cover.

Percentage canopy cover was measured in a similar way using an 8-m-radius circular plot, but points were spaced 2 m apart on the N-S, E-W transects and 4 m apart on the N-S and E-W transects. For each habitat location, we collected the same data for a random point. Distance (30-100 m) and azimuth (1-360°) from each snake location were determined a priori using a random number generator. Even though these plots are not truly random because distance is restricted, they reasonably reflect the habitat available to each snake between successive locations (see Blouin-Demers and Weatherhead 2001).

Because of logistical constraints, collection of habitat data was limited to 2007. During post-harvest we will expand our sampling efforts (number of snakes and units) and data collection methodology. We will measure the distance to the nearest rock (maximum length ≥10 cm), log (maximum diameter ≥10 cm), tree

(diameter at breast height [d.b.h.] ≥7.5 cm), understory tree (d.b.h. <7.5 cm, height >2.0 m), and shrub (height <2.0 m) in four quadrants (Reinert 1984). Values will be averaged among the four quadrants for each location and random point. We also will record for logs their maximum diameter, decay class (Maser et al. 1979), and a visual estimate of the percentage lying on the ground.

Only totals and means are reported in this paper. Detailed statistical analyses of pre- and post-harvest results are presented elsewhere. Those analyses will use a generalized linear mixed model to test for differences in home range and habitat use among treatment types pre- and post-harvest (SAS Institute Inc. 2004). We will include unit, year, sex (and breeding condition), and their interactions as fixed effects and animal ID nested in year as a random effect. Habitat models also will include behavior type (e.g., feeding, resting) because of its potential influence on habitat use (e.g., downed woody debris). Moreover, use (or avoidance) of harvest openings may result from a specific habitat requirement related to timber rattlesnake biology.

RESULTS

Fifty-five rattlesnakes were captured and marked in both years. We tracked 23 rattlesnakes (10 males, 13 females) during part or all of the 2007 and 2008 active seasons in the HEE control units 2 and 5 and the HEE even-aged units 6 and 9 (Table 1). Gravid and non-gravid females were grouped separately since gravid females are known to reduce movements and shift habitat use (Brown et al. 1982, Reinert and Zappalorti 1988, Reinert et al. 2011, Walker 2000). One female we tracked was gravid in 2007; three females were gravid in 2008.

Table 1.—MCP Home-range size of timber rattlesnakes, 2007 and 2008. Values for gravid females are noted with an asterisk (*).

HEE unit (treatment)	Snake	Sex	Weight[a]	2007 Home range (ha)[b] (number of locations)	2008 Home range (ha)[b] (number of locations)
2 (control)	249	F	950		75.6 (36)
	881	F	800		- (5)
	272	F	840		28.3 (29)
	161	F	1,170	41.1 (23)	*10.7 (40)
	528	M	2,770	96.9 (23)	143.2 (37)
	122	M	930	27.4 (23)	34.6 (40)
	201	M	1,525		75.6 (25)
5 (control)	41	F	260	1.6 (22)	20.5 (37)
	961	F	560		16.1 (24)
	980	F	1,080		*12.2 (24)
	900	F	1,165		*7.6 (19)
	489	M	1,520	77.0 (34)	71.2 (38)
	510	M	2,570	60.7 (27)	90.7 (27)
	448	M	1,040		103.4 (29)
6 (even-aged)	23	F	420	5.1 (34)	8.8 (43)
	431	F	700	12.8 (53)	12.5 (44)
	401	F	730	*39.8 (46)	5.4 (43)
	292	M	1,130	92.0 (49)	54.3 (46)
	313	M	1,240	13.8 (44)	25.1 (42)
	471	M	760		7.8 (42)
9 (even-aged)	231	F	380		20.1 (34)
	80	F	700		- (4)
	1580	M	1,450		93.8 (26)

[a] Weight at initial capture, in grams.

[b] 95-percent Minimum Convex Polygon, Hawth's Analysis Tools, ArcGIS 9.0

Average MCP home-range sizes varied by sex and ranged from 1.6 to 103.4 ha (Table 1). The home-range size of males averaged 65.7 ha (SE = 9.8). The home-range of non-gravid females averaged 20.6 ha (SE = 5.9); gravid females averaged 17.6 ha (SE = 10.5). Four gravid females were tracked during the pre-harvest period; only one of these was on an even-aged unit. Thus, average home-range size of gravid snakes should be viewed cautiously. Although the number of locations per snake was variable (range 19-53, average 34) because some snakes were not tracked an entire field season, home-range size was not correlated to sampling intensity (r = 0.09).

Home-range size was generally consistent among units with two exceptions: females in unit 2 had a larger home range and males in unit 6 had a smaller home range (Fig. 1). Home-range sizes were similar when grouped by treatment type (Fig. 2). Inconsistencies among units and between treatments may be due in part to body size (i.e., larger females and males tended to be in control units, Table 1). Regardless of sex, larger snakes had larger home ranges (r = 0.73).

SUMMARY AND FUTURE WORK

This study provides information on the spatial ecology of timber rattlesnakes in a managed forest. Our estimates of home-range size were comparable to other studies conducted in New Jersey (Reinert and Zappalorti 1988), New York (Brown et al. 1982), and West Virginia (Adams 2005). However, our home-range estimates were less than half those of Walker (2000) (males = 174 ha, females = 72 ha) even though both studies were conducted within the same region of Indiana with similar topography and plant composition.

Walker (2000) studied rattlesnakes at Brown County State Park, which differs from the HEE sites with

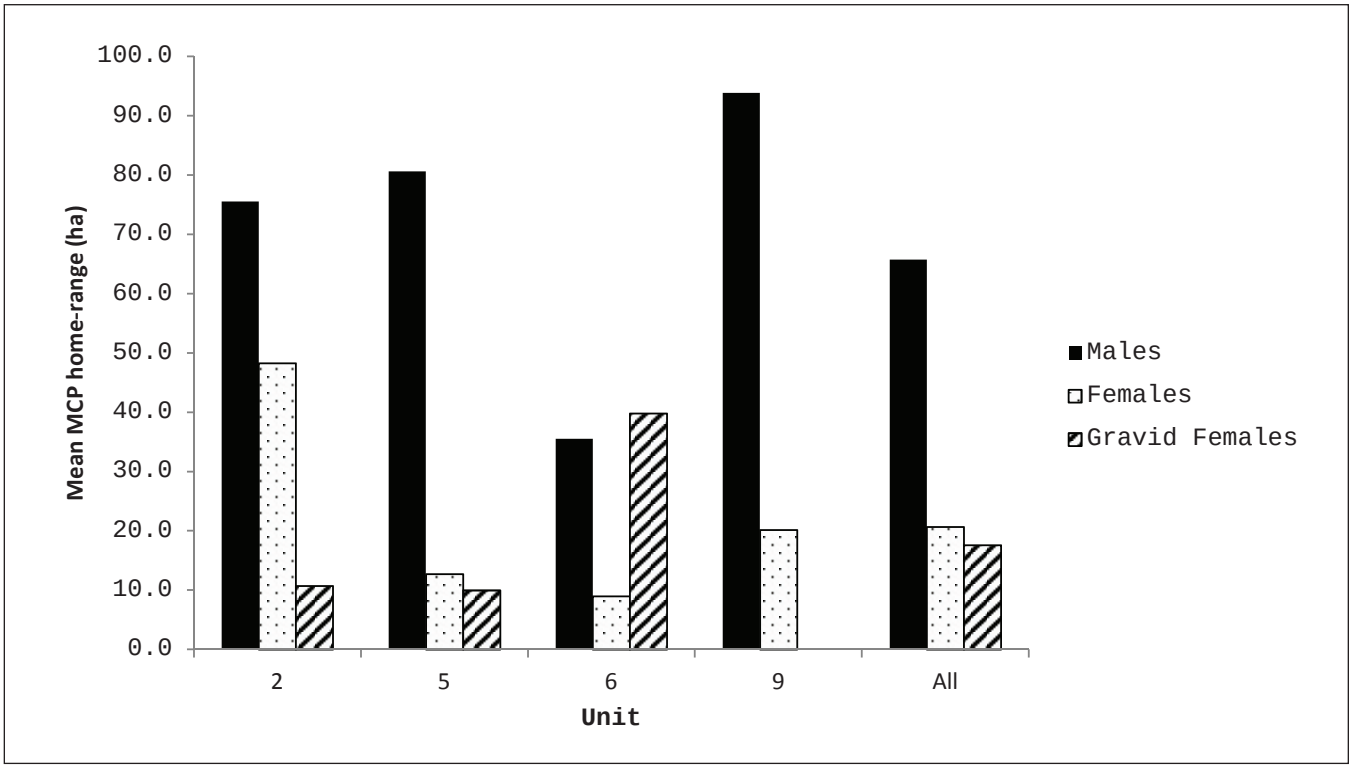


Figure 1.—Mean home-range sizes (95-percent Minimum Convex Polygon) pooled across years for male, non-gravid female, and gravid female timber rattlesnakes in HEE management units (2, 5, 6, and 9) during the pre-harvest period (2007 and 2008).

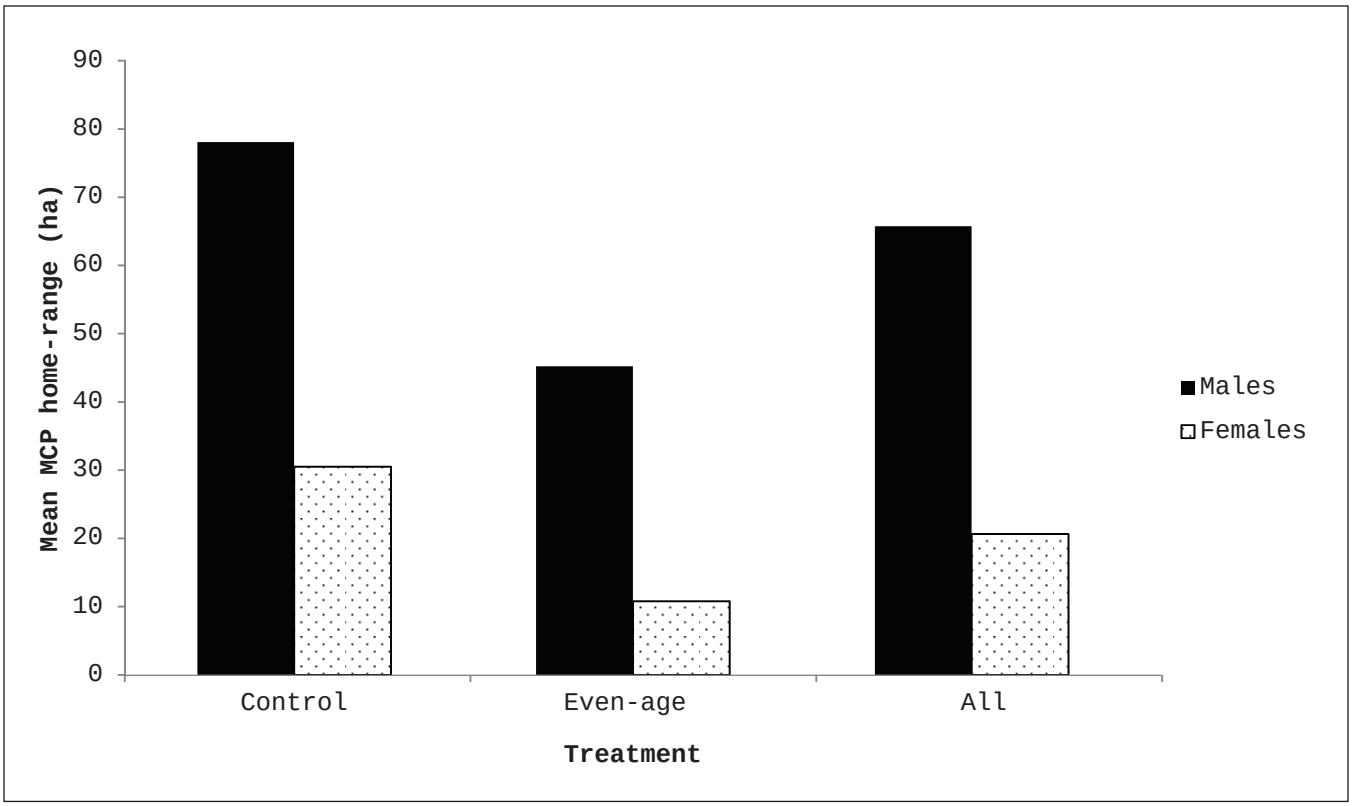


Figure 2.—Mean home-range sizes (95-percent Minimum Convex Polygon) pooled across years for male and female timber rattlesnakes in control (2 and 5) and even-age (6 and 9) HEE management units during the pre-harvest period (2007 and 2008).

respect to forest management (no timber is harvested except for removal of hazards and maintenance of vistas) and deer management. Before regular deer herd-reduction hunts began in 1993, deer at Brown County State Park had not been subjected to a legal harvest for decades and had an estimated prehunt density of 21.7 deer/km^2 (Swihart et al. 1998). The relatively high deer density resulted in reduced plant heights and densities that varied from neighboring Yellowwood State Forest (Webster 1997). It is not clear whether differing deer and/or timber management regimes or other factors produced the wide difference in rattlesnake home-range size between the studies.

Methods of harvesting that create early successional habitat and woody debris, which are both considered beneficial to timber rattlesnakes (Rittenhouse et al. 2007), could enhance habitat for timber rattlesnakes. Unmerchantable logs and tree tops were left in the HEE harvest areas and could provide foraging habitat for rattlesnakes as suggested by Rittenhouse et al. (2007). Urban and Swihart (this publication) studied the small mammal communities on the HEE as the basis for quantifying small mammal response to timber harvests. Future work could explicitly test timber rattlesnake responses to harvest in terms of predicted small mammal abundances rather than habitat structure alone.

Reinert et al. (2011) found that changes in habitat structure caused by logging in Pennsylvania did not impact timber rattlesnake use, at least in the short term. Habitat structure of sites they measured were more variable immediately after logging and tended to have more fallen log cover and decreased surface vegetation. Reinert et al. (2011) also found that mortality (intentional and accidental) was primarily a function of the logging operation itself rather than changes in habitat structure. They suggested that educational programs and policies restricting the

intentional killing of rattlesnakes may further reduce negative impacts associated with harvests and harvest activities.

Management strategies to reduce mortality may also consider minimizing logger encounters with timber rattlesnakes by adjusting the timing and proximity of harvests to timber rattlesnake den sites. Timber harvesting conducted in proximity to den sites, especially during ingress or egress when snakes are more concentrated near dens, would likely increase chance encounters. However, adjusting the timing of harvest outside of egress and ingress may not reduce vulnerability of gravid females, which tend to stay closer to the dens throughout the active season (Brown 1991, Martin 1993) and bask in open areas (Gibson et al. 2008, Walker 2000). The loss of reproductive females could have a disproportionate impact on population sustainability given their age to sexual maturity and infrequent reproduction (Brown 1991, Martin 1993).

In our study, gravid females had a slightly smaller home-range size than non-gravid females on average. The home-range size of female #161 decreased 74 percent from 2007 (non-gravid) to 2008 (gravid), whereas female #401 demonstrated the opposite trend (Table 1). It remains unclear how the movement behavior of gravid females could inform timber harvest decisions, but the areas around den sites should be a priority for future work.

We had no a priori knowledge of rattlesnake den locations on the HEE sites. The 23 snakes we tracked during the pre-harvest period returned to 13 different den sites, none of which was located within a planned harvest boundary. As the cutting cycles on the HEE progress, we will investigate potential impacts as a result of timber harvesting and the associated activities relative to proximity to den sites, based on home-range size and shift, linear movement parameters, and habitat use.

Although we can make some predictions of how timber rattlesnakes will respond to even- and uneven-age management based on prior studies and their biological needs, the HEE will help us quantitatively assess responses. With these data, we will be able to make management recommendations for timber rattlesnakes that may also benefit other forest reptiles (see Currylow et al., this publication). For example, many species of snakes (eastern hog-nosed snake [*Heterodon platirhinos*], milk snake [*Lampropeltis triangulum*], rough green snake [*Opheodrys aestivus*], gray rat snake [*Pantherophis spiloides*]) and lizards (five-lined skink [*Plestiodon fasciatus*], broad-headed skink [*P. laticeps*], eastern fence lizard [*Sceloporus undulates*]) were observed when we tracked timber rattlesnakes for our study. Some of these species potentially could be impacted by management techniques (Ross et al. 2000) that result in canopy openings and edges and increased amount of down woody debris. Current management strategies for reptiles and amphibians are based on the scientific information available but lack species-specific recommendations for mammals and birds (e.g., Kingsbury and Gibson 2002). With information gained from the HEE, we should be able to inform forest management decisions that are more inclusive of the specific needs of timber rattlesnakes and the other forest reptiles that will help maintain the integrity of forest ecosystems in the Central Hardwoods Region.

This paper summarizes pre-harvest timber rattlesnake home ranges. Future work will provide a more complete picture of timber rattlesnake ecology and the impacts of timber harvesting. First, we will determine the immediate effects of timber harvests on timber rattlesnake spatial ecology and habitat use by comparing home-range size and shift, movement parameters, and vegetation and habitat characteristics between treatment types pre- and post-harvest. Timber rattlesnakes on the HEE sites spend more than half their lives hibernating. We will also evaluate the structural and thermal characteristics of den sites.

These data will be compared with available sites in harvest openings and the forest matrix and can also serve as a baseline level with which to compare as the habitat structure changes with future harvests. Finally, we will estimate timber rattlesnake adult annual survivorship during the active and hibernal seasons and determine the genetic structure and variability of timber rattlesnakes on the HEE. These studies will provide data useful for evaluating the long-term population viability of timber rattlesnakes in managed forests.

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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.

TARGETED WILDLIFE AND PLANT COMMUNITIES

PRE-TREATMENT ANALYSIS OF WOODY VEGETATION COMPOSITION AND STRUCTURE ON THE HARDWOOD ECOSYSTEM EXPERIMENT RESEARCH UNITS

Michael R. Saunders and Justin E. Arseneault[1]

Abstract.—In long-term, large-scale forest management studies, documentation of pre-treatment differences among and variability within experimental units is critical for drawing the proper inferences from imposed treatments. We compared pre-treatment overstory and large shrub communities (diameters at breast height >1.5 cm) for the 9 research cores with the Hardwood Ecosystem Experiment (HEE) in south-central Indiana with a total of 683 0.1-ha sampling plots. There were many significant differences ($p < 0.05$) in stand structure among the cores for many attributes, but intra-core variability was not severe enough to lead to significant differences in structural attributes for the proposed management treatments (that is, when data were averaged across three replicates in each management treatment, there were no significant differences). Ordinations of plot-level data showed that all cores occupied much of the same ordination space, suggesting that forest structure was highly variable within cores but did not differ as much between cores. Plot-level maps of species composition and size structure also showed large degrees of variability within each research core and hinted at spatial patterning not yet captured well by HEE inventories. We suggest that future projects and subsequent analyses of the HEE better capture intra-stand variability in both abiotic and biotic components through spatially explicit methodologies.

INTRODUCTION

Before the early 1990s, much of the emphasis for management of public lands across the United States was on timber production (Curtis et al. 2007, Seymour et al. 2006). Silvicultural research questions often were concerned with growth and yield predictions, regeneration responses, thinning practices, and other stand-level responses (Puettmann et al. 2009). Researchers often used either small, very uniform plots—often 0.10 ha or less—in their designs to isolate the treatment responses (e.g., Cochran and Barrett 1999, D'Amato et al. 2011), or larger, unreplicated (or poorly replicated) areas to demonstrate harvest practices (e.g., Kolbe and McKay 1939, Sánchez Meador and Moore 2010, Sendak et al. 2003). Consequently, these plots and areas, even if maintained over long time periods, had limited use for analysis of broader environmental questions outside the scope of the original study (Curtis and DeBell 2010, Puettmann et al. 2009).

Contemporary silvicultural research has shifted its focus much more to balancing timber production objectives with wildlife, water, biodiversity, social, and a multitude of other objectives (Long 2009, Seymour et al. 2006). This change in focus has resulted in a shift to much larger experimental units, often on the order of 10 ha, that incorporate heterogeneous areas (Curtis and DeBell 2010, Seymour et al. 2006). Larger-scale experimentation allows linkages of forest structural

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response from treatment to wildlife habitat use and movement, nutrient cycling, natural disturbances, and other ecological processes that vary strongly temporally and spatially. Given the variability of the experimental units, however, these experiments are often poorly replicated because they are expensive to monitor and maintain (Curtis and DeBell 2010). Problems in documenting treatment responses may result as statistical power is often low, particularly if researchers and reviewers subscribe to the agricultural research model that summarizes treatment effects with a single mean value, thus ignoring intra-experiment unit variability (Puettmann et al. 2009).

Mixed-effects models, likelihood methods, spatial statistics, Bayesian approaches, and other tools can be used to help overcome some of these issues by explicitly incorporating variability into the statistical analysis. Regardless of the approach taken, documentation of pre-treatment differences in forest structure and composition among experimental units is essential to understanding how implemented treatments influence forest development over time (Ganio and Puettmann 2008, Kabrick et al. 1997). This documentation is particularly important for collaborative studies where comparative, pre-treatment analyses can inform subsequent wildlife and other ecological studies on potential additional needs for data collection.

The Hardwood Ecosystem Experiment (HEE) is such an example of a contemporary, large-scale, forest management study. This experiment, begun in 2006, is documenting the long-term ecological and social impacts of forest management on public lands in southern Indiana. The HEE experiment is in its infancy as pre-harvest data collection has been completed and early post-harvest data collection to quantify plant and animal responses to even- and uneven-aged management is ongoing.

During study site selection for HEE, researchers used blocks of management units that were thought to be rather uniform as a whole, but likely were variable in age, composition, and structure at finer scales (Kalb and Mycroft, this publication). Therefore, we approached the goal of documenting pre-treatment differences among units in two ways. First, we asked: In a spatially explicit context, are there differences in tree and large shrub community composition and structure among the HEE management units (i.e., research cores) that could mask overall treatment effects? Second, we asked: In a spatially implicit context, are there strong differences in the plot-level variability of the tree and large shrub communities among the cores?

STUDY AREA

The HEE study consists of nine management units throughout Morgan-Monroe and Yellowwood State Forests in south-central Indiana (Kalb and Mycroft, this publication: Fig. 1). Management units range from 303 to 483 ha (Kalb and Mycroft, this publication: Table 1), with each unit consisting of a buffer area and a research core. The buffer area around each research core is managed consistently across units using uneven-aged management, a typical management strategy used by the Indiana Department of Natural Resources-Division of Forestry at this time (Haulton, this publication). Research cores range in size from 78 to 110 ha (Kalb and Mycroft, this publication: Table 1) and are randomly assigned to one of three management regimes: an experimental control, even-aged management using three-stage shelterwood and clearcut approaches, and uneven-aged management using single-tree selection and patch cutting. There are, therefore, three replicates of each treatment, blocked roughly north to south across the two State Forests (Kalb and Mycroft, this publication: Fig. 1). Research cores contain most of the detailed inventory systems for the experiment.

MATERIALS AND METHODS

Sampling Design

From January 2008 to August 2010, 683 sampling plots were established on 75m x 150m grids within each of the nine HEE research cores. Sampling plots are circular and 0.1 ha in size. Sampling intensity ranges from 67 to 89 plots per unit, with an average area-based intensity of 8.5 percent (range: 8.0-9.2 percent). Plots are scheduled to be sampled again 4 years post-harvest (2013) and every 4 years thereafter.

Within each sampling plot, all overstory trees with diameters at breast height (d.b.h.; 1.37 m from ground) >11.5 cm are measured. Saplings, defined as all woody stems between 1.5 and 11.5 cm d.b.h., are sampled with a 0.025-ha circular subplot centered within the overstory sampling plot. For all overstory trees and saplings, species, d.b.h., condition (e.g., live, dead, cull, broken top, dying), light field (sensu Bechtold 2003), stratum, and spatial location (i.e., azimuth and distance from plot center) are recorded. Total tree height and height of the lowest live crown are subsampled on approximately four overstory trees and four saplings per sampling plot. These individuals are systematically selected as the first observed tree and sapling in each quarter of the circular plot as crews move clockwise from due north. If the individual is dead, species, d.b.h., total height, and decay class (Cline et al. 1980) are recorded.

Data Analysis

Using the data collected from the pre-treatment inventory, we tested for differences in several structural attributes among research cores using analysis of variance (ANOVA). These attributes were as follows: 1) number of species per plot, 2) density of stems per ha, 3) quadratic mean tree diameter, and 4) basal area per ha. Individual sampling plots were the experimental units for these comparisons. Analyses were conducted for only overstory trees with d.b.h. ≥30.0 cm, and for all overstory trees and saplings combined (i.e., all stems with d.b.h. ≥1.5 cm). We also tested for differences in these attributes for several key species groups (Table 1).

Table 1.—Scientific and common names of overstory trees and saplings recorded on the Hardwood Ecosystem Experiment sites. Species groups were defined based on their prevalence on the sites and the shade tolerance scale of Niinemets and Valladares (2006) with 1 = very intolerant, 2 = intolerant, 3 = moderately tolerant, 4 = tolerant, and 5 = very tolerant.

Scientific name	Common name	Shade tolerance	Species group
Acer negundo	boxelder	3.5	Tolerant
Acer nigrum	black maple	3.0	Tolerant
Acer rubrum	red maple	3.4	Tolerant
Acer saccharum	sugar maple	4.8	Tolerant
Alnus glutinosa	European black alder	2.7	Noncommercial
Amelanchier spp.	serviceberry	4.3	Noncommercial
Asimina triloba	paw paw	4.0	Noncommercial
Carpinus caroliniana	American hornbeam	4.6	Noncommercial
Carya tomentosa	mockernut hickory	2.4†	Oak & Hickory
Carya cordiformis	bitternut hickory	2.1	Oak & Hickory
Carya glabra	pignut hickory	2.7	Oak & Hickory
Carya ovata	shagbark hickory	3.4	Oak & Hickory
Celtis occidentalis	hackberry	3.2	Other Commer.
Cercis canadensis	eastern redbud	3.0	Noncommercial
Cornus alternifolia	alternate-leaf dogwood	4.0	Noncommercial
Cornus florida	flowering dogwood	4.9	Noncommercial
Cornus stolonifera	red-osier dogwood	3.8†	Noncommercial
Corylus americana	American hazelnut	3.5	Noncommercial

(Table 1 continued on next page)

Table 1 (continued).—Scientific and common names of overstory trees and saplings recorded on the Hardwood Ecosystem Experiment sites. Species groups were defined based on their prevalence on the sites and the shade tolerance scale of Niinemets and Valladares (2006) with 1 = very intolerant, 2 = intolerant, 3 = moderately tolerant, 4 = tolerant, and 5 = very tolerant.

Scientific name	Common name	Shade tolerance	Species group
Crataegus spp.	hawthorn	2.3†	Noncommercial
Diervilla lonicera	bush honeysuckle		Noncommercial
Elaeagnus umbellata	autumn olive	1.4†	Noncommercial
Fagus grandifolia	American beech	4.8	Tolerant
Fraxinus americana	white ash	2.5	Other Commer.
Fraxinus nigra	black ash	3.0	Other Commer.
Fraxinus pennsylvanica	green ash	3.1	Other Commer.
Gymnocladus dioicus	Kentucky coffee-tree	2.5	Other Commer.
Hamamelis virginiana	witch-hazel	3.0	Noncommercial
Juglans cinerea	butternut	1.9	Intolerant
Juglans nigra	black walnut	1.9	Intolerant
Juniperus virginiana	eastern red-cedar	1.3	Other Commer.
Lindera benzoin	spicebush	3.0	Noncommercial
Liriodendron tulipifera	tulip poplar	2.1	Intolerant
Lonicera maackii	amur honeysuckle	3.6†	Noncommercial
Maclura pomifera	osage orange	1.5	Noncommercial
Morus alba	white mulberry	1.4	Other Commer.
Nyssa sylvatica	blackgum	3.5	Other Commer.
Ostrya virginiana	eastern hophornbeam	4.6	Noncommercial
Pinus banksiana	jack pine	1.4	Other Commer.
Pinus resinosa	red pine	1.9	Other Commer.
Pinus strobus	eastern white pine	3.2	Other Commer.
Pinus virginiana	Virginia pine	2.0	Other Commer.
Platanus occidentalis	American sycamore	2.9	Other Commer.
Populus deltoides	eastern cottonwood	1.8	Other Commer.
Populus grandidentata	bigtooth aspen	1.2	Other Commer.
Populus tremuloides	quaking aspen	1.2	Other Commer.
Prunus americana	American plum	2.2†	Noncommercial
Prunus serotina	black cherry	2.5	Intolerant
Quercus alba	white oak	2.9	Oak & Hickory
Quercus coccinea	scarlet oak	2.1	Oak & Hickory
Quercus macrocarpa	bur oak	2.7	Oak & Hickory
Quercus muehlenbergii	chinkapin oak	2.2	Oak & Hickory
Quercus palustris	pin oak	2.5	Oak & Hickory
Quercus prinus	chestnut oak	2.9	Oak & Hickory
Quercus rubra	northern red oak	2.8	Oak & Hickory
Quercus velutina	black oak	2.7	Oak & Hickory
Rhus spp.	sumac	1.5	Noncommercial
Robinia pseudoacacia	black locust	1.7	Other Commer.
Sassafras albidum	sassafras	1.7	Intolerant
Staphylea trifolia	bladdernut		Noncommercial
Tilia americana	American basswood	4.0	Tolerant
Ulmus americana	American elm	3.1	Other Commer.
Ulmus rubra	red elm	3.3	Other Commer.
Viburnum prunifolium	smooth blackhaw	3.3†	Noncommercial
Viburnum recognitum	northern arrowwood	3.3†	Noncommercial
Vitis spp.	grape		Noncommercial

† Value for species was not given. Value is estimated from average of all listed North American species in that genus.

To evaluate differences among treatment units prior to harvesting, we used ANOVA with the core-level estimates (i.e., research cores were the experimental unit). For each structural attribute, we fit the fixed effects model:

$$Y_i = \mu + \alpha_i + \varepsilon_i , \quad [1]$$

where μ is the overall mean of the response, α_i is the treatment effect (df = 2) on the response, and ε_i is the error (df = 6) independently distributed on $N(0,\sigma^2)$. Since variances were not equal for each core-level estimate, we weighted the ANOVA by σ_{core}^{-2}. A significance level of alpha = 0.05 was used for all former analyses.

Non-metric multidimensional scaling (NMDS) was used to graphically summarize pretreatment, plot-level differences in species importance values (IV) among both plots and the research core estimates. For the plot-level analysis, IV was calculated for each species as the percentage of total basal area in a sampling plot plus the percentage of total stem density in a sampling plot; therefore IVs could range from zero to 200. For the core-level analysis, IV includes percentage basal area, percentage stem density, and percentage of sampling plots where a species was recorded within the research core; therefore, IVs could range from zero to 300. To simplify the ordination, rare and isolated species (i.e., those found in only one research core or in <10 plots) were grouped for analysis. Both ordinations were run 250 times on both the real and randomized data using Sorensen distance measurements, random starting coordinates, and a stability criterion of 0.00001.

Lastly, we plotted plot-level, spatial summaries of basal area proportion by each species group using a geographic information system (ArcMap 10, ESRI, Redlands, CA) to evaluate within-research core variation in species composition. A similar analysis was conducted to assess within-research core variation in size class distributions after stratifying plot-level data into four, timber-based size classes: sapling (0–12.4 cm d.b.h.), pole (12.5–30.5 cm d.b.h.), small sawtimber (30.6–45.6 cm d.b.h.), and large sawtimber (≥45.7 cm d.b.h.).

Raw data were processed and summarized using the plyr package (version 1.4; Wickham 2011) in R 2.12 (R Development Core Team 2010). PC-ORD 5.0 (McCune and Grace 2002, McCune and Mefford 2006) was used for the NMDS analysis. Figures were produced using R 2.12 and the ggplot2 package (version 0.8.9; Wickham 2009).

RESULTS

Core-Level Differences

There were some differences in structure among the research cores. Total stem density averaged from 923 to 1,527 trees ha^{-1}, and overstory stem density (i.e., d.b.h. ≥30.0 cm) averaged from 93 to 150 trees ha^{-1} (Table 2; Fig. 1). Total basal area averaged from 21.7 to 29.9 m^2 ha^{-1}, with overstory trees making up 15.5 to 22.5 m^2 ha^{-1} (Table 2; Fig. 2). Although the cores were significantly different for many attributes (Appendix 1), differences were not pronounced aside from a few exceptions (Table 2). For example, Research Core 7 had significantly lower total and overstory stem density, and total and overstory basal area, than most other cores. Research Cores 3 and 8 had the highest total stem density and total basal area, respectively.

Diameter (Fig. 3) and basal area (Fig. 4) distributions were similar among most cores. Core 7 was notably different in having a less peaked basal area and fewer small sawlog-sized (i.e., 30-50 cm d.b.h.) trees than other research cores. Although all cores had negative exponential diameter distributions that suggested a balanced uneven-aged structure (Fig. 3), the bell-shaped basal area distributions suggested that the age structures were strongly irregular (Fig. 4), likely comprising an older oak-dominated cohort with multiple younger cohorts of differing species (Figs. 1 and 2).

Table 2.—Mean ± standard error and range of observations by research core for selected plot-level attributes. Density, quadratic mean diameter (QMD) and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥ 30 cm d.b.h.).

Attribute	Research core								
	1	2	3	4[†]	5	6	7	8	9
All Species									
No. of species per plot	9.5 ± 0.2 (5,16)	11.1 ± 0.3 (6,18)	11.6 ± 0.3 (5,19)	9.9 ± 0.3 (6,15)	10.3 ± 0.2 (6,15)	10.9 ± 0.3 (5,18)	8.4 ± 0.3 (4,17)	10.6 ± 0.3 (5,17)	9.8 ± 0.3 (4,17)
Tot. density (trees ha^{-1})	932 ± 36 (380,1780)	1420 ± 111 (470,4560)	1527 ± 90 (350,5140)	1156 ± 62 (420,2240)	1435 ± 63 (500,2860)	1161 ± 59 (390,3000)	923 ± 56 (210,3260)	1441 ± 68 (560,3030)	962 ± 53 (180,2830)
Ovs. density (trees ha^{-1})	140 ± 5 (60,240)	116 ± 6 (0,260)	103 ± 5 (10,250)	150 ± 6 (50,280)	113 ± 5 (40,260)	120 ± 5 (0,210)	93 ± 4 (10,180)	136 ± 6 (50,290)	121 ± 6 (0,290)
Tot. QMD (cm)	20.6 ± 0.5 (11.7,34.6)	17.7 ± 0.7 (6.9,29.3)	15.6 ± 0.5 (6.2,28.3)	19.0 ± 0.6 (11.4,31.0)	16.0 ± 0.4 (8.9,26.1)	17.7 ± 0.5 (8.4,29.0)	18.8 ± 0.6 (6.3,40.3)	17.0 ± 0.5 (10.6,31.6)	19.4 ± 0.7 (5.8,40.0)
Ovs. QMD (cm)[‡]	44.1 ± 0.5 (34.7,54.5)	45.6 ± 0.5 (36.5,62.3)	45.4 ± 0.5 (31.5,56.8)	44.2 ± 0.7 (34.8,58.0)	45.9 ± 0.6 (36.3,58.0)	43.5 ± 0.6 (43.5,57.7)	45.9 ± 0.6 (31.4,59.2)	44.6 ± 0.6 (32.6,57.7)	43.9 ± 0.6 (33.0,63.4)
Tot. Basal area (m^2 ha^{-1})	28.8 ± 0.7 (16.6,50.6)	27.8 ± 0.9 (13,42.5)	24.4 ± 0.7 (13.7,46.0)	29.6 ± 0.8 (17.8,44.7)	26.3 ± 0.7 (14.6,43.6)	25.4 ± 0.7 (12.6,38.5)	21.7 ± 0.7 (3.6,49.2)	29.9 ± 1.1 (10.9,56.1)	24.6 ± 0.8 (7.5,48.4)
Ovs. Basal area (m^2 ha^{-1})	21.1 ± 0.7 (6.9,44.9)	19.0 ± 1.1 (0,37.0)	16.9 ± 0.8 (0.8,37.5)	22.5 ± 0.8 (10.2,37.5)	18.5 ± 0.8 (5.4,35.4)	17.6 ± 0.7 (0,32.3)	15.5 ± 0.7 (2.3,33.8)	21.3 ± 1.1 (6.6,53.5)	18.4 ± 0.8 (0,39.6)
Oak & Hickory									
No. of species per plot	3.8 ± 0.2 (0,6)	3.6 ± 0.2 (0,6)	3.5 ± 0.2 (0,7)	3.6 ± 0.2 (0,6)	3.9 ± 0.1 (1,7)	4.2 ± 0.1 (1,8)	2.6 ± 0.1 (0,5)	3.2 ± 0.2 (0,6)	3.8 ± 0.2 (0,7)
Tot. density (trees ha^{-1})	235 ± 15 (0,680)	205 ± 29 (0,1440)	166 ± 17 (0,730)	204 ± 15 (0,530)	192 ± 15 (10,670)	219 ± 19 (10,1160)	85 ± 8 (0,430)	163 ± 13 (0,430)	186 ± 15 (0,500)
Ovs. density (trees ha^{-1})	120 ± 7 (0,230)	81 ± 8 (0,260)	63 ± 5 (0,170)	123 ± 8 (0,240)	92 ± 5 (10,220)	92 ± 6 (0,200)	48 ± 4 (0,150)	77 ± 6 (0,180)	105 ± 7 (0,270)
Tot. QMD (cm)	33.6 ± 1.1 (0,55.6)	34.8 ± 1.7 (0,64.3)	32.8 ± 1.5 (0,60.3)	37.2 ± 1.1 (0,53.8)	36.2 ± 0.9 (18.6,53.6)	33.8 ± 0.9 (10.7,61.3)	37.7 ± 1.5 (0,66.4)	32.2 ± 1.6 (0,51.1)	37.0 ± 1.3 (0,61.7)
Ovs. QMD (cm)[‡]	44.7 ± 0.6 (34.7,56.5)	46.3 ± 0.9 (33.0,64.3)	46.6 ± 0.7 (31.5,62.6)	44.8 ± 0.8 (34.6,61.1)	46.4 ± 0.7 (34.3,69.7)	43.9 ± 0.7 (33.3,65.0)	48.2 ± 0.9 (31.5,66.4)	45.6 ± 0.6 (34.7,57.2)	45.3 ± 0.8 (33.1,67.4)
Tot. Basal area (m^2 ha^{-1})	21.5 ± 1.0 (0,43.8)	15.7 ± 1.3 (0,39.8)	11.9 ± 0.9 (0,40.1)	21.7 ± 1.3 (0,37.9)	17.4 ± 0.9 (1.0,34.0)	16.8 ± 0.9 (0.3,34.0)	9.6 ± 0.7 (0,24.7)	14.4 ± 1.1 (0,37.6)	18.2 ± 1.2 (0,40.0)
Ovs. Basal area (m^2 ha^{-1})	18.3 ± 1.0 (0,41.3)	13.6 ± 1.2 (0,37.0)	10.6 ± 0.8 (0,36.3)	19.3 ± 1.2 (0,35.5)	15.3 ± 0.9 (0,32.0)	13.6 ± 0.8 (0,27.7)	8.6 ± 0.7 (0,22.8)	12.5 ± 1.0 (0,36.1)	16.0 ± 1.0 (0,37.3)

(Table 2 continued on next page)

Table 2 (continued).—Mean ± standard error and range of observations by research core for selected plot-level attributes. Density, quadratic mean diameter (QMD) and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥ 30 cm d.b.h.).

Attribute	Research core								
	1	2	3	4[†]	5	6	7	8	9
Tolerant									
No. of species per plot	1.9 ± 0.1 (0,4)	2.6 ± 0.1 (0,4)	2.6 ± 0.1 (1,4)	2.5 ± 0.1 (1,4)	2.6 ± 0.1 (1,4)	2.2 ± 0.1 (0,4)	2.1 ± 0.1 (0,3)	2.7 ± 0.1 (1,3)	2.3 ± 0.1 (1,4)
Tot. density (trees ha^{-1})	297 ± 22 (0,920)	607 ± 45 (0,1870)	556 ± 33 (20,1350)	583 ± 53 (80,1560)	736 ± 47 (10,1950)	469 ± 42 (0,1630)	529 ± 34 (0,1600)	784 ± 45 (110,1950)	394 ± 36 (10,1470)
Ovs. density (trees ha^{-1})	7 ± 1 (0,60)	12 ± 2 (0,80)	14 ± 2 (0,60)	5 ± 2 (0,80)	8 ± 1 (0,50)	7 ± 2 (0,60)	22 ± 2 (0,80)	14 ± 2 (0,90)	7 ± 2 (0,40)
Tot. QMD (cm)	10.3 ± 0.5 (0,23.1)	10.8 ± 0.6 (0,26.9)	10.7 ± 0.5 (0.2,30.2)	8.6 ± 0.5 (0.2,18.1)	9.6 ± 0.5 (5.3,26.6)	9.4 ± 0.5 (0,22.0)	13.3 ± 0.6 (0,33.7)	9.2 ± 0.5 (3.1,19.6)	10.9 ± 0.7 (4.3,31.4)
Ovs. QMD (cm)[‡]	40.9 ± 2.5 (30.5,82.3)	39.7 ± 1.4 (30.9,69.5)	40.0 ± 0.9 (30.5,53.5)	40.3 ± 2.5 (30.4,58.0)	43.2 ± 2.7 (30.1,101.1)	39.2 ± 1.2 (30.0,53.9)	40.7 ± 1.0 (30.2,72.1)	40.7 ± 1.3 (31.0,61.0)	42.7 ± 1.8 (32.4,65.0)
Tot. Basal area (m^2 ha^{-1})	2.9 ± 0.3 (0,12.9)	5.3 ± 0.4 (0,16.3)	4.7 ± 0.3 (0.2,14.4)	3.2 ± 0.4 (0.2,16.5)	4.7 ± 0.3 (0.2,16.5)	3.4 ± 0.4 (0,15.1)	6.5 ± 0.5 (0,20.7)	5.4 ± 0.5 (0.2,21.9)	3.1 ± 0.3 (0.1,12.8)
Ovs. Basal area (m^2 ha^{-1})	0.9 ± 0.2 (0,8.3)	1.6 ± 0.3 (0,11.3)	1.8 ± 0.3 (0,9.9)	0.6 ± 0.2 (0,9.7)	1.3 ± 0.3 (0,10.3)	1.0 ± 0.3 (0,11.4)	3.1 ± 0.4 (0,15.2)	2.0 ± 0.4 (0,17.3)	1.0 ± 0.2 (0,8.3)
Intolerant									
No. of species per plot	1.4 ± 0.1 (0,4)	1.7 ± 0.1 (0,4)	1.7 ± 0.1 (0,3)	1.2 ± 0.1 (0,3)	1.2 ± 0.1 (0,3)	1.3 ± 0.1 (0,3)	1.2 ± 0.1 (0,3)	1.1 ± 0.1 (0,4)	1.1 ± 0.1 (0,3)
Tot. density (trees ha^{-1})	161 ± 17 (0,700)	206 ± 30 (0,1240)	354 ± 48 (0,2010)	125 ± 21 (0,730)	208 ± 47 (0,1890)	141 ± 27 (0,1440)	88 ± 14 (0,810)	51 ± 12 (0,650)	132 ± 22 (0,720)
Ovs. density (trees ha^{-1})	8 ± 2 (0,110)	15 ± 2 (0,70)	18 ± 3 (0,160)	8 ± 3 (0,90)	5 ± 1 (0,70)	11 ± 2 (0,100)	13 ± 2 (0,90)	5 ± 1 (0,40)	4 ± 1 (0,40)
Tot. QMD (cm)	12.7 ± 1.1 (0,42.9)	20.8 ± 2.0 (0,74.2)	14.8 ± 1.5 (0,53.1)	12.5 ± 2.1 (0,55.1)	13.4 ± 1.6 (0,54.2)	17.6 ± 1.9 (0,62.3)	20.1 ± 1.8 (0,60.8)	12.3 ± 1.8 (0,49.8)	9.9 ± 1.3 (0,45.2)
Ovs. QMD (cm)[‡]	44.0 ± 2.0 (32.8,70.9)	48.0 ± 1.9 (31.8,74.2)	45.8 ± 1.7 (31.2,65.9)	45.7 ± 2.6 (34.8,61.7)	45.1 ± 2.1 (30.9,70.0)	46.8 ± 2.0 (32.1,83.8)	46.5 ± 1.5 (31.8,71.4)	44.5 ± 1.9 (32.0,69.9)	39.2 ± 1.9 (30.0,51.6)
Tot. Basal area (m^2 ha^{-1})	2.5 ± 0.5 (0,22.1)	4.5 ± 0.6 (0,16.7)	4.9 ± 0.7 (0,29.7)	2.1 ± 0.6 (0,19.6)	2.2 ± 0.4 (0,13.8)	2.8 ± 0.5 (0,22.2)	3.5 ± 0.6 (0,20.5)	1.2 ± 0.2 (0,6.6)	1.7 ± 0.3 (0,13.1)
Ovs. Basal area (m^2 ha^{-1})	1.2 ± 0.3 (0,16.2)	2.8 ± 0.5 (0,14.4)	2.9 ± 0.6 (0,24.0)	1.3 ± 0.4 (0,18.0)	0.9 ± 0.2 (0,9.0)	2.0 ± 0.4 (0,18.8)	2.5 ± 0.5 (0,18.1)	0.8 ± 0.2 (0,5.7)	0.6 ± 0.2 (0,8.4)

(Table 2 continued on next page)

Table 2 (continued).—Mean ± standard error and range of observations by research core for selected plot-level attributes. Density, quadratic mean diameter (QMD) and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥ 30 cm d.b.h.).

Attribute	Research core								
	1	2	3	4[†]	5	6	7	8	9
Other Commercial									
No. of species per plot	0.9 ± 0.1 (0,3)	1.3 ± 0.1 (0,5)	1.6 ± 0.1 (0,7)	1.2 ± 0.2 (0,5)	1.3 ± 0.1 (0,3)	1.5 ± 0.1 (0,5)	1.1 ± 0.1 (0,5)	1.6 ± 0.2 (0,6)	1.1 ± 0.1 (0,5)
Tot. density (trees ha^{-1})	95 ± 20 (0,920)	79 ± 14 (0,480)	137 ± 19 (0,850)	125 ± 21 (0,540)	106 ± 12 (0,460)	141 ± 21 (0,710)	45 ± 8 (0,440)	150 ± 28 (0,1290)	69 ± 15 (0,760)
Ovs. density (trees ha^{-1})	5 ± 1 (0,80)	7 ± 1 (0,50)	9 ± 2 (0,140)	13 ± 5 (0,170)	7 ± 3 (0,190)	9 ± 2 (0,70)	8 ± 3 (0,150)	39 ± 9 (0,290)	4 ± 1 (0,70)
Tot. QMD (cm)	8.9 ± 1.1 (0,44.4)	13.4 ± 1.7 (0,49.5)	12.5 ± 1.3 (0,81.3)	8.8 ± 1.2 (0,31.4)	14.1 ± 1.3 (0,59.0)	13.4 ± 1.2 (0,43.3)	12.3 ± 1.4 (0,46.5)	15.6 ± 1.7 (0,47.0)	12.3 ± 1.5 (0,51.6)
Ovs. QMD (cm)[‡]	42.8 ± 2.7 (30.5,74.7)	39.1 ± 1.3 (31.2,54.3)	44.8 ± 2.1 (31.2,81.3)	37.6 ± 1.9 (31.4,50.4)	41.4 ± 1.8 (30.0,59.0)	39.0 ± 1.1 (30.0,56.9)	39.6 ± 1.5 (31.8,56.6)	40.5 ± 1.4 (30.7,57.4)	42.7 ± 2.5 (30.2,55.7)
Tot. Basal area (m^2 ha^{-1})	1.2 ± 0.2 (0,10.7)	1.6 ± 0.4 (0,18.4)	2.2 ± 0.4 (0,25.3)	2.3 ± 0.7 (0,23.8)	1.7 ± 0.4 (0,30.4)	1.9 ± 0.3 (0,13.0)	1.7 ± 0.5 (0,31.1)	8.4 ± 1.9 (0,53.7)	1.3 ± 0.3 (0,15.0)
Ovs. Basal area (m^2 ha^{-1})	0.7 ± 0.2 (0,8.1)	0.9 ± 0.2 (0,8.3)	1.5 ± 0.4 (0,21.2)	1.3 ± 0.5 (0,17.9)	1.0 ± 0.4 (0,26.4)	1.1 ± 0.2 (0,10.1)	1.2 ± 0.4 (0,26.4)	6.0 ± 1.6 (0,53.5)	0.7 ± 0.2 (0,13.9)
Noncommercial									
No. of species per plot	1.5 ± 0.1 (0,5)	1.8 ± 0.2 (0,4)	2.2 ± 0.2 (0,6)	1.4 ± 0.2 (0,5)	1.4 ± 0.1 (0,4)	1.8 ± 0.1 (0,5)	1.4 ± 0.1 (0,6)	2.0 ± 0.2 (0,6)	1.5 ± 0.2 (0,7)
Tot. density (trees ha^{-1})	145 ± 18 (0,920)	324 ± 60 (0,2610)	314 ± 33 (0,1430)	119 ± 24 (0,920)	193 ± 34 (0,1480)	192 ± 21 (0,1100)	175 ± 40 (0,2920)	293 ± 32 (0,1120)	181 ± 40 (0,2080)
Ovs. density (trees ha^{-1})	1.2 ± 0.4 (0,10)	0.4 ± 0.3 (0,10)	0.4 ± 0.2 (0,10)	0.8 ± 0.4 (0,10)	0.1 ± 0.1 (0,10)	0.5 ± 0.3 (0,20)	1.1 ± 0.4 (0,20)	0.3 ± 0.2 (0,10)	0.3 ± 0.2 (0,10)
Tot. QMD (cm)	7.9 ± 0.8 (0,39.9)	6.7 ± 0.8 (0,38.5)	5.2 ± 0.3 (0,14.5)	4.3 ± 0.6 (0,21.3)	4.7 ± 0.4 (0,19.1)	5.9 ± 0.7 (0,43.9)	5.5 ± 0.7 (0,32.3)	4.2 ± 0.4 (0,21.9)	3.8 ± 0.3 (0,9.8)
Ovs. QMD (cm)[‡]	39.1 ± 2.1 (31.2,54.9)	44.6 ± 4.6 (38.5,53.6)	38.6 ± 2.9 (32.0,44.0)	36.1 ± 2.0 (33.5,42.0)	32.0 (32.0,32.0)	37.0 ± 3.6 (32.0,43.9)	40.0 ± 6.7 (30.7,80.0)	43.6 ± 0.4 (43.2,43.9)	34.1 ± 1.2 (32.8,35.3)
Tot. Basal area (m^2 ha^{-1})	0.68 ± 0.08 (0,39.9)	0.73 ± 0.11 (0,38.5)	0.76 ± 0.10 (0,14.5)	0.32 ± 0.06 (0,21.3)	0.35 ± 0.05 (0,19.1)	0.50 ± 0.06 (0,43.9)	0.48 ± 0.09 (0,32.3)	0.52 ± 0.07 (0,21.9)	0.30 ± 0.05 (0,9.8)
Ovs. Basal area (m^2 ha^{-1})	0.15 ± 0.05 (0,2.4)	0.07 ± 0.04 (0,2.3)	0.05 ± 0.03 (0,1.5)	0.08 ± 0.04 (0,1.4)	0.01 ± 0.01 (0,0.8)	0.05 ± 0.03 (0,1.9)	0.15 ± 0.07 (0,5.0)	0.04 ± 0.03 (0,1.5)	0.03 ± 0.02 (0,1.0)

[†] Data from 19 plots were discarded from this analysis, due to questionable plot-level basal areas far exceeding 50 m^2 ha^{-1}.

[‡] Plots without overstory trees were ignored for this attribute.

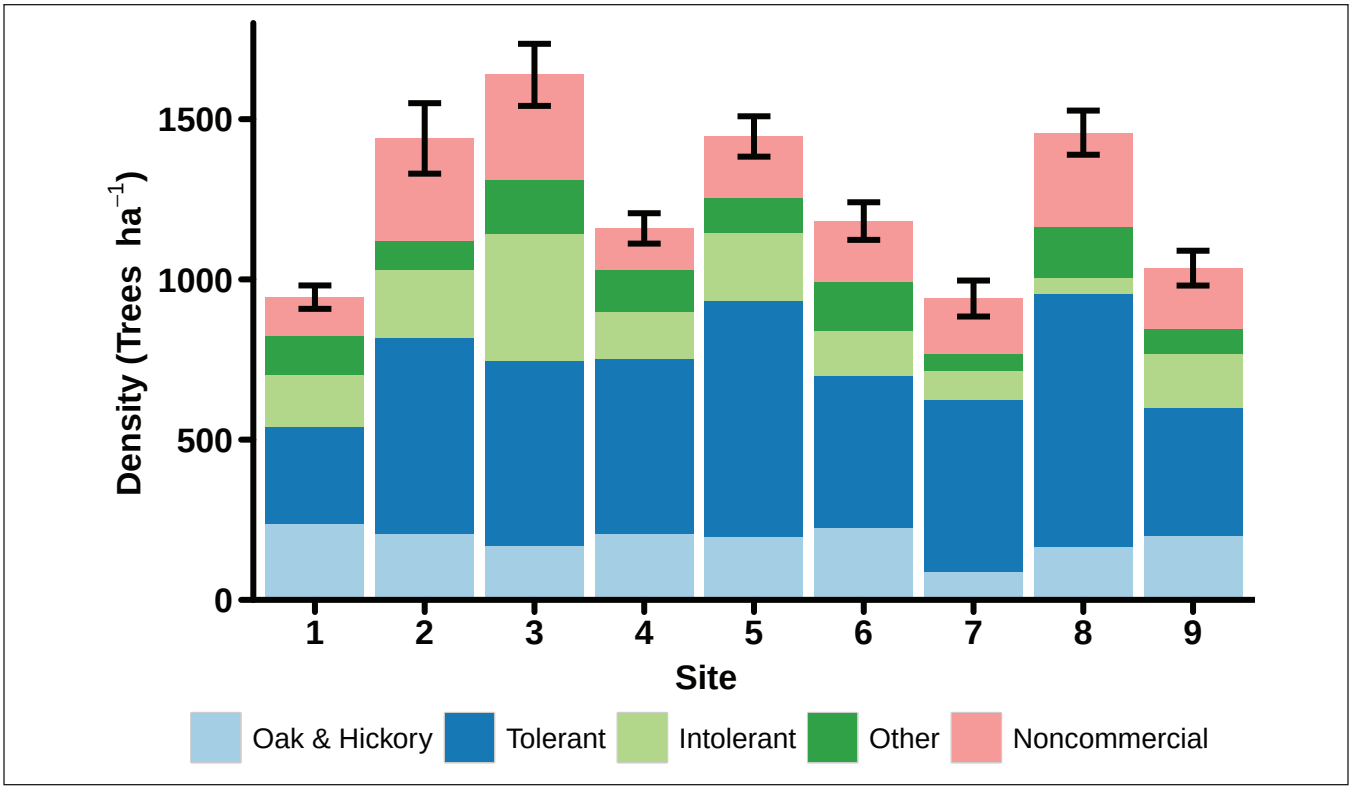


Figure 1.—Stand-level, average tree density of all stems >1.5 cm diameter at breast height by research core. Error bars approximate ± 1 standard error of the total, stand-level tree density. Species group definitions are given in Table 1.

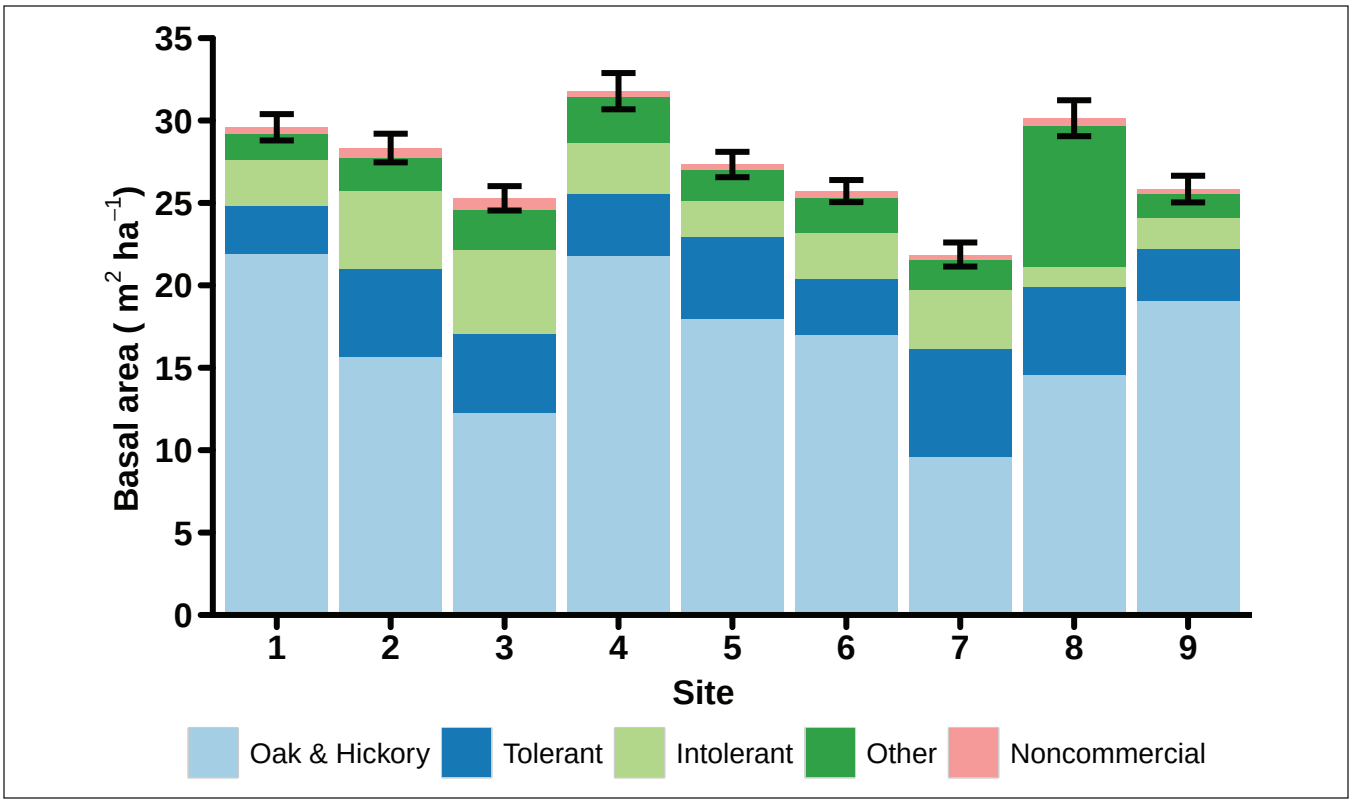


Figure 2.—Stand-level, average basal area of all stems >1.5 cm diameter at breast height by research core. Error bars approximate ± 1 standard error of the total, stand-level basal area. Species group definitions are given in Table 1.

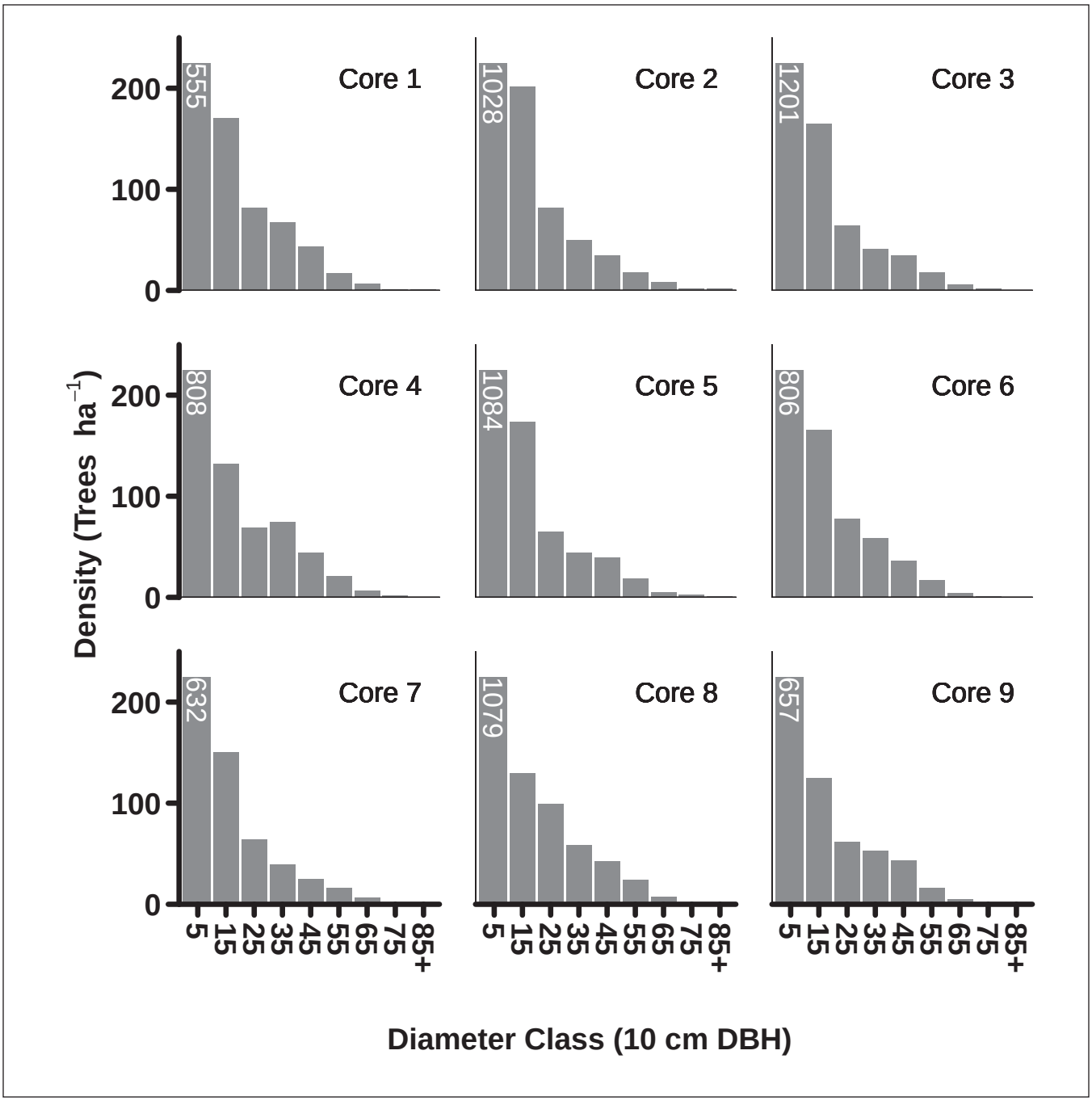


Figure 3.—Tree density distributions by 10-cm diameter classes (labeled at midpoint of each class) of all stems >1.5 cm diameter at breast height for each research core. The white number in the bar for the 0-10 cm diameter class for each core indicated observed density, as it far exceeded the density of all other diameter classes.

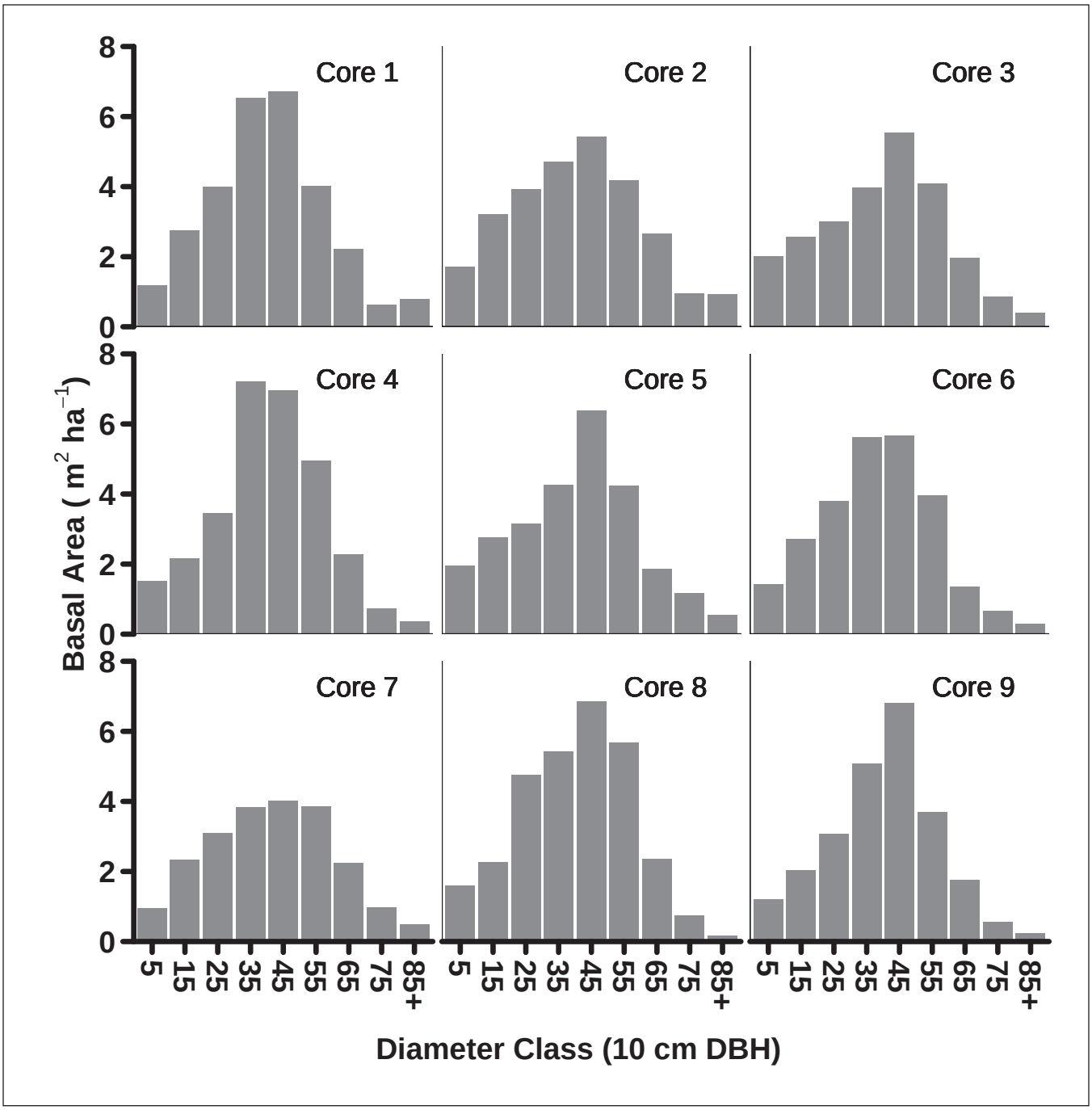


Figure 4.—Basal area distributions by 10-cm diameter classes (labeled at midpoint of each class) of all stems >1.5 cm diameter at breast height for each research core.

Tree species diversity differed significantly across the research cores, with species counts ranging between 4 and 19 species in a sampling plot (Table 2; Fig. 5; Appendix 1). Cores averaged between 2.6 and 4.2 oak (see Table 1 for list of common and scientific names) and hickory species per sampling plot (Table 2), with the most common species being chestnut (experiment averages: 5.0 m^2 ha^{-1} and 59 trees ha^{-1}), white (3.4 m^2 ha^{-1} and 33 trees ha^{-1}), black (2.8 m^2 ha^{-1} and 21 trees ha^{-1}), and northern red (2.7 m^2 ha^{-1} and 24 trees ha^{-1}) oaks; and pignut hickory (1.2 m^2 ha^{-1} and 28 trees ha^{-1}). Oak and hickory species dominated stand basal area in most cores, varying between 44 and 75 percent of total basal area (Fig. 2), but this basal area was concentrated in relatively few large overstory trees (Fig. 1). For example, across all research cores, the average d.b.h. of oaks and hickories in a plot was 36.0 ± 0.4 cm (mean ± standard error), whereas all other species averaged 11.2 ± 0.2 cm.

The majority of stems in all units were shade-tolerant and other commercial species groups. Tolerant species, mostly red and sugar maple, accounted for 297 to 784 trees ha^{-1} but averaged only 8.6 to 10.9 cm in diameter (Table 2). In fact, tolerant species were nearly absent from the overstory size class with only 5 to 22 trees ha^{-1} (Table 2). The noncommercial species group also made up a large percentage of the stems in most research cores, but like the tolerant group, was not particularly large (3.8 to 7.9 cm average diameter) nor numerous in the overstory size class (<1.0 trees ha^{-1}; Table 2). Shade-intolerant species were generally low in abundance in most cores (51 to 354 trees ha^{-1}) but made up a large amount of basal area in many (1.2 to 4.9 m^2 ha^{-1}) (Table 2, Figs. 1 and 2).

There were some significant differences in forest composition among the research cores. Research Core 7 was more heavily dominated by shade-tolerant

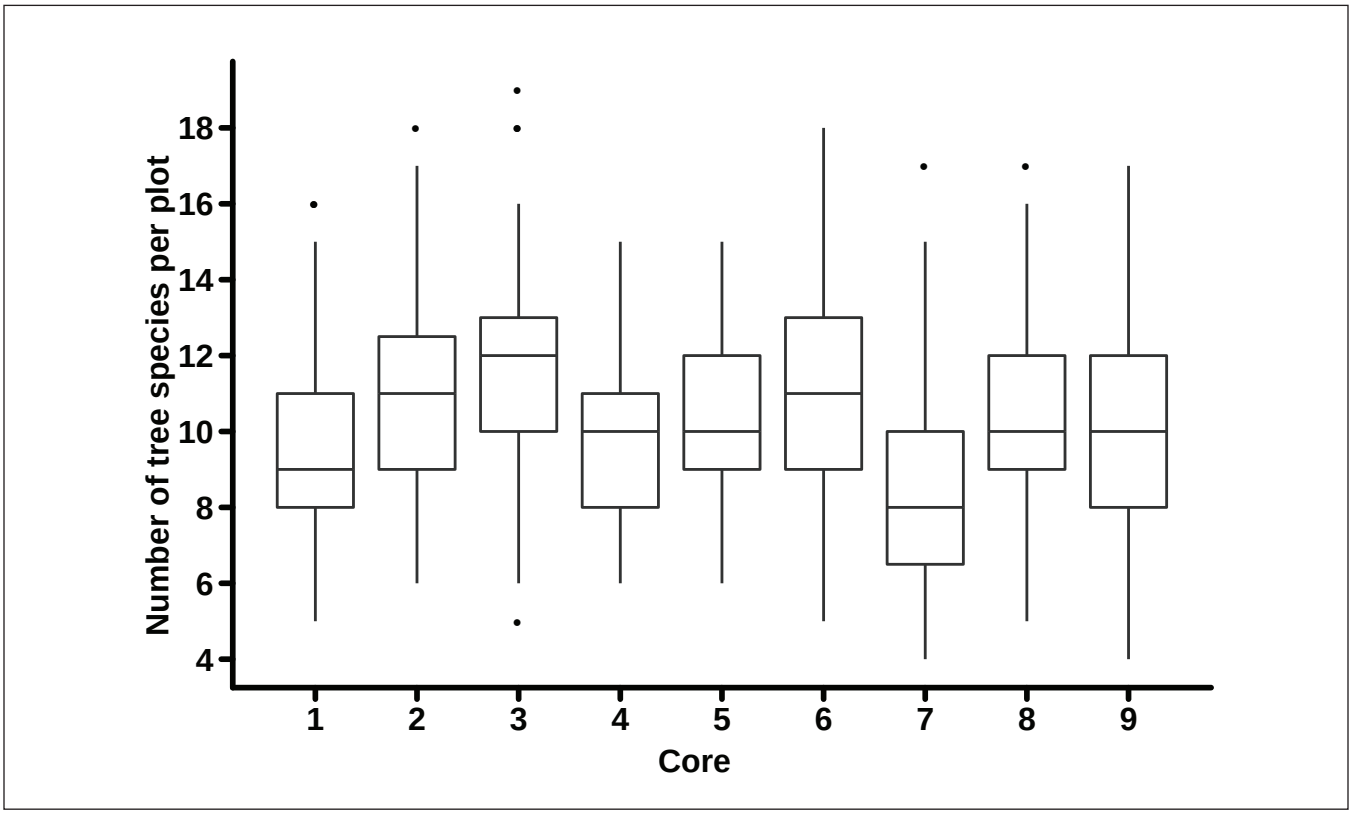


Figure 5.—Box plots of the number of species recorded in plots by research core. Boxes correspond to upper and lower quartiles of the data, with the horizontal line in each box showing the median, and the whiskers approximating the range of the data. Significant outliers in the distribution are shown as individual points.

species and had significantly less oak and hickory basal area and density than almost all other units (Table 2, Appendix 1). Further, it was significantly less diverse than almost any other core ($p < 0.05$; Appendix 1). Research Core 3, on the other hand, was the most diverse and had the highest basal area and density of intolerant species (Table 2, Figs. 1 and 2), suggesting that it had been disturbed more recently than other cores. Research Core 8 was notably different in composition in that it had a much higher density and basal area in the other commercial species group, in terms of both total and overstory stems ($p < 0.05$; Appendix 1). This core had numerous plantings of Virginia pine and eastern white pine throughout; pine species made up 66 percent of the stems and 70 percent of the basal area in this core. Other than Research Core 8, pine was represented well only in Research Core 4, where it composed 43 percent of the stems and 33 percent of the basal area in the overstory size class.

The plot-level NMDS ordination converged on a three-dimension solution after 500 iterations with a stress of 19.33, which is a marginal fit (Kruskal 1964). Nevertheless, this ordination hinted at these minor differences in species importance among research cores (Fig. 6a). The centroid for all plots within Research Core 7 was slightly separated from the other cores in this ordination, and Research Core 8 had several plots in ordination space outside plots of any other research core (i.e., the convex hull extended into quadrat 1 further than any other core). However, the convex hulls of all research cores strongly overlapped one another in ordination space, suggesting a significant overlap in species importance values among all cores (Fig. 6a).

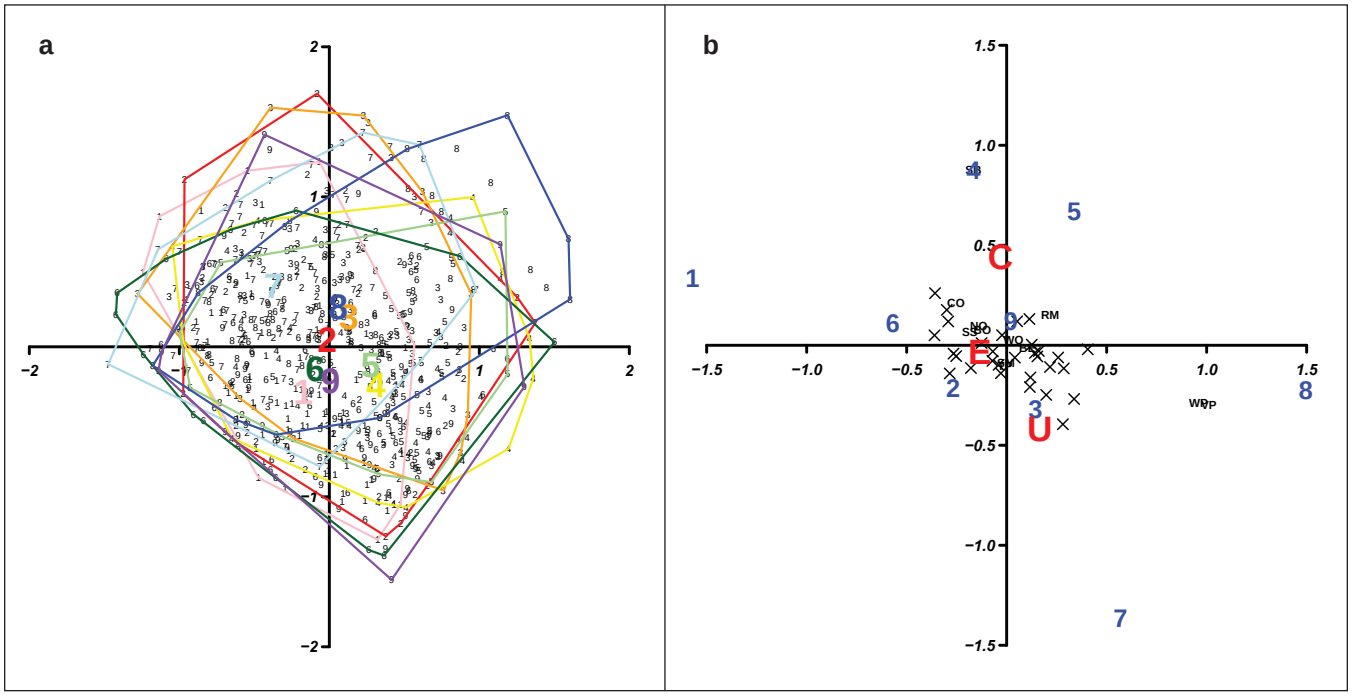


Figure 6.—Nonparametric multidimensional scaling ordinations of species importance values a) by individual plots and b) by research cores. In a), plots are labeled by research core, with centroids and convex hulls of each core shown by colored numbers and lines, respectively. In b), coordinates for individual species and species groups are represented by black "x" or species abbreviations as follows: BO = black oak, CO = chestnut oak, RO = northern red oak, WO = white oak, SM = sugar maple, RM = red maple, BE = American beech, TP = tulip poplar, SS = sassafras, WP = white pine, VP = Virginia pine, and SB = serviceberry. Treatment centroids are shown in red and are abbreviated as follows: C = control, E = even-aged, and U = uneven-aged.

Pre-Treatment Differences

Even though there were significant differences among research cores for many attributes, there were no significant treatment-level differences in tree species diversity, trees per hectare, quadratic mean diameter, or basal area across all treatments, either for all trees or overstory trees alone (p-values ranged from 0.23 to 0.76; Table 3). ANOVAs of major species groups also did not detect significant treatment-level differences in those response variables (p-values ranged from 0.15 to 0.92; Tables 4-8).

Core-level NDMS yielded much more usable results (Fig. 6b). Average stress after 56 iterations for this ordination was 0.045 on two axes, within the good fit zone suggested by Kruskal (1964). Centroids for each research core were distinct from one another in ordination space (e.g., Research Cores 1, 7, and 8), which may explain the significant core-level comparisons. Differences in importance values among cores for certain species caused some of this separation; for example, Research Core 8 was distinctly separated from the other cores because of the strong importance of pine species in that core. However, the centroids of the treatments (red letters, Fig. 6b) are separated much less and, thus, less distinct from one another.

Table 3.—Core means for all woody species by proposed management treatment. Density, quadratic mean diameter (QMD), and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥30 cm d.b.h.). Differences among treatments for each attribute were tested using Eq. 1.

	Proposed treatment				
Attribute	No harvest (cores 2, 4, 5)	Even-aged (cores 3, 6, 9)	Uneven-aged (cores 1, 7, 8)	F	p
No. of species per plot	10.4	10.8	9.5	1.89	0.23
Tot. density (trees ha^{-1})	1337	1217	1098	1.02	0.41
Ovs. density (trees ha^{-1})	126	115	123	0.28	0.76
Tot. QMD (cm)	17.6	17.6	18.8	0.46	0.65
Ovs. QMD (cm)[†]	45.3	44.5	44.8	0.61	0.57
Tot. basal area (m^2 ha^{-1})	27.9	24.8	26.8	1.11	0.39
Ovs. basal area (m^2 ha^{-1})	20.0	17.6	19.3	0.98	0.43

[†] Plots without overstory trees were ignored for this attribute.

Table 4.—Core means for the oak and hickory species group by proposed management treatment. Density, quadratic mean diameter (QMD), and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥30 cm d.b.h.). Differences among treatments for each attribute were tested using Eq. 1.

	Proposed treatment				
Attribute	No harvest (cores 2, 4, 5)	Even-aged (cores 3, 6, 9)	Uneven-aged (cores 1, 7, 8)	F	p
No. of species per plot	3.7	3.8	3.2	2.67	0.15
Tot. density (trees ha^{-1})	200	190	191	1.64	0.27
Ovs. density (trees ha^{-1})	99	87	82	0.80	0.48
Tot. QMD (cm)	37.0	35.1	36.1	0.98	0.42
Ovs. QMD (cm)[†]	45.8	45.3	46.2	0.11	0.90
Tot. basal area (m^2 ha^{-1})	18.2	15.6	15.2	0.80	0.49
Ovs. basal area (m^2 ha^{-1})	16.0	13.4	13.1	0.87	0.46

[†] Plots without overstory trees were ignored for this attribute.

Table 5.—Core means for the shade-tolerant species group by proposed management treatment. Density, quadratic mean diameter (QMD), and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥30 cm d.b.h.). Differences among treatments for each attribute were tested using Eq. 1.

	Proposed treatment				
Attribute	No harvest (cores 2, 4, 5)	Even-aged (cores 3, 6, 9)	Uneven-aged (cores 1, 7, 8)	F	p
No. of species per plot	2.6	2.4	2.2	0.56	0.60
Tot. density (trees ha^{-1})	642	473	537	1.03	0.41
Ovs. density (trees ha^{-1})	8	9	14	0.52	0.62
Tot. QMD (cm)	9.7	10.4	11.1	0.69	0.54
Ovs. QMD (cm)[†]	41.0	40.7	40.8	0.08	0.92
Tot. basal area (m^2 ha^{-1})	4.4	3.7	4.9	0.20	0.82
Ovs. basal area (m^2 ha^{-1})	1.2	1.3	2.0	0.33	0.73

[†] Plots without overstory trees were ignored for this attribute.

Table 6.—Core means for the shade-intolerant species group by proposed management treatment. Density, quadratic mean diameter (QMD), and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥30 cm d.b.h.). Differences among treatments for each attribute were tested using Eq. 1.

	Proposed treatment				
Attribute	No harvest (cores 2, 4, 5)	Even-aged (cores 3, 6, 9)	Uneven-aged (cores 1, 7, 8)	F	p
No. of species per plot	1.3	1.4	1.2	0.15	0.86
Tot. density (trees ha^{-1})	180	209	100	1.56	0.28
Ovs. density (trees ha^{-1})	10	11	9	0.10	0.90
Tot. QMD (cm)	17.8	17.0	19.6	0.17	0.85
Ovs. QMD (cm)[†]	46.3	43.9	45.0	1.04	0.41
Tot. basal area (m^2 ha^{-1})	2.9	3.1	2.4	0.78	0.50
Ovs. basal area (m^2 ha^{-1})	1.7	1.8	1.5	0.14	0.87

[†] Plots without overstory trees were ignored for this attribute.

Table 7.—Core means for the other commercial species group by proposed management treatment. Density, quadratic mean diameter (QMD), and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥30 cm d.b.h.). Differences among treatments for each attribute were tested using Eq. 1.

	Proposed treatment				
Attribute	No harvest (cores 2, 4, 5)	Even-aged (cores 3, 6, 9)	Uneven-aged (cores 1, 7, 8)	F	p
No. of species per plot	1.3	1.4	1.2	1.37	0.32
Tot. density (trees ha^{-1})	103	116	96	1.48	0.30
Ovs. density (trees ha^{-1})	9	7	17	0.10	0.90
Tot. QMD (cm)	15.3	15.9	18.4	1.13	0.38
Ovs. QMD (cm)[†]	39.4	42.2	41.0	0.65	0.55
Tot. basal area (m^2 ha^{-1})	1.9	1.8	3.8	0.19	0.83
Ovs. basal area (m^2 ha^{-1})	1.0	1.1	2.6	0.09	0.91

[†] Plots without overstory trees were ignored for this attribute.

Table 8.—Core means for the noncommercial species group by proposed management treatment. Density, quadratic mean diameter (QMD), and basal area are summarized across all size classes (Tot.; ≥1.5 cm d.b.h.) and for only overstory trees (Ovs.; ≥30 cm d.b.h.). Differences among treatments for each attribute were tested using Eq. 1.

Attribute	Proposed treatment: No harvest (cores 2, 4, 5)	Even-aged (cores 3, 6, 9)	Uneven-aged (cores 1, 7, 8)	F	p
No. of species per plot	1.5	1.8	1.6	1.03	0.41
Tot. density (trees ha^{-1})	212	229	204	0.48	0.64
Ovs. density (trees ha^{-1})	0.4	0.4	0.9	1.62	0.27
Tot. QMD (cm)	6.4	5.7	7.0	0.33	0.73
Ovs. QMD (cm)†	37.6	36.6	40.9	0.88	0.46
Tot. basal area (m^2 ha^{-1})	0.5	0.5	0.6	0.98	0.43
Ovs. basal area (m^2 ha^{-1})	0.05	0.04	0.11	2.30	0.18

† Plots without overstory trees were ignored for this attribute.

Intra-Core Variability

As highlighted by the plot-level NMDS (Fig. 6a), there was a large amount of intra-core variability across all research units. Figures 7 through 15 display plot-level relative basal area composition by species group and by size class for all nine research cores. These maps can be used to look for differences in forest development stages across each core, and by cross-referencing with topographic maps or aerial photography, riparian areas, and other features.

DISCUSSION

Research Core Differences

Even though great care was taken during site selection for the HEE, the woody communities still differed among individual research cores as measured by several structural attributes in this study. Several reasons may explain these differences. Although all the HEE cores are within the Brown County Section of the Highland Rim Natural Region, and all are underlain by the Wellston-Berks-Gillpin soil complex (Jenkins, this publication), some differences among cores may result from small-scale and/or more variable topographic and soil conditions within individual cores. For example, Research Cores 6 and 9 are characterized by much more extreme relief than most of the other cores, and all cores differ strongly by the amount of bottomland soil types distributed along the numerous ephemeral and perennial streams that dissect and/or border each site (Natural Resources Conservation Service 2011). Often these bottomland areas are characterized by a much larger proportion of the "other commercial" species group, which can be detected on Figures 7 through 15 as linear features within the cores (e.g., on Figure 9 from the northwest corner running southeast two-thirds of the way through the core).

Overall land use and disturbance histories were likely similar for most cores, as a significant part of each was likely tilled and/or grazed until Indiana acquired the Morgan-Monroe and Yellowwood State Forest lands in 1929 and 1947, respectively (Carman, this publication; Jenkins, this publication). For example, many overstory oaks within the research cores date to the mid- to late-1800s and have relatively rapid juvenile growth[2], suggesting that stands were originally at a much lower stem density. After State acquisition, stands likely slowly increased in density as more shade-tolerant species became established in the understory with fire suppression (Jenkins, this publication). Although a full stand reconstruction has not been done for the HEE cores, this development pattern was also seen in the nearby Hoosier National Forest (Jenkins and Parker 2000).

[2] Unpublished data on file with M.R. Saunders, Department of Forestry and Natural Resources, Purdue University, 715 West State St., West Lafayette, IN 47907.

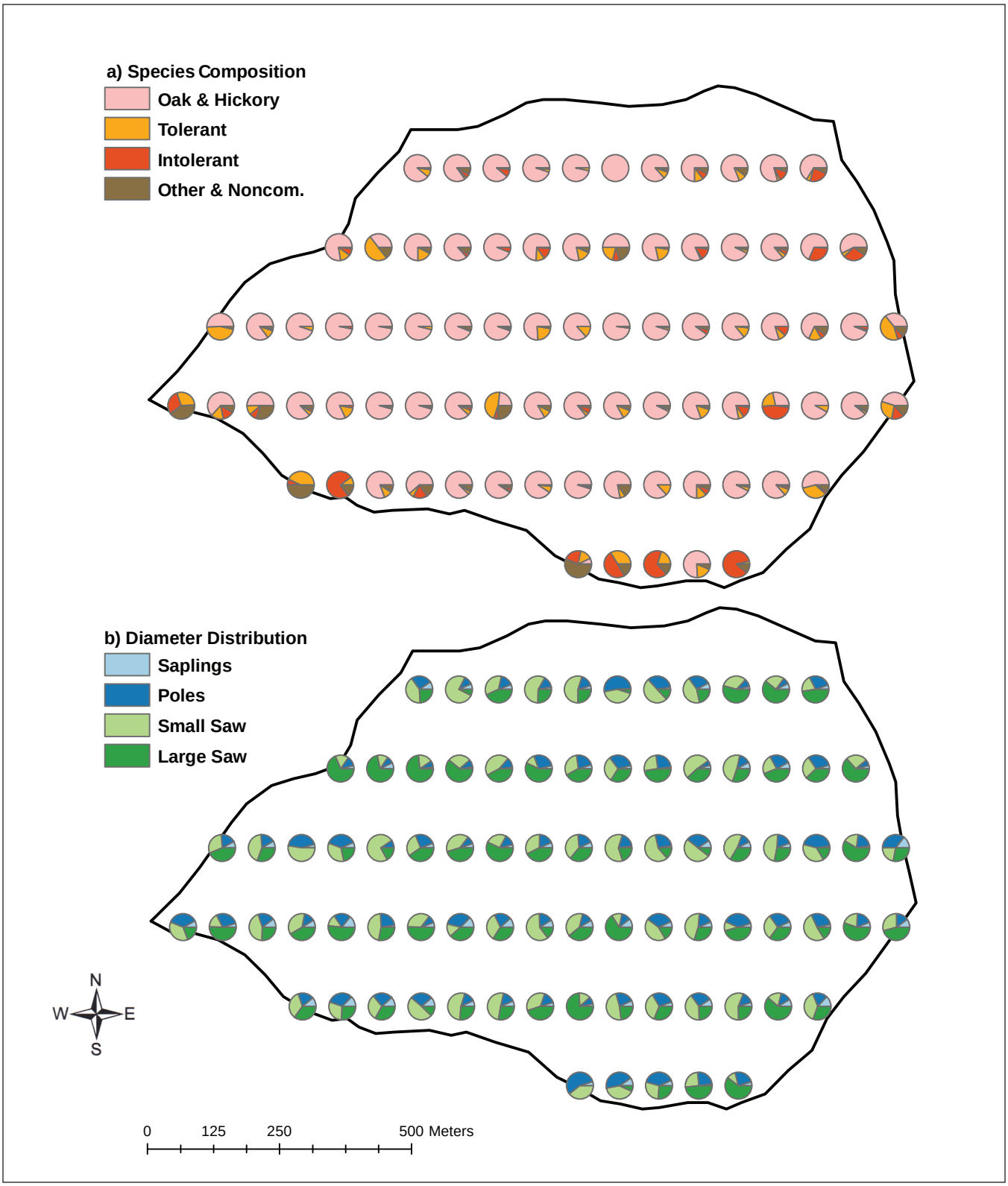


Figure 7.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 1 at Morgan-Monroe State Forest in south-central Indiana. Individual pie charts are plotted over plot center positions and represent proportions of total plot basal area within each species group (a) or size class (b). Species group definitions are given in Table 1; diameter distributions are defined as saplings, 1.5 – 12.4 cm; poles, 12.5 – 30.5 cm; small sawtimber, 30.6 – 45.6 cm; and large sawtimber, 45.7 cm and above.

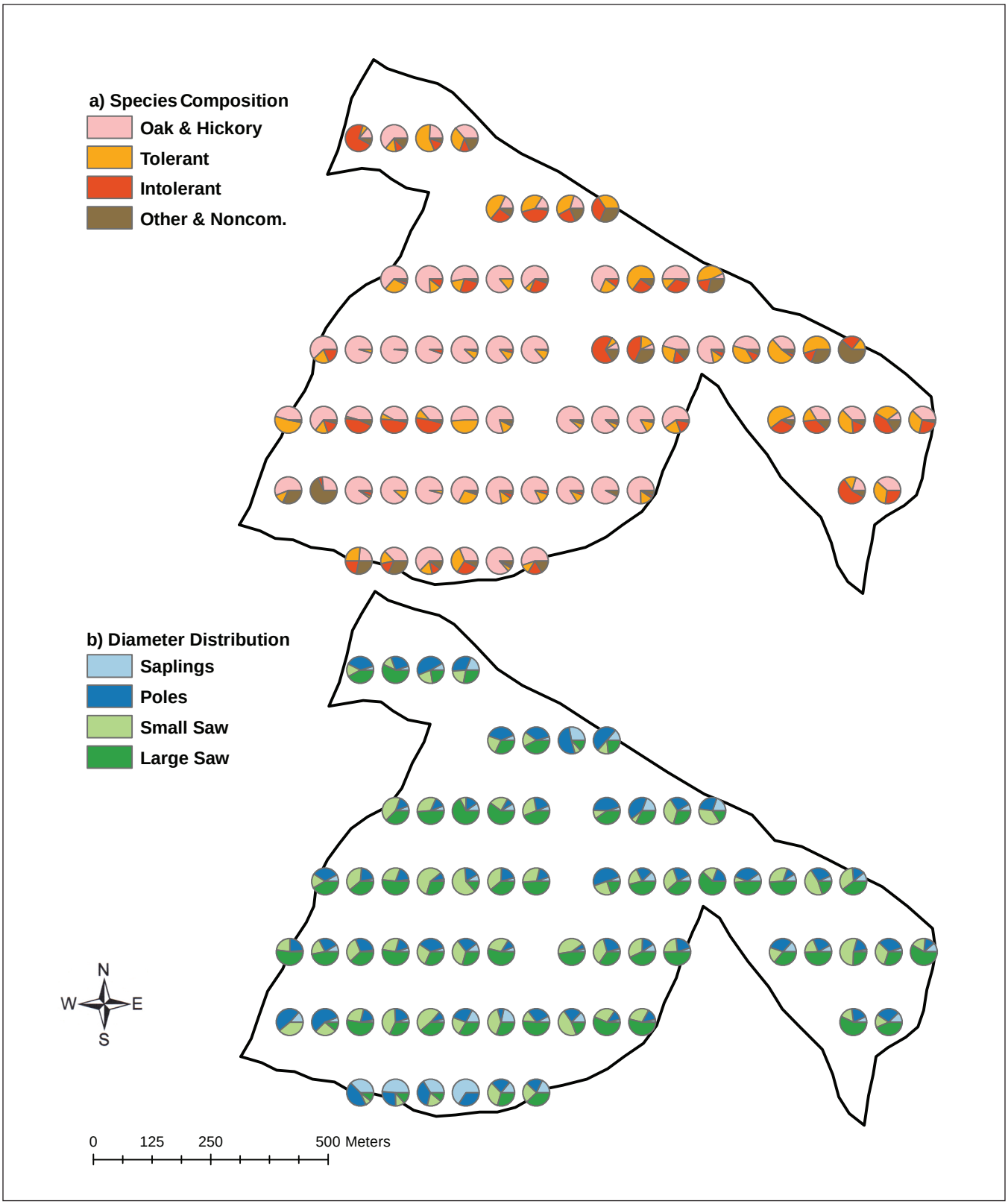


Figure 8.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 2 at Morgan-Monroe State Forest in south-central Indiana. Information summarized as in Figure 7.

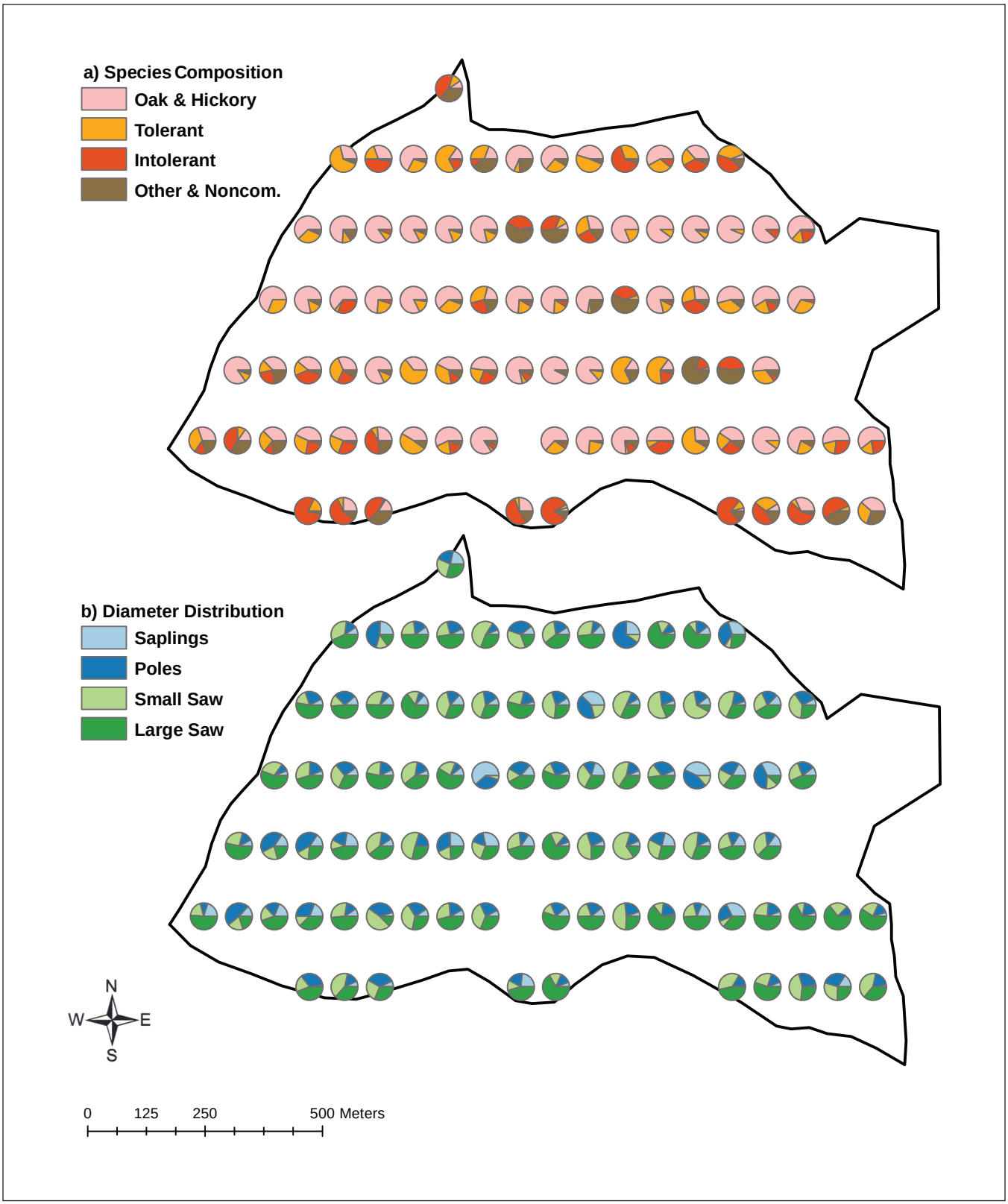


Figure 9.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 3 at Morgan-Monroe State Forest in south-central Indiana. Information summarized as in Figure 7.



Figure 10.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 4 at Morgan-Monroe State Forest in south-central Indiana. Information summarized as in Figure 7.

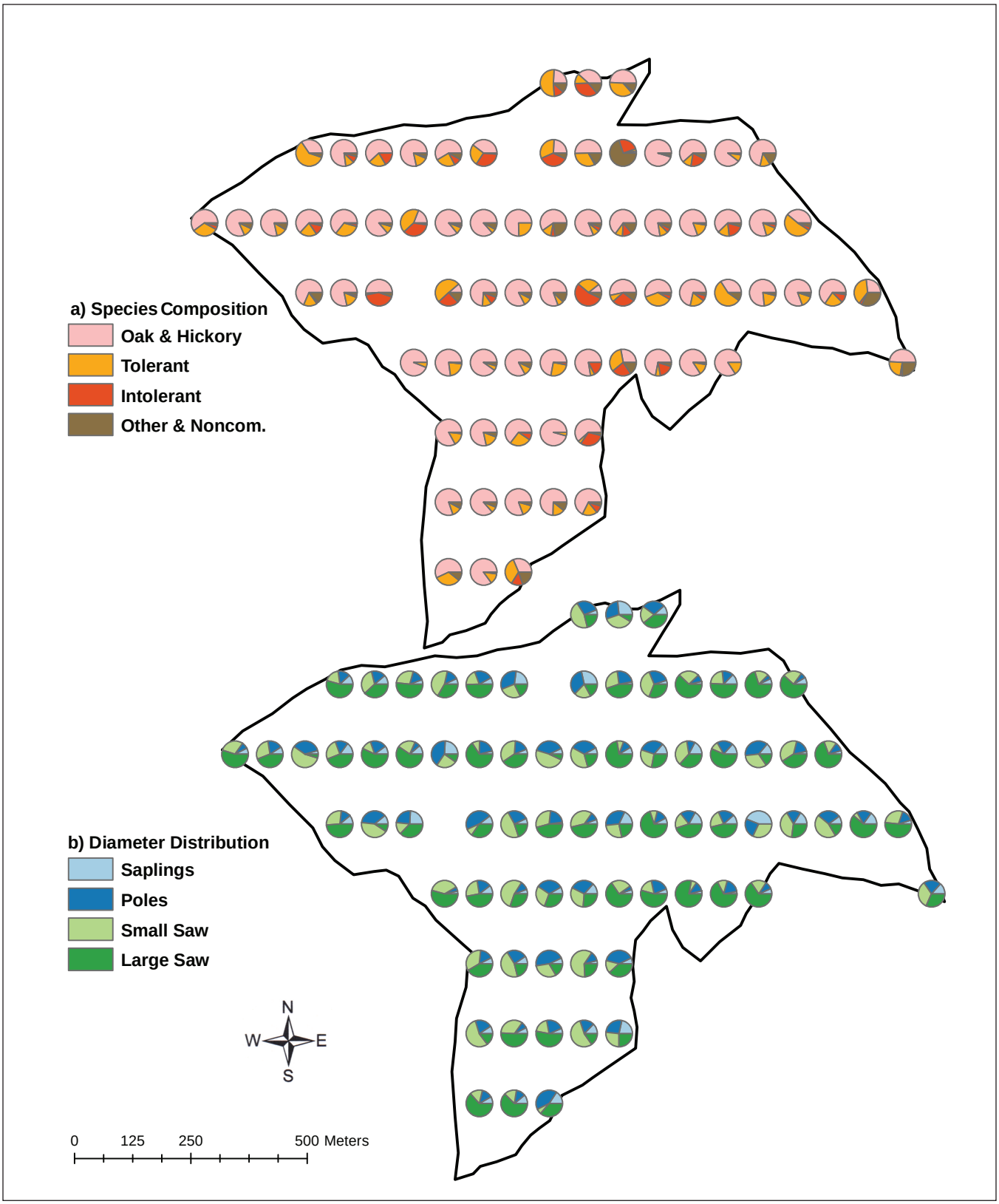


Figure 11.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 5 at Morgan-Monroe and Yellowwood State Forests in south-central Indiana. Information summarized as in Figure 7.

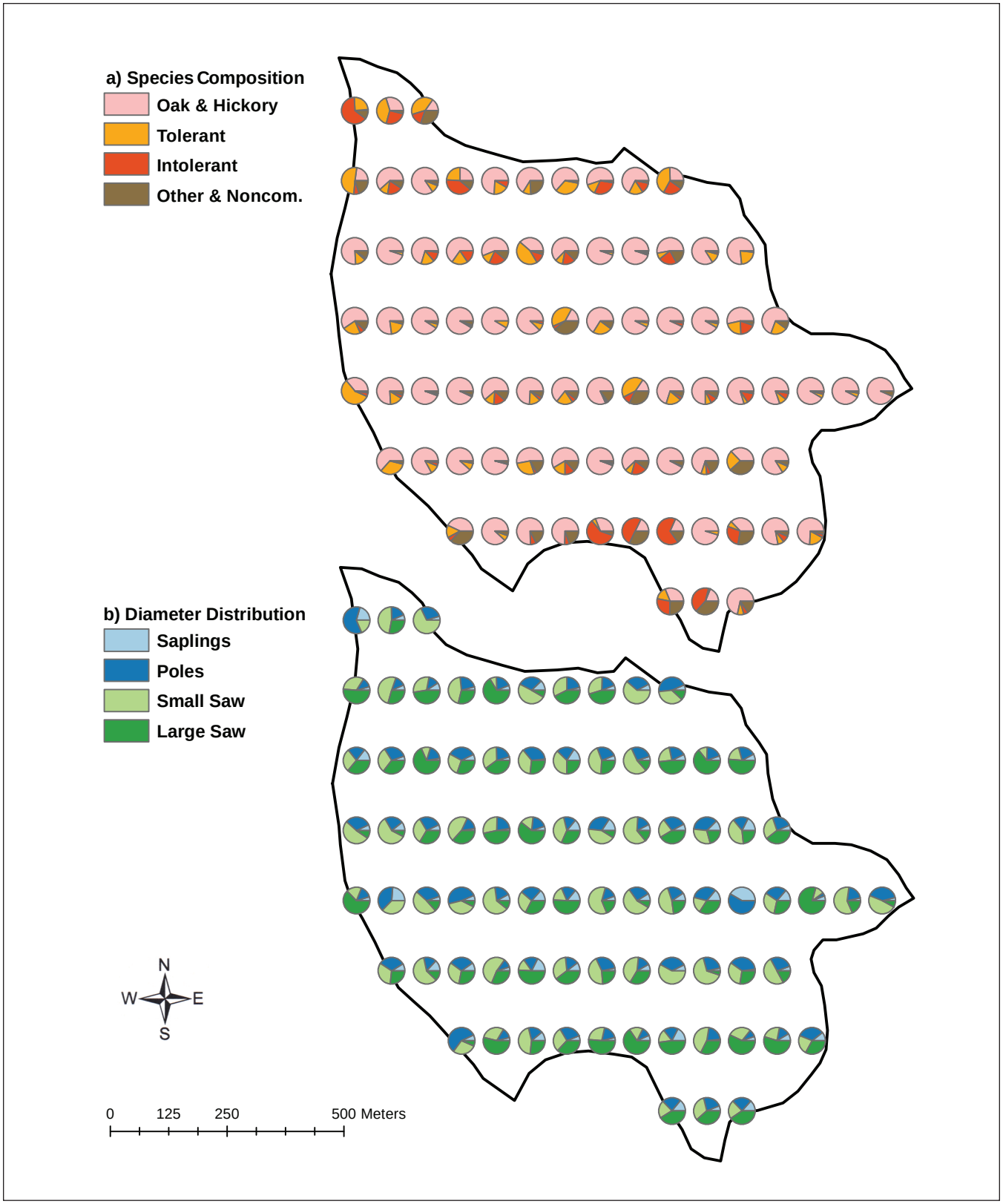


Figure 12.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 6 at Morgan-Monroe and Yellowwood State Forests in south-central Indiana. Information summarized as in Figure 7.

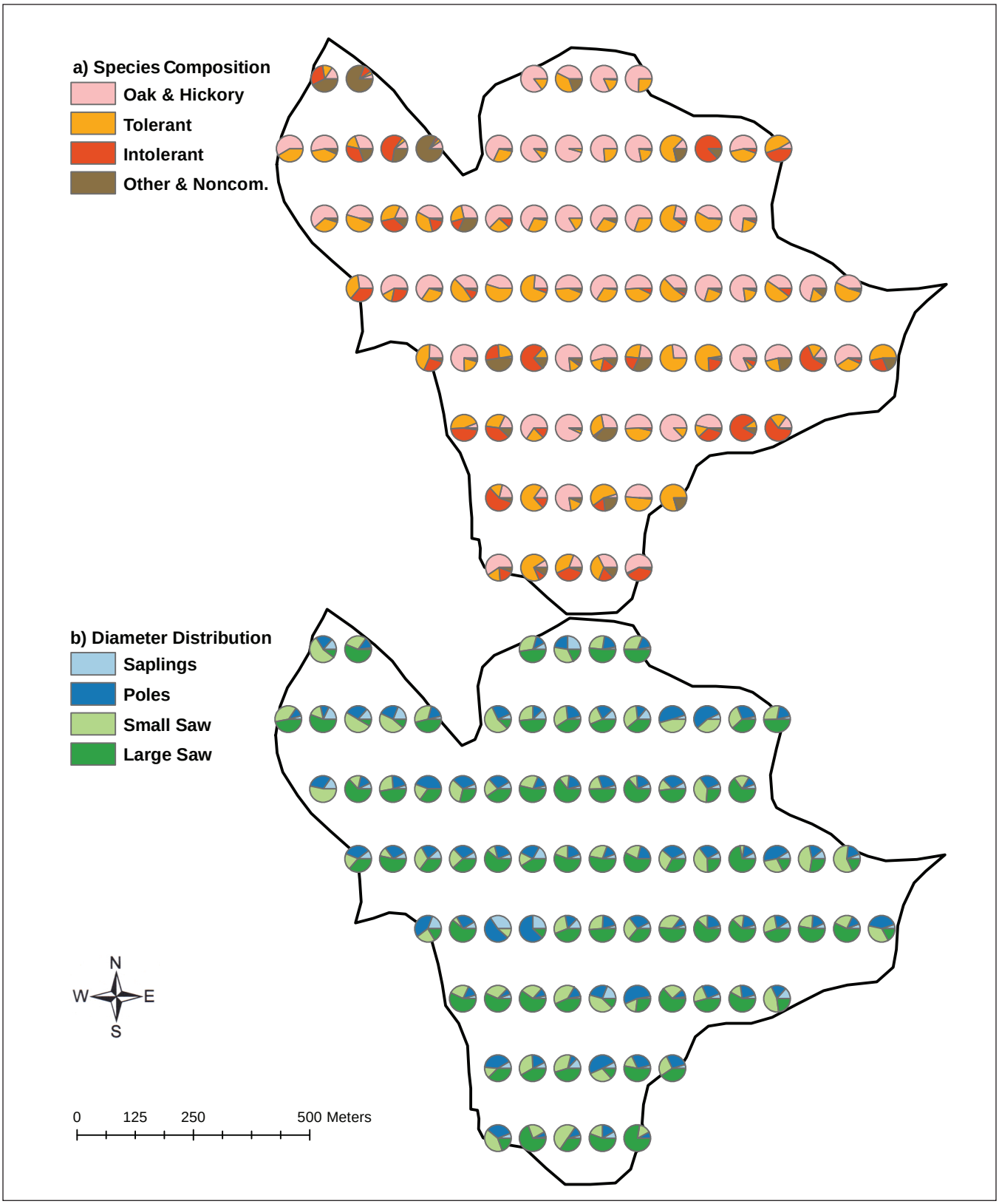


Figure 13.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 7 at Yellowwood State Forest in south-central Indiana. Information summarized as in Figure 7.

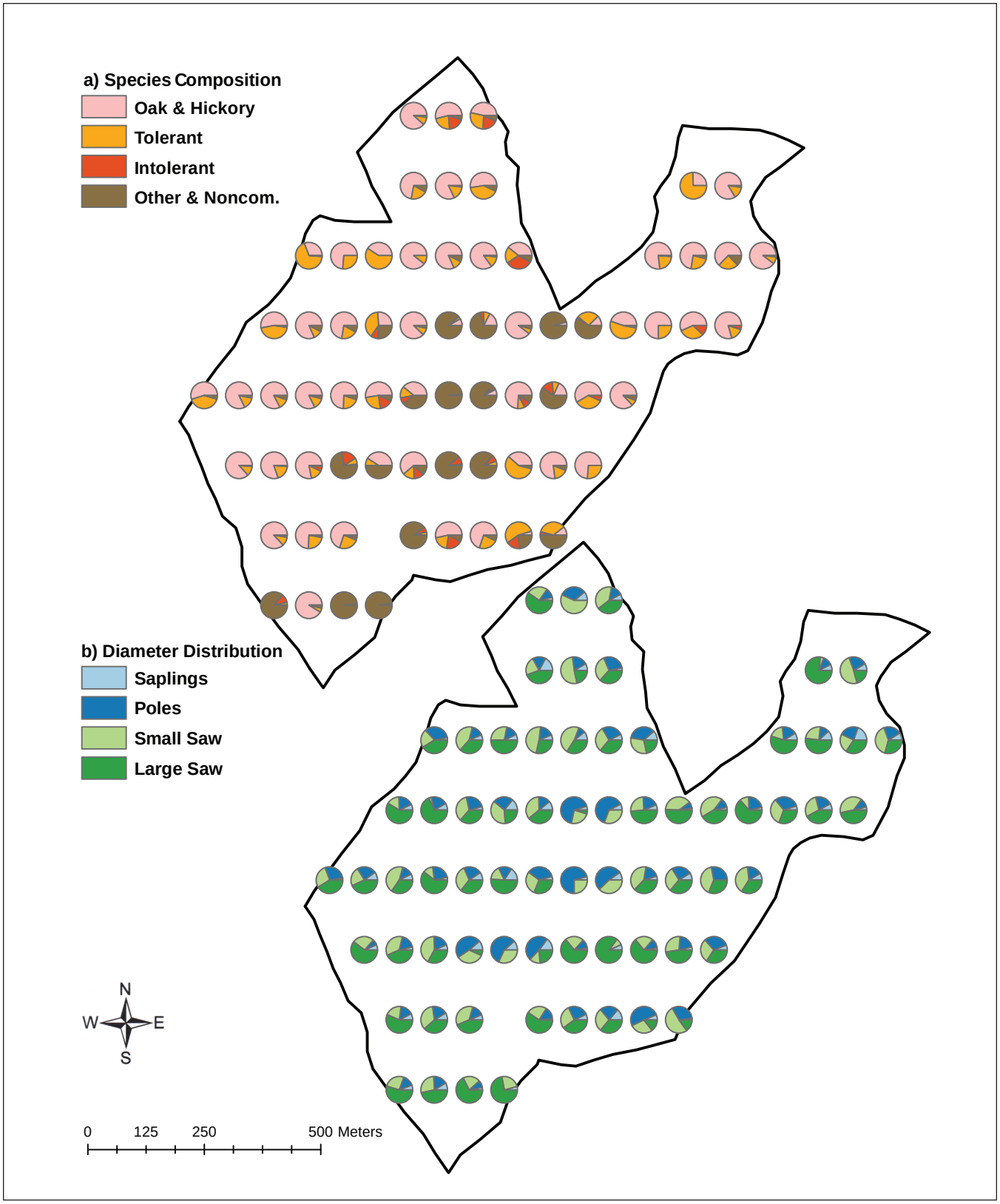


Figure 14.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 8 at Yellowwood State Forest in south-central Indiana. Information summarized as in Figure 7.

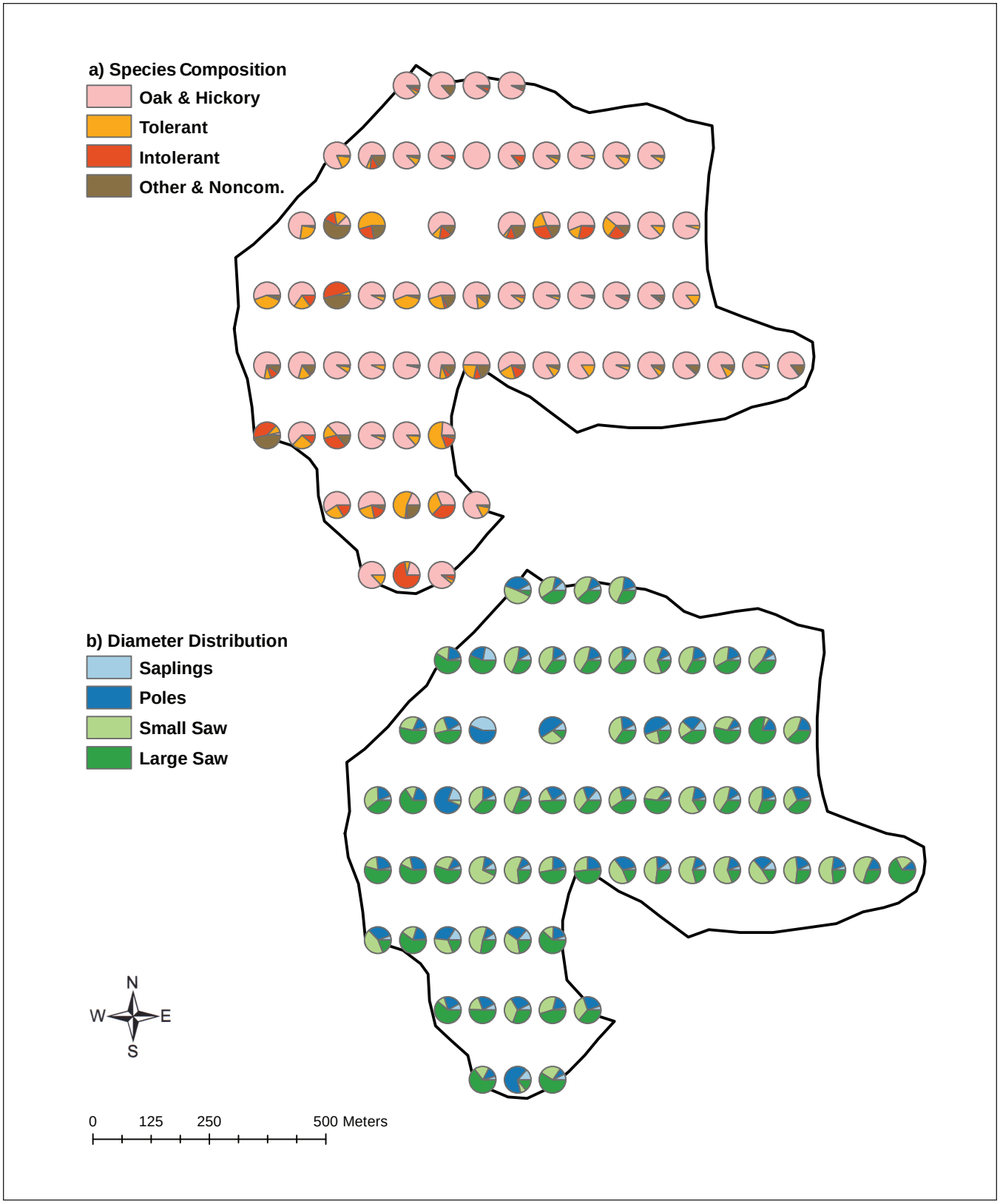


Figure 15.—Pre-treatment, plot-level a) species composition and b) diameter distribution of all stems >1.5 cm diameter at breast height within Research Core 9 at Yellowwood State Forest in south-central Indiana. Information summarized as in Figure 7.

Nevertheless, more recent differences in land use and disturbance histories likely have contributed to the structural and compositional differences among cores. For example in Research Core 8 (Fig. 14), the large block of plots with a higher proportion of "other commercial" species and a higher proportion of poles than elsewhere corresponds to the numerous white pine and Virginia pine plantings within that core. These plantings have been associated with a Depression-era homestead within a riparian area on the site. The more northern cores located within the Morgan-Monroe State Forest (i.e., Research Cores 1-4) have been disturbed more frequently from windstorms and tornados than the southern sites over the last 20 years.[3] These events resulted in a significantly higher component of intolerant tree species in some of these cores, particularly for Research Core 3 (Table 1, Appendix 1).

Implications for Future Research

Fortuitously, differences among the research cores offset one another when averaged across the three replicates of each treatment. We can say confidently—at the proposed treatment level—that there are no significant pre-treatment differences in overstory and large shrub communities. However, we recommend that researchers be cognizant of potentially strong site-to-site differences that could mask treatment effects, particularly if only subsets of the experimental units are used in the analysis. For example, the most northern set of replicates, Research Cores 1–3, may likely respond very differently from the other units because of the higher proportion of intolerant stems.

Blocking may be effective, in this case, to alleviate these concerns, although the geographical isolation of Research Cores 4 and 5 may make them difficult to use as experimental controls for the southern cores (Kalb and Mycroft, this publication: Fig. 1). Blocking was used a posteriori to account for harvesting differences among contractors (Kalb and Mycroft, this publication), as the time frame allowed for the harvest and the total area was deemed too difficult for one contractor to do alone. Research Cores 1 and 3, 6 and 7, and 8 and 9 were bid separately, although the last two sets were purchased by the same contractor. Analysis after the first complete post-treatment inventory in 2013 should clarify whether a contractor effect exists and whether blocking will be necessary in those analyses.

Lastly, Figures 7 through 15 highlight the high degree of variability in structure within each core. As Puettmann et al. (2009) point out, understanding this variability is critical to assessing post-treatment tree regeneration, herbaceous diversity, and other variables, as mean values alone may mask the true environmental and ecological drivers resulting from treatment. Resources need to be dedicated to finer scale, plot-level modeling of abiotic factors, most notably soil type, and past disturbance and land use history. Spatially explicit sampling and analysis should allow better linkage of pattern with process, particularly at finer scales than the research core. With that intent, we encourage individuals to overlay sampling designs on this grid of overstory plots, thereby providing benefits to other researchers involved in the project.

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[3] Jim Allen, Indiana Department of Natural Resources-Division of Forestry, Yellowwood/Morgan-Monroe State Forest, personal communication

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APPENDIX 1.

Denominator degrees of freedom (n), F statistics, and p-values for one-way analysis of variance tests for differences among research cores for the following attributes. Research cores with the same letter are not significantly different for that attribute using a Tukey's HSD test at α = 0.05. Means, standard errors, and ranges for each attribute are found in Table 2.

				Research Core								
Attribute	n	$F_{8,n}$	p	1	2	3	4†	5	6	7	8	9
All Species												
No. of species per plot	654	12.01	<0.001	cd	ab	a	bc	bc	ab	d	abc	bc
Tot. density (trees ha^{-1})	654	12.83	<0.001	c	ab	a	bc	ab	bc	c	ab	c
Ovs. density (trees ha^{-1})	654	12.10	<0.001	ab	cde	de	a	de	bcd	e	abc	bcd
Tot. QMD (cm)	654	9.56	<0.001	a	bc	c	ab	c	bc	ab	bc	ab
Ovs. QMD (cm)‡	651	2.46	0.012	a	a	a	a	a	a	a	a	a
Tot. Basal area (m^2 ha^{-1})	654	12.40	<0.001	abc	abcd	de	ab	bcd	cd	e	a	de
Ovs. Basal area (m^2 ha^{-1})	654	7.18	<0.001	ab	abc	c	a	bc	bc	c	ab	bc
Oak and Hickory												
No. of species per plot	654	8.54	<0.001	ab	ab	ab	ab	a	a	c	bc	ab
Tot. density (trees ha^{-1})	654	7.23	<0.001	a	a	a	a	a	ab	b	a	a
Ovs. density (trees ha^{-1})	654	16.82	<0.001	ab	cde	ef	a	bcd	bc	f	de	abc
Tot. QMD (cm)	654	2.29	0.020	a	a	a	a	a	a	a	a	a
Ovs. QMD (cm)‡	609	3.14	0.002	b	ab	ab	b	ab	b	a	ab	ab
Tot. Basal area (m^2 ha^{-1})	654	15.78	<0.001	a	bc	cd	a	ab	bc	d	bc	ab
Ovs. Basal area (m^2 ha^{-1})	654	13.46	<0.001	a	bc	cd	a	ab	bc	d	bcd	ab
Tolerant												
No. of species per plot	654	13.20	<0.001	e	ab	ab	abc	abc	cde	de	a	bcd
Tot. density (trees ha^{-1})	654	15.24	<0.001	e	bc	cd	bc	ab	cde	cd	a	de
Ovs. density (trees ha^{-1})	654	8.89	<0.001	bc	bc	b	c	bc	bc	a	ab	bc
Tot. QMD (cm)	654	6.59	<0.001	b	b	b	b	b	b	a	b	ab
Ovs. QMD (cm)‡	276	0.59	0.785	a	a	a	a	a	a	a	a	a
Tot. Basal area (m^2 ha^{-1})	654	10.37	<0.001	d	ab	bc	cd	bc	cd	a	ab	cd
Ovs. Basal area (m^2 ha^{-1})	654	7.20	<0.001	bc	bc	abc	c	bc	bc	a	ab	bc

(Appendix 1 continued on next page)

APPENDIX 1 (continued).

Denominator degrees of freedom (n), F statistics, and p-values for one-way analysis of variance tests for differences among research cores for the following attributes. Research cores with the same letter are not significantly different for that attribute using a Tukey's HSD test at α = 0.05. Means, standard errors, and ranges for each attribute are found in Table 2.

				Research Core								
Attribute	n	$F_{8,n}$	p	1	2	3	4[†]	5	6	7	8	9
Intolerant												
No. of species per plot	654	5.40	<0.001	abc	ab	a	c	abc	abc	bc	c	c
Tot. density (trees ha^{-1})	654	8.87	<0.001	bc	b	a	bc	bc	bc	bc	c	bc
Ovs. density (trees ha^{-1})	654	4.47	<0.001	abc	ab	a	abc	abc	abc	abc	bc	c
Tot. QMD (cm)	654	4.86	<0.001	bcd	a	abcd	cd	abcd	abc	ab	cd	d
Ovs. QMD (cm)[‡]	228	1.27	0.308	a	a	a	a	a	a	a	a	a
Tot. Basal area (m^2 ha^{-1})	654	6.26	<0.001	bcd	ab	a	cd	cd	abcd	abc	d	cd
Ovs. Basal area (m^2 ha^{-1})	654	5.19	<0.001	abc	a	a	abc	bc	abc	ab	bc	c
Other Commercial												
No. of species per plot	654	3.63	<0.001	b	ab	a	ab	ab	a	ab	a	ab
Tot. density (trees ha^{-1})	654	4.00	<0.001	ab	ab	a	ab	ab	a	b	a	ab
Ovs. density (trees ha^{-1})	654	8.43	<0.001	b	b	b	b	b	b	b	a	b
Tot. QMD (cm)	654	2.42	0.014	b	ab	ab	b	ab	ab	ab	a	ab
Ovs. QMD (cm)[‡]	182	1.64	0.117	a	a	a	a	a	a	a	a	a
Tot. Basal area (m^2 ha^{-1})	654	9.62	<0.001	b	b	b	b	b	b	b	a	b
Ovs. Basal area (m^2 ha^{-1})	654	7.95	<0.001	b	b	b	b	b	b	b	a	b
Noncommercial												
No. of species per plot	654	4.77	<0.001	b	ab	a	b	b	ab	b	ab	b
Tot. density (trees ha^{-1})	654	4.42	<0.001	bc	a	a	c	abc	abc	abc	ab	abc
Ovs. density (trees ha^{-1})	654	1.69	0.098	a	a	a	a	a	a	a	a	a
Tot. QMD (cm)	654	4.97	<0.001	a	ab	bc	bc	bc	abc	abc	bc	c
Ovs. QMD (cm)[‡]	27	0.37	0.929	a	a	a	a	a	a	a	a	a
Tot. Basal area (m^2 ha^{-1})	654	4.90	<0.001	ab	a	a	c	bc	abc	abc	abc	c
Ovs. Basal area (m^2 ha^{-1})	654	1.54	0.139	a	a	a	a	a	a	a	a	a

[†] Data from 19 plots were discarded from this analysis due to questionable plot-level basal areas far exceeding 50 m^2 ha^{-1}.

[‡] Plots without overstory trees in the species group were ignored.

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.

BREEDING BIRD COMMUNITIES OF THE HARDWOOD ECOSYSTEM EXPERIMENT

Melissa C. Malloy and John B. Dunning, Jr.[1]

Abstract.—Declining population trends of breeding birds associated with mature forests of the eastern and central United States have been a major concern for conservationists and land managers. As a landscape-scale, long-term, manipulative experiment, the Hardwood Ecosystem Experiment (HEE) in Indiana may provide important insights into factors associated with these declines. Accordingly, we quantified the characteristics of the breeding bird community at the HEE sites using 3 years of pre-treatment data. We also compared the HEE breeding bird community to similar data collected for the long-term monitoring program of the nearby Hoosier National Forest (HNF). From 2006 through 2008, we observed 74 species of birds across 240 survey points within the HEE sites, for a total of 17,806 detections. Of these, 62 species representing 17,340 detections were known to breed in the area. We adjusted the maximum counts at each survey point for differences in detectability, with small changes in adjusted density estimates for most species. Of the covariates used in modeling detectability, only sampling period (early/late in the morning) appeared consistently in the better fitting models. Abundance rankings showed that the HEE breeding avifauna was very similar to that of the HNF. Overall, the results suggested that the HEE sites supported a breeding bird community typical of mature Central Hardwood Forests and that these pre-treatment data will be useful to detect changes associated with the HEE treatments.

INTRODUCTION

The decline of Neotropical migrant birds has concerned researchers for at least three decades (Finch and Stangel 1993, Herkert 1995, Peterjohn et al. 1995, Robbins et al. 1989, Terborgh 1980). This concern has been especially focused on forest songbirds of the eastern and central United States. A reduction in the quality and/or quantity of the breeding grounds of these birds has been hypothesized as a contributing factor to this decline. Loss of wintering habitat or stopover migration habitat might also contribute to the decline, but research has been less conclusive on those impacts (Clawson et al. 1997, Packett and Dunning 2009).

Concerns over these declines have prompted considerable research examining the effects of forest management on avian communities (Duguay et al. 2000). However, much of the current published research consists of studies that were conducted with limited spatial and temporal scales (Mitchell et al. 2001) or limited replication (Thompson et al. 2000). Such studies lack utility because of their limited experimental design (Gram et al. 2003, Marzluff et al. 2000, Sallabanks et al. 2000). There is a need for more long-term studies (Collins 2001) as well as more manipulative experiments in heavily forested landscapes in the Central Hardwoods Region (Sallabanks et al. 2000, Thompson et al. 2000).

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Although several researchers have studied effects of forest management on migratory birds in forest-dominated landscapes (Annand and Thompson 1997, Thompson et al. 1992), these studies were not manipulative experiments and hence the researchers were able to examine only correlative or observational effects.

There also has been considerably less research investigating the effects of uneven-aged silvicultural schemes compared to even-aged management (Campbell et al. 2007). Other avian ecologists have conducted studies in fragmented habitats, and the results might not be applicable to more forested regions (Hartley and Hunter 1998, Robinson and Robinson 1999). Forest tracts in fragmented regions generally show decreased nesting success of mature forest birds, due to higher rates of brood parasitism by brown-headed cowbird (see Table 3 for list of common and scientific names of bird species) and increased nest predation by mammals, snakes, American crow, blue jay, and common grackle (Gates and Gysel 1978, Robbins 1979, Robinson et al. 1995, Wilcove 1985). Several studies have found that nest parasitism and predation are ameliorated when the forest tracts are part of a larger forested region (Donovan et al. 1997, Hanski 1996, Hartley and Hunter 1998, Temple and Cary 1988). Conversely, Flaspohler et al. (2001), King et al. (1996), and Manolis et al. (2002) have found decreased nesting success in relation to distance from clearcut edges even in heavily forested habitats. Landscape-level factors might determine the severity of local-level factors (Robinson et al. 1995).

In addition to declines in species that prefer mature forests, a decline of disturbance-dependent species is a concern (Brawn et al. 2001, Herkert 1995). These species can benefit from forest management, but to varying degrees (Schlossberg and King 2009). Some species prefer very young forests and peak in abundance immediately after harvest; other species have the highest abundance roughly 10 years after harvest (Schlossberg and King 2009). After about 20 years, most disturbance-dependent species are absent because the canopy has closed and shade eliminates understory vegetation (DeGraaf 1991).

The Hardwood Ecosystem Experiment (HEE) is located in the Central Hardwoods Region of the midwestern United States and is designed as a landscape-scale, long-term, manipulative research experiment. The project will examine the ecological and social impacts of forest management through a replicated series of study areas in south-central Indiana. The HEE is designed as a 100-year project to study the effects of even-aged management, uneven-aged management, and no timber harvest (control) on fauna and flora. There are three replicates of each management type spread over 30 linear kilometers and a total research area of more than 3,600 ha. Pre-treatment data collection began in 2006 and continued for 3 years prior to any manipulation. This process allowed us to study the spatial distribution of species in the region in the absence of treatments and will enable us to compare those findings to post-treatment data at each stage of the experiment. The silvicultural prescriptions of the HEE provide an opportunity to test the impacts of forest management strategies on mature forest-dwelling Neotropical migrants as well as those associated with early-successional habitat.

South-central Indiana is one of the few largely forested regions remaining in the midwestern United States, and it is possible that the area serves as a source population for other, more fragmented, areas of the Midwest (Robinson et al. 1995). Even though the research sites of the HEE are not located in a fully contiguous tract of forest, they are part of an overall forest region and are not isolated patches within agricultural land (Kalb and Mycroft, this publication: Fig. 1). The oak-hickory (*Quercus-Carya*) dominance of the HEE and elsewhere in the Central Hardwoods Region is threatened by altered disturbance regimes, increased white-tailed deer (*Odocoileus virginianus*) populations, and introduced diseases and pathogens (Kalb and Mycroft, this publication; McShea et al.

2007). The decline of oaks and hickories would have a profound impact on ecological and economic factors in Indiana, and the HEE will provide insight on possible methods of supporting oak-hickory forests.

Our objective was to compare conditions across management regimes before treatment to provide a baseline that shows how bird populations fluctuate over space and time. The long-term objective in this study is to determine the effects of forest management on breeding birds by assessing population differences among even-aged, uneven-aged, and no harvest management regimes.

A second objective in this paper is to compare our results to those of similar surveys conducted in mature forest communities in nearby areas. The U.S. Department of Agriculture, Forest Service has conducted breeding bird monitoring surveys in the nearby Hoosier National Forest (HNF) since 1991[2] (Thompson et al. 2002). The HNF surveys are conducted in mature forest stands and both projects use similar survey protocols. Therefore we expect the bird communities to be similar. By comparing HEE and HNF data from the same years, we expect to show that, prior to the HEE harvest treatments, the HEE sites supported bird communities typical of mature forests as measured in other long-term monitoring projects in the region.

STUDY AREA

This study took place in Morgan-Monroe and Yellowwood State Forests in south-central Indiana. The landscape consists of an oak-hickory forest type with steep hills and valleys in bedrock-derived soils (Jenkins, this publication). The area was once dominated by agriculture but began regenerating into forest in the early 1900s when the land became incorporated into the HNF and state forests (Jenkins, this publication). Kalb and Mycroft (this publication) give greater details of the HEE research area and study design.

The HEE uses nearly 20 percent of the Morgan-Monroe and Yellowwood State Forests and covers 3,603 ha. There are nine management units, and each includes a core research area and a buffer area (Kalb and Mycroft, this publication: Fig 1). These units are divided into three control (no harvest) units, three even-aged management units (i.e., clearcut and shelterwood methods), and three uneven-aged management units (i.e., single-tree selection and patch cuts). The management units range from 303 to 483 ha and the research cores range from 78 to 110 ha (Kalb and Mycroft, this publication: Table 1).

The HNF is located in southern Indiana, beginning approximately 20 km south of Yellowwood State Forest and extending to the Ohio River. It encompasses about 80,900 ha divided into four sections that are surrounded by other forested properties. Some sections also have open habitats such as farmland and pasture in the surrounding landscape. Historically, there have been some differences in the management of HNF and state forests (e.g., single-tree harvesting has been more prevalent in the state forests). HNF has a few more areas consisting of early successional habitat, but both regions are largely dominated by middle-aged to mature deciduous hardwood forests. Essentially, the HNF is part of a landscape similar to that of the HEE.

MATERIALS AND METHODS

Sampling

Survey points were placed in a manner to analyze breeding bird distributions and abundances at several spatial scales within the nine research cores. At the local scale, we classified points as being within a proposed harvest area (“harvest” sites), in the forest but within 100 m of a proposed harvest (“edge” sites), or greater than 100 m from a proposed harvest and

[2] HNF breeding bird survey (2006-2008) data on file with Purdue University, Department of Forestry and Natural Resources, West Lafayette, IN

within the unharvested portion of the research core ("landscape" sites). At a broader scale, we classified the points according to the planned silvicultural treatments being implemented at HEE: control units, even-aged units (i.e., clearcuts and shelterwoods), and uneven-aged units (i.e., single-tree selection and patch cuts). After placing at least one point within each area receiving a clearcut, shelterwood, or patch cut harvest, and 2-4 points within 50 m of the edge of each of those areas, we systematically distributed points on a 150m x 150m grid throughout each management unit. Any grid-based points within 100 m of a harvest edge, within 100 m of the research core boundary, or within 100 m of an existing point were ignored (see Kalb and Mycroft, this publication: Fig. 2 for an example). All survey points were witnessed by marking a nearby large tree with paint and an aluminum tag, and were located to the most accurate possible meter (usually 6-10 m) using a Garmin GPS 12 (GARMIN International, Inc., Olathe, KS) handheld global positioning system (GPS) receiver.

Each management unit contained a different number of survey points due to the variation in size of the units and the number and size of the treatment sites (Kalb and Mycroft, this publication: Table 1). There were 240 survey points across the nine management units, with an average of 26.7 points per unit (Table 1).

Table 1.—Sample sizes of avian survey points within each management unit.

Management unit	Treatment type	No. of survey points
1	uneven-aged	31
2	control	19
3	even-aged	31
4	control	25
5	control	21
6	even-aged	27
7	uneven-aged	28
8	uneven-aged	32
9	even-aged	26
Total		240

We used aural point count surveys to determine densities of individual species of breeding birds at each of the survey points throughout the management units. We employed standard point count protocol, which can be an effective means of demonstrating changes in relative abundance of forest birds (Dawson 1981, Thompson et al. 2002). These surveys were conducted annually from 20 May - 20 June from 0600 to 1100 EDT. Each season, we hired 4-6 field workers who were trained before beginning the sampling. We sent compact discs of bird songs/calls of southern Indiana birds to field workers as soon as they were hired. If the technicians lived locally, we began training them in early May. Prior to the field sampling, technicians then underwent 1 week of intensive training within the HEE sites on song identification, distance estimation, and GPS usage. Fortunately, the same two expert birders trained the staff for the first 5 years of the study, which reduced potential inconsistency in the training. As the surveys were completed, the field technician coordinator reviewed the results to identify potential errors, and conducted re-training exercises as necessary.

We sampled each point twice per season using a different field technician to help minimize observer bias. After arriving at each survey point, the technician recorded wind speed, air temperature, and sky cover. The observer allowed a minute to pass for the birds to "settle" and then conducted a 10-minute survey. During the survey, the technician recorded each species of bird, sex of bird, estimated distance from technician, and whether identification was made through song, call, and/or sight. To use mark-recapture analysis to estimate differences in detectability for each species, we also recorded the minute period of the 10-minute survey in which the bird was detected (Farnsworth et al. 2002, Moore et al. 2004).

Data Analysis

Point-count surveys are a popular method of monitoring bird populations by estimating abundance and trends in abundance (Rosenstock et al. 2002). To make reliable comparisons among data sets and

between species, raw counts should be adjusted for differences in species detectability (Farnsworth et al. 2002, Thompson and La Sorte 2008). To adjust our survey counts, we employed a capture-removal model that uses the same methodology as removal modeling of closed populations (Otis et al. 1978). Farnsworth et al. (2002) developed a method where individual covariates can be used to explain variation in detection probability by calculating maximum-likelihood estimates of model parameters.

Using a standardized point-count protocol controls for variation in seasonal and weather differences among survey periods, but other factors cannot be easily controlled (Moore et al. 2004). Therefore we added year, management unit (each unit was identified individually as a categorical variable), harvest type (local-scale - described in the Sampling section: harvest, edge, or landscape), observer rank (rank 4 = average, rank 3 = good, rank 2 = excellent, or rank 1 = expert), and binary time of day (detected before or after 0745 hours) as covariates to the analysis. We analyzed the data using a Huggins closed-capture model (Huggins 1989, 1991) in program MARK (White and Burnham 1999).

During each 10-minute survey period, the technician recorded the 1-minute time interval in which each individual was initially detected. After dividing the survey period into five 2-minute intervals (assuming equal per-minute detection probabilities within the survey), these intervals are treated as trapping occasions. Each detected individual was considered "removed" from the population (i.e., was not recorded in subsequent time periods within the same survey) after its initial detection period by setting recapture probability at 0. The most general model we tested was $M_{y+u+h+o+t}$ (y = year, u = management unit, h = harvest type, o = observer rank, t = binary time of day). Reduced models included all possible additive combinations of covariates (e.g., M_{y+h} and M_{h+o+t}) and all single-variable models. We did not separate the two sampling occasions for each survey point when building these models because we had no expectation that the sampling occasion periods would affect detectability, as all surveys were conducted within a short time period.

A unique goodness-of-fit procedure was not available for closed-population capture-recapture models in MARK, so we used a variance inflation factor, c, to assess the fit of each species' global model. The parameter c is estimated from a goodness-of-fit chi-square statistic and divided by its degrees of freedom, $\hat{c} = \chi^2/df$. The values for the models were $2.54 < c < 3.14$, which indicates that some overdispersion was present but not a structural lack of fit (Burnham and Anderson 1998). To correct for overdispersion, we conducted a second capture-recapture analysis for each species, where the $\hat{c}$ from the global or subglobal model was incorporated into calculations of best-fit models for all models in the set. This approach does not affect the values of the parameters, but it can change the ordering of the best-fit models because it favors models with fewer parameters. This procedure also inflates the standard errors of the parameters by a value of $\sqrt{\hat{c}}$ to account for the uncertainty due to overdispersion (White et al. 2001).

We used the Akaike information criterion (AIC) to compare our models. When adjustments to $\hat{c}$ are employed, the following equation is used to calculate a modified version of AIC:

$$QAIC_c = -\left[2\log\left(\mathcal{L}\left(\hat{\theta}\right)\right)/\hat{c}\right] + 2K + \frac{2K\,(K+1)}{n-K-1}$$

We found substantial model uncertainty for most species where there was no clear single model that best fit the data (Table 2). We used a weighted average of parameter estimates to conduct full-model averaging in which all subsets of the candidate set of models are included (Table 2) (Symonds and Moussalli 2011), calculated by:

$$\tilde{\bar{\beta}} = \sum_{i=1}^{R} w_i \hat{\beta}_i$$

Table 2.—Best-ranked models for detection probability estimates when individual covariates are considered. For covariates with asterisk (*), the individual covariate was found to be significant because its 95-percent confidence interval did not include zero.

Species	Model[a]	$QAIC_c$ weights	Covariate[b]	$\tilde{\beta}$	SE
Acadian flycatcher	M_y	0.35	2006	-0.53	0.30
			2007	0.09	0.16
			2008	-0.15	0.17
Brown-headed cowbird	M_t	0.56	TOD > 0745	-0.10	0.30
Carolina wren	M_t	0.57	TOD > 0745	-0.43	0.64
Cerulean warbler	M_t	0.63	TOD > 0745	-0.21	0.27
Hooded warbler	M_t	0.36	TOD > 0745	-0.02	0.14
Indigo bunting	M_h	0.34	Landscape	-0.20	0.51
			Edge	-0.03	0.37
Ovenbird	M_h	0.32	Landscape	-0.05	0.16
			Edge	-0.02	0.17
Red-eyed vireo	M_{y+o}	0.42	2006	0.34	0.23
			2007*	0.48	0.18
			2008*	-0.38	0.19
			Rank 1	-0.30	0.27
			Rank 2	0.21	0.27
			Rank 3*	-0.45	0.22
Eastern towhee	M_t	0.67	TOD > 0745	-0.56	0.41
Scarlet tanager	M_t	0.42	TOD > 0745	-0.35	0.21
Worm-eating warbler	M_{o+t}	0.32	Rank 1*	-0.65	0.26
			Rank 2	-0.17	0.25
			Rank 3	-0.30	0.22
			TOD > 0745	-0.12	0.12
Wood thrush	M_t	0.52	TOD > 0745	-0.19	0.16

[a] Models tested for effects of the following covariates on species detectability: *y* = year, *u* = management unit, *h* = harvest type, *o* = observer rank, *t* = binary time of day (TOD).

[b] See Materials and Methods section for detailed description of individual covariates.

A formula for estimating the unconditional variance of a model-averaged parameter has not yet been developed. Burnham and Anderson (1998) recommend using the following equation:

$$\widehat{var}\left(\bar{\tilde{\beta}}\right) = \sum w_i \left[\widehat{var}(\beta_i) + \left(\beta_i - \bar{\tilde{\beta}}\right)^2\right]$$

Covariate $\tilde{\beta}$ values were calculated as a logit function and then converted into real values during the probability detection calculations.

Let p_i = the probability a bird is detected in any given detection interval. We evaluated each species separately and each interval was of equal length, so we assumed equal detection probabilities for each "capture occasion" ($p_1 = p_2 = p_3 = p_4 = p_5$). After we calculated a detection probability for each time period, the combined detection probability for the entire count was calculated by $\hat{p} = 1 - (1 - p_i)^5$. Dividing the observed count by the combined detection probability ($\hat{p}$) yielded the adjusted abundance estimate (Moore et al. 2004). In addition to program MARK, we used Microsoft Excel 2010 (Microsoft Corporation, Redmond, WA) for calculations and model manipulation.

For the initial pretreatment analysis, we placed a number of species into two habitat guilds. For the mature-forest guild, we chose acadian flycatcher, cerulean warbler, ovenbird, red-eyed vireo, scarlet tanager, worm-eating warbler, and wood thrush. These species were selected because they were comparatively easy to detect with vocal, territorial males, and were present in relatively high numbers in our study area. For the early successional guild, we chose brown-headed cowbird, Carolina wren, indigo bunting, and eastern towhee. Brown-headed cowbirds were also selected because they are nest parasites and can have interspecific effects (Friesen et al. 1999). Although hooded warblers are usually considered a mature forest species, they are often associated with small gaps within mature forest. Therefore, they may respond positively to harvest treatments much like the early successional species. Across both guilds, hooded warbler, worm-eating warbler, and cerulean warbler are species of concern in Indiana.

Upon reviewing the multi-year database, we noticed an unusual pattern in the observations of worm-eating warbler and chipping sparrow. Although the combined totals of these species were similar across years, their relative detections changed annually. Specifically, we detected many more chipping sparrows than worm-eating warblers in 2007, but the reverse was true in all other years. Field technicians often had difficulties with the similar-sounding songs of these two species. We believe that in 2007 most of our technicians had the species' songs interchanged. Therefore, we combined the chipping sparrow detections and worm-eating warbler detections for each year and re-classified the combined total as worm-eating warbler. Since chipping sparrow detections were a small fraction of those for worm-eating warbler in most years (e.g., 13 sparrows, 406 warblers in 2006), we feel that our approach reduced overall error across the analysis.

Comparison with HNF

The HNF monitoring survey has the same protocol as the HEE and concentrates on mature forest birds, thereby providing a comparison to the pre-treatment HEE data.[3] Using HNF data for 2006-08, we summarized the total detections recorded in these years. HNF data have not been adjusted for detectability, and different points are surveyed in odd and even years. Thus, we compared the total raw detections across all survey points in the HNF and HEE to determine if the relative abundance rankings of breeding species were similar and if the same overall trends in total detections were apparent in both databases.

RESULTS

General

From 2006 through 2008, we observed 74 bird species, of which 62 were breeding species, and recorded 17,806 total detections, of which 17,340 were of breeding species. Each year we detected some nonbreeding birds, most commonly migrants that were still in the HEE area during the first week of the survey period. As expected, our most abundant species were birds that use mature forest habitat, whereas typical early-successional species were recorded in relatively small numbers (Table 3). The red-eyed vireo was the most abundant species with 12.9 percent of all detections, followed by worm-eating warbler (7.6 percent), acadian flycatcher (7.4 percent), eastern wood-pewee (7.3 percent), and ovenbird (6.3 percent). Other commonly detected species included tufted titmouse, wood thrush, hooded warbler, scarlet tanager, American crow, and brown-headed cowbird. Together these 11 species accounted for 67.6 percent of detections for pre-treatment years.

[3] HNF breeding bird survey (2006-2008) data on file with Purdue University, Department of Forestry and Natural Resources, West Lafayette, IN

Table 3.—Total detections for all HEE survey points during each pre-treatment sampling year.

Common name	Scientific name	2006	2007	2008
Acadian flycatcher	*Empidonax virescens*	165	550	562
American crow	*Corvus brachyrhynchos*	261	276	176
American goldfinch	*Spinus tristis*	2	3	0
American redstart	*Setophaga ruticilla*	4	5	4
American robin	*Turdus migratorius*	5	1	3
Barred owl	*Strix varia*	7	0	0
Black-and-white warbler	*Mniotilta varia*	11	1	165
Blackpoll warbler	*Setophaga striata*	0	0	33
Black-throated green warbler	*Setophaga virens*	5	13	16
Blue jay	*Cyanocitta cristata*	155	271	161
Blue-gray gnatcatcher	*Polioptila caerulea*	42	130	229
Blue-headed vireo	*Vireo solitarius*	0	0	1
Blue-winged warbler	*Vermivora cyanoptera*	1	0	0
Broad-winged hawk	*Buteo platypterus*	0	5	6
Brown-headed cowbird	*Molothrus ater*	81	305	256
Canada goose	*Branta canadensis*	0	8	8
Carolina chickadee	*Poecile carolinensis*	102	60	89
Carolina wren	*Thryothorus ludovicianus*	4	5	14
Cedar waxwing	*Bombycilla cedrorum*	37	18	36
Cerulean warbler	*Setophaga cerulea*	15	55	141
Chestnut-sided warbler	*Setophaga pensylvanica*	3	2	0
Chimney swift	*Chaetura pelagica*	0	4	11
Chipping sparrow[a]	*Spizella passerina*	0	0	0
Common grackle	*Quiscalus quiscula*	1	0	0
Common yellowthroat	*Geothlypis trichas*	2	0	0
Cooper's hawk	*Accipiter cooperii*	0	0	1
Downy woodpecker	*Picoides pubescens*	37	41	17
Eastern phoebe	*Sayornis phoebe*	6	9	2
Eastern screech-owl	*Megascops asio*	0	0	1
Eastern towhee	*Pipilo erythrophthalmus*	0	44	33
Eastern wood-pewee	*Contopus virens*	440	412	413
Field sparrow	*Spizella pusilla*	0	0	1
Gray catbird	*Dumetella carolinensis*	8	8	6
Great crested flycatcher	*Myiarchus crinitus*	28	42	11
Great horned owl	*Bubo virginianus*	1	0	1
Hairy woodpecker	*Picoides villosus*	0	1	12
Hooded warbler	*Setophaga citrina*	232	243	273
Indigo bunting	*Passerina cyanea*	27	13	29
Kentucky warbler	*Geothlypis formosa*	20	41	102
Louisiana waterthrush	*Parkesia motacilla*	41	30	22
Magnolia warbler	*Setophaga magnolia*	1	0	2
Mourning dove	*Zenaida macroura*	4	24	4
Northern cardinal	*Cardinalis cardinalis*	256	184	112
Northern flicker	*Colaptes auritus*	10	19	47
Northern mockingbird	*Mimus polyglottos*	1	0	0
Northern parula	*Setophaga americana*	105	34	33
Northern waterthrush	*Parkesia noveboracensis*	1	0	3
Ovenbird	*Seiurus aurocapilla*	298	376	413
Pileated woodpecker	*Dryocopus pileatus*	60	97	82
Pine warbler	*Setophaga pinus*	12	16	12
Red-bellied woodpecker	*Melanerpes carolinus*	258	136	172
Red-eyed vireo	*Vireo olivaceus*	622	797	821
Red-headed woodpecker	*Melanerpes erythrocephalus*	30	3	5
Red-shouldered hawk	*Buteo lineatus*	0	2	0
Red-tailed hawk	*Buteo jamaicensis*	4	4	4

(Table 3 continued on next page)

Table 3 (continued).—Total detections for all HEE survey points during each pre-treatment sampling year.

Common name	Scientific name	2006	2007	2008
Rose-breasted grosbeak	*Pheucticus ludovicianus*	0	2	3
Ruby-throated hummingbird	*Archilochus colibris*	12	34	3
Scarlet tanager	*Piranga olivacea*	79	260	380
Summer tanager	*Piranga rubra*	2	5	12
Swainson's thrush	*Catharus ustulatus*	0	2	15
Tennessee warbler	*Oreothlypis peregrina*	3	50	176
Tree swallow	*Tachycineta bicolor*	2	0	0
Tufted titmouse	*Baeolophus bicolor*	110	296	451
Warbling vireo	*Vireo gilvus*	0	4	0
Whip-poor-will	*Caprimulgus vociferus*	0	0	4
White-breasted nuthatch	*Sitta carolinensis*	335	96	157
White-eyed vireo	*Vireo griseus*	1	6	0
Wild turkey	*Meleagris gallopavo*	21	6	16
Wood thrush	*Hylocichla mustelina*	272	242	339
Worm-eating warbler[a]	*Helmitheros vermivorum*	419	506	402
Yellow warbler	*Setophaga petechia*	0	137	3
Yellow-billed cuckoo	*Coccyzus americanus*	1	187	29
Yellow-throated vireo	*Vireo flavifrons*	163	88	101
Yellow-throated warbler	*Setophaga dominica*	8	31	97
Total		4,833	6,240	6,733

[a] Chipping Sparrow and Worm-eating Warbler have been combined and placed under Worm-eating Warbler. See Materials and Methods section for explanation.

Many of our species of interest exhibited an increase in detections throughout the study period (Table 4). Notable changes from 2006 to 2007 included a substantial increase in detections of acadian flycatcher, brown-headed cowbird, scarlet tanager, cerulean warbler, and eastern towhee. Besides another marked increase in scarlet tanager, wood thrush, and cerulean warbler, most of these species exhibited minor fluctuations from 2007 to 2008.

Table 4.—Unadjusted detections and relative abundances for focal species at all HEE survey points.

Species	Detections (relative abundance) 2006	2007	2008	Total
Red-eyed vireo	622 (0.13)	797 (0.13)	821 (0.13)	2,240 (0.13)
Worm-eating warbler[a]	419 (0.09)	506 (0.08)	402 (0.06)	1,327 (0.08)
Acadian flycatcher	165 (0.03)	550 (0.09)	562 (0.09)	1,277 (0.07)
Ovenbird	298 (0.06)	376 (0.06)	413 (0.06)	1,087 (0.06)
Wood thrush	272 (0.06)	242 (0.04)	339 (0.05)	853 (0.05)
Hooded warbler	232 (0.05)	243 (0.04)	273 (0.04)	748 (0.04)
Scarlet tanager	79 (0.02)	260 (0.04)	380 (0.06)	719 (0.04)
Brown-headed cowbird	81 (0.02)	305 (0.05)	256 (0.04)	642 (0.04)
Cerulean warbler	15 (0.00)	55 (0.01)	141 (0.02)	211 (0.01)
Eastern towhee	0 (0.00)	44 (0.01)	33 (0.01)	77 (0.00)
Indigo bunting	27 (0.01)	13 (0.00)	29 (0.00)	69 (0.00)
Carolina wren	4 (0.00)	5 (0.00)	14 (0.00)	23 (0.00)
Total breeding bird detections	4,823	6,037	6,480	17,340

[a] Chipping sparrow and worm-eating warbler have been combined and placed under worm-eating warbler. See Methods section for explanation.

We estimated the mean detections per point for each proposed management type using the maximum number of individuals detected over the two point visits (Table 5). Individual species showed little variation in abundance among the management types.

Detection Models

The covariates that affected detectability and their associated weights differed for each species (Table 2). Across all species, the binary time of day was the covariate that was most consistently included in the best models. Even when the time of day was not present in the best model, it was usually contained in the next-best model. Year was contained in the best model for some species (e.g., acadian flycatcher and red-eyed vireo), but for other species (e.g., eastern towhee and worm-eating warbler) it did not appear within the best 90 percent of models. Harvest type was usually found within the top 80 percent of models. Observer rank had a relatively small impact on detectability, and management unit was contained only once within the models contributing over 99 percent of the parameter weights.

A model-averaged covariate significantly contributed to detectability estimates if its 95-percent confidence interval did not include zero. Significant covariates within the red-eyed vireo models were the years 2007 and 2008, and observer rank 3 (see Methods: Data Analysis for definitions of observer rank). Worm-eating warbler models contained observer rank 1 as a significant covariate (Table 2). Most of the covariate values were very small and/or had large standard errors. Overall, the individual coefficients had a minimal effect on the estimates of detection probability.

Our method of model averaging yielded a unique probability of detection ($\hat{p}$) for each combination of covariates. Most of these values were close to 1.0 due to the small covariate estimates, thus resulting in relatively small changes in relative abundance (Table 6). The average detection probabilities for individual species ranged from 0.67 to 0.96. In general, detectability was positively correlated with abundance estimates. Most species showed a relative abundance change of less than 0.5 percent. The exception was Brown-headed Cowbird, whose relative abundance increased by 1.56 percent (mean $\hat{p} = 0.67$) when adjusted for detectability, which is still a small change.

Table 5.—Mean maximum abundance (of the two visits) estimated for focal species after being adjusted for differences in detectability. Values represent the expected maximum abundance/point (with standard deviation) over the complete survey period.

Species	Proposed management type		
	Control	Even-aged	Uneven-aged
Red-eyed vireo	2.20 (0.33)	2.22 (0.41)	2.22 (0.63)
Worm-eating warbler	1.34 (0.24)	1.40 (0.18)	1.36 (0.22)
Acadian flycatcher	1.22 (0.46)	1.20 (0.43)	1.36 (0.57)
Brown-headed cowbird	1.19 (0.76)	1.26 (0.64)	0.99 (0.57)
Ovenbird	0.92 (0.52)	1.13 (0.55)	1.37 (0.38)
Scarlet tanager	0.94 (0.46)	0.93 (0.49)	0.90 (0.60)
Wood thrush	0.78 (0.18)	1.10 (0.66)	0.87 (0.39)
Hooded warbler	0.79 (0.28)	0.87 (0.28)	0.99 (0.15)
Cerulean warbler	0.23 (0.21)	0.27 (0.22)	0.27 (0.48)
Eastern towhee	0.13 (0.16)	0.11 (0.14)	0.12 (0.14)
Indigo bunting	0.08 (0.06)	0.11 (0.07)	0.12 (0.13)
Carolina wren	0.01 (0.02)	0.05 (0.05)	0.06 (0.10)

Table 6.—Changes in count totals and relative abundances (R.A.) of focal species between raw detections and detections adjusted for variations in detectability. Probability of detection values = mean (minimum, maximum).

	Pre-adjustment			Post-adjustment		
Species	Total	R.A. (%)	Probability of detection	Total	R.A. (%)	Change in R.A.
Red-eyed vireo	2,240	12.92	0.95 (0.83, 1.00)	2,353.11	12.92	0.00
Worm-eating warbler	1,327	7.65	0.96 (0.88, 0.99)	1,382.34	7.59	-0.06
Acadian flycatcher	1,277	7.36	0.96 (0.87, 0.98)	1,335.67	7.33	-0.03
Ovenbird	1,087	6.27	0.96 (0.95, 0.97)	1,135.59	6.23	-0.04
Wood thrush	853	4.92	0.95 (0.94, 0.97)	895.10	4.91	-0.01
Hooded warbler	748	4.31	0.92 (0.89, 0.93)	816.75	4.48	0.17
Scarlet tanager	719	4.15	0.86 (0.79, 0.93)	837.18	4.60	0.45
Brown-headed cowbird	642	3.70	0.67 (0.60, 0.72)	957.93	5.26	1.56
Cerulean warbler	211	1.22	0.91 (0.89, 0.96)	231.22	1.27	0.05
Eastern towhee	77	0.44	0.85 (0.79, 0.95)	90.90	0.50	0.05
Indigo bunting	69	0.40	0.83 (0.76, 0.89)	83.20	0.46	0.06
Carolina wren	23	0.13	0.75 (0.67, 0.86)	31.04	0.17	0.04
Total breeding bird detections	17,340			18,217		

HEE and HNF Comparisons

The HEE and HNF surveys resulted in similar species lists and abundance rankings in the comparison period of 2006-08 (Table 7). We examined the abundance rankings of the 10 most common species in the HEE database, plus the brown-headed cowbird, which is of particular management interest and was ranked in the HEE database as the 11th most common species. Though not in identical order, the most numerous 11 species in the HEE during 2006-08 matched the most numerous species in HNF with two exceptions: brown-headed cowbird and hooded warbler ranked 15th and 16th in abundance in HNF, rather than within the top 11 (Table 7). These top 11 species differed in relative abundance between the two study areas by less than 2 percent except for red-eyed vireo and worm-eating warbler, which were higher in HEE by a difference of 6.1 percent and 4.0 percent, respectively.

We saw an increase in total detections in the HEE database each year with a substantial increase from 2006 to 2007, and a smaller increase from 2007 to

Table 7.—Comparative rankings of most abundant species in the HEE and HNF database, 2006-08. Order of species is given in column 1 based on rank order of HEE species across all 3 years.

	HEE				HNF			
Species	Total	2006	2007	2008	Total	2006	2007	2008
Red-eyed vireo	1	1	1	1	3	5	7	1
Worm-eating warbler	2	3	3	6	10	9	9	16
Acadian flycatcher	3	11	2	2	4	10	2	4
Eastern wood-pewee	4	2	4	4	6	2	4	6
Ovenbird	5	5	5	5	1	1	5	2
Tufted titmouse	6	14	7	3	7	6	12	7
Wood thrush	7	6	12	8	2	3	3	3
Hooded warbler	8	10	11	9	16	18	17	10
Scarlet tanager	9	18	10	7	9	11	10	9
American crow	10	7	8	12	5	4	1	8
Brown-headed cowbird	11	17	6	10	15	19	18	5

2008 (Table 1). The increase between the first 2 years was reflected in the HNF survey as well. Total detections per point (summarizing both visits and comparing 2006 to 2007) increased from 20.1 to 26.0 and from 20.8 to 26.1 for the HEE and HNF, respectively. The small increase in detections from 26.0 to 28.1 in the HEE database during the final 2 years (2007 compared to 2008) was not matched in the HNF database, in which total detections decreased slightly to 26.1 in 2008.

DISCUSSION

The increase in total detections each year in the HEE database is somewhat puzzling as there were no weather-related or logistic issues that could account for the increase. Field technicians were trained in the same way across years, and the increased numbers were not associated with the records of particular individuals. It was therefore of interest to find the same increase in total detections in the HNF database, especially from 2006-07. The HNF data were collected by different observers trained with the same overall techniques.

Because we employed full model averaging, all covariates contributed something to the adjusted detection probability. The amount that the covariate contributes to the average shrinks toward zero by the degree to which it is uninformative. Models that do not contain the covariates of interest simply contribute zero to the average. It is not surprising that the binary time of day was the largest contributor because most birds sing more frequently in the early morning, making them easier to detect during the earlier period.

Considering that our data were collected prior to any harvests, it is noteworthy that the proposed harvest type had an effect on detectability. Saunders and Arseneault (this publication) found differences in forest structure within the study area, so we will examine the possibility of those effects in future analyses. Management unit appeared to have no substantial effect on detectability. We were expecting the observer rank to have more of an effect on detectability than the results showed. However, it is encouraging that observer differences only marginally affected detections, because the HEE will have a great variety of observers in the future.

Adjustments for detectability were minor due to a combination of small β values for covariate coefficients and the resulting large detection probabilities. Red-eyed vireo and worm-eating warbler were the only species whose models contained covariates that significantly affected detection probabilities. The largest change in relative abundance occurred in brown-headed cowbird, which is reasonable since its calls are challenging to some technicians.

It is encouraging to discover that the HEE sites support substantial populations of Indiana species of concern, and this finding stresses the importance of the study area as habitat for these species. Hooded warbler and worm-eating warbler had higher relative abundances in the HEE than in the HNF sites (Table 7). The highest record in the 2006-08 study period for cerulean warbler was 141 detections (2.2 percent relative abundance), whereas the highest record in HNF was 16 detections (0.3 percent relative abundance).

We also detected some rarer species within the HEE sites that could be breeding in our research area. These species included American redstart with 4-5 detections per year, and black-throated green warbler with 5-16 detections per year. We detected black-and-white warbler 11 times in 2006 and once in 2007, but saw an increase to 165 detections in 2008. There was a higher frequency of black-and-white warbler detections during the beginning of the field season, which can implicate them as migrants, but a strong presence until the end of the season supports the theory that some stayed to breed in the area. The high numbers of black-and-white warblers in the region in 2008 were consistent with anecdotal observations

in the area that summer.[4] Broad-winged hawk (five detections in 2007, six in 2008) and Cooper's hawk (one detection in 2008) were also rare species to find in a mature forest.

We had a few data sets that caused problems when fitting appropriate models. Maximum likelihood estimation would fail to converge, or convergence was successful but with parameter estimates that were unrealistic. These results occurred with Carolina wren, indigo bunting, and eastern towhee, which are early-successional species with relatively sparse observations. In each case, there were still some valid results, so we used those models to calculate the averages.

CONCLUSION

The similarities between the HEE and HNF results give us confidence that our surveys during the pre-treatment years of the HEE project adequately quantified the bird species common in mature Indiana forests and present on the HEE sites. Individual management units and proposed harvest type were not found to be significant predictors of detectability. Observer rank and year were significant predictors for two species. There was few differences among pre-treatment management types based on the mean detections/point for each species. We are therefore confident that the HEE pre-treatment data will be a suitable basis for comparison with those of post-harvest years.

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[4] Personal communication with J.K. Riegel, Purdue University

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RELATIVE ABUNDANCE AND SPECIES RICHNESS OF TERRESTRIAL SALAMANDERS ON HARDWOOD ECOSYSTEM EXPERIMENT SITES BEFORE HARVESTING

Jami E. MacNeil and Rod N. Williams[1]

Abstract.—Terrestrial salamanders are ideal indicators of forest ecosystem integrity due to their abundance, their role in nutrient cycling, and their sensitivity to environmental change. To understand better how terrestrial salamanders are affected by forest management practices, we monitored species diversity and abundance before implementation of timber harvests within the forested landscape of the Hardwood Ecosystem Experiment in Indiana. We monitored 66 cover-object grids in fall 2007 and spring 2008 and conducted quadrat surveys at each grid in spring 2008. Cover-object sampling and quadrat surveys detected six salamander species. The most commonly encountered species were eastern red-backed (*Plethodon cinereus*) (n=3621 encounters) and northern zigzag salamanders (*P. dorsalis*) (n=1603 encounters). Mean salamander encounters per sampling occasion found by cover objects ranged among units from 6.6 to 11.1, whereas those found by quadrat surveys ranged from 1.5 to 7.3. Treatment types did not differ according to cover-object data, but quadrat surveys found greater mean encounter rates in control and even-aged units than in uneven-aged units. Encounter rates were greater during spring sampling compared to fall, and rates were greater on northeast-facing slopes in general.

INTRODUCTION

Terrestrial salamanders (family Plethodontidae) serve as excellent indicators of forest ecosystem health due to their important function in the ecosystem and their sensitivity to changes in the environment (Welsh and Droege 2001). These species are abundant vertebrates in eastern U.S. forests, occurring in densities as high as two per square meter (Jaeger 1980, Petranka 1998) and making up a large proportion of the biomass in forest habitats (Burton and Likens 1975). As top predators in the soil, terrestrial salamanders facilitate nutrient cycling by preying on small invertebrates and being consumed by larger predators (Davic and Welsh 2004). Lacking lungs, plethodontid salamanders require cool, moist microhabitats to facilitate cutaneous respiration (Petranka 1998). Furthermore, most members of Plethodontidae are terrestrial breeders with small territories, suggesting a limited ability to disperse in the wake of a disturbance (Kleeberger and Werner 1982, Welsh and Droege 2001). The high abundance of terrestrial salamanders, their role in nutrient cycling, their sensitivity to desiccation, and their limited dispersal ability make them a useful group to monitor before and after forest disturbance, because negative effects on salamanders could reflect negative effects on the wider ecosystem.

Previous studies on the effects of timber harvest techniques on salamanders have produced conflicting results and differ widely in duration, scale, region, and technique (deMaynadier and Hunter 1995). Many studies have found higher abundance of salamanders

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in mature forest stands compared to recent clearcuts (Grialou et al. 2000, Herbeck and Larsen 1999, Petranka et al. 1993), but Renken et al. (2004) found no differences in abundance among 13 herpetofaunal species up to 3 years post-harvest. Researchers disagree on the time it takes salamander populations to recover following clearcutting; estimates range from 15-20 years (Duguay and Wood 2002) to 50-60 years (Ford et al. 2002, Petranka et al. 1993, Pough et al. 1987).

The effects of other timber harvest techniques such as group selection cuts and shelterwoods are less clear than those of clearcuts. Some research indicates that negative effects may be mitigated by basal area left on site (Harpole and Haas 1999, Knapp et al. 2003, Ross et al. 2000). Other studies suggest effects may be short-lived but that repeated entries of multi-stage harvests could interfere with salamander population recovery (Morneault et al. 2004). Given the wide variation in study design and the sometimes conflicting conclusions of past research, further investigation is needed to understand how timber harvests affect terrestrial salamanders.

We examine the effects of even-aged and uneven-aged forest management on terrestrial salamanders within the context of the Hardwood Ecosystem Experiment (HEE). Objectives are to:

1. Compare salamander diversity and relative abundance across all treatment areas before harvesting,
2. Resample across treatment areas after timber harvests to examine effects of forest management on salamander diversity and relative abundance,
3. Evaluate salamander detection using multiple sampling techniques, and
4. Determine how salamander relative abundance and diversity are affected by proximity to a forest clearcut edge.

This paper describes the methods and results of pre-harvest data collection (objective 1), which took place in fall 2007 and spring 2008.

STUDY AREA

The study took place in Morgan-Monroe State Forest (MMSF) and Yellowwood State Forest (YSF) in Morgan, Monroe, and Brown Counties in south-central Indiana. The area is characterized by a mixture of oak-hickory (*Quercus-Carya*) and American beech-maple (*Fagus grandifolia-Acer*) forest across steep ridges and valleys. The study area and the HEE study design are described in detail by Kalb and Mycroft (this publication).

MATERIALS AND METHODS

Sampling

In May 2007, we established 66 cover-object grids throughout the 9 HEE study areas. Cover objects were 30cm x 30cm x 5cm untreated pine boards. Grids consisted of 30 objects arranged in a 6 x 5 array with 5-m spacing between each object (Fig. 1). Each object was placed in direct contact with the soil, with leaf litter and debris scraped away. Each of the three control units received two randomly placed grids. The three uneven-aged management units received eight grids each, one inside each of the eight areas designated for harvest openings (see Kalb and Mycroft, this publication: Fig. 5). The 3 even-aged management units received 12 grids each, 2 inside and 1 outside (at least 40 m from edge) of each of the 4 areas designated for harvest (see Kalb and Mycroft, this publication: Fig. 2).

Grids were placed approximately mid-slope and, to the extent possible, each management type received an equal number of grids on northeast- and southwest-facing slopes. North- and east-facing slopes tend to receive less solar energy and retain more precipitation than south- and west-facing slopes (Chen et al. 1999); thus, we tried to equalize sampling effort between these two extremes. Daytime searches of cover-object grids were conducted five times at 2-week intervals during September-November 2007 and four times at 2-week intervals during March-April 2008, before the implementation of harvests. Observers recorded the

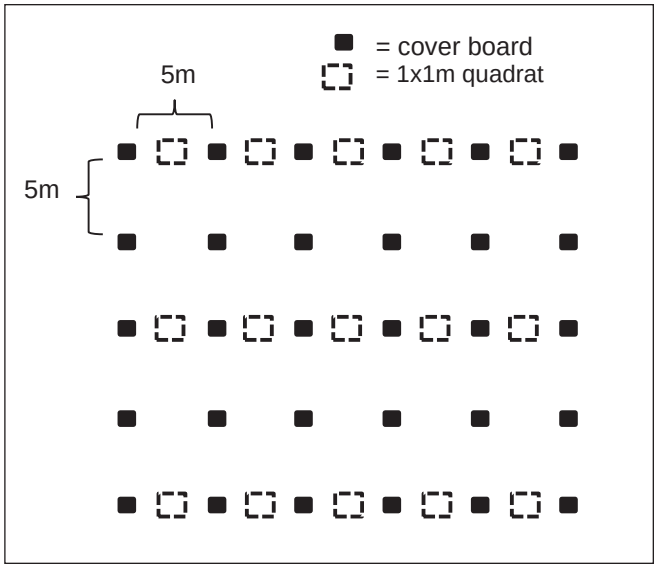


Figure 1.—Diagram of cover-object grid and quadrat placement. Cover objects were 30cm x 30cm x 5cm untreated pine boards spaced 5 m apart and placed in contact with the soil. Quadrats were 1m x 1m squares placed systematically between cover objects. During the first round of quadrat surveys in spring 2008, quadrats were placed on rows 1, 3, and 5 as shown here; during the second search they were placed on rows 2 and 4 and 5 m below row 5.

species and age class (adult snout-vent length [SVL] ≥34 mm; juvenile SVL ≤33 mm) (Petranka 1998) of salamanders found under objects.

Detection of terrestrial salamanders by any one sampling technique is imperfect because salamander surface activity varies with rainfall, soil moisture, and temperature (Hyde and Simons 2001, Jaeger 1980, Williams and Berkson 2004). The use of multiple sampling techniques may improve estimates of relative abundance (deMaynadier and Hunter 1995, Ryan et al. 2002). Thus, we employed the additional sampling technique of daytime quadrat searches. At each grid, observers carefully sifted through leaf litter and debris inside fifteen 1m x 1m quadrats placed between cover objects (Fig. 1), recording the species and age class of all salamanders encountered. Each grid received 2 such searches during March-April 2008, for a total of 1,980 quadrat searches. During the second round of quadrat sampling, the 15 plots were shifted downslope 5 m to avoid repeated disturbance of the same square-meter plots.

Habitat Characteristics

Detection of salamanders by artificial cover objects may vary depending on the amount of natural cover available (Hyde and Simons 2001). To determine if sites varied in natural cover prior to harvests, we measured volume of down woody debris (d.w.d.) using a line-intercept method (Kaiser 1983) at each cover-object grid in spring 2008. Observers walked two 20-m linear transects, 5 m upslope and 5 m downslope of the grid, and recorded the diameter of each piece of d.w.d. ≥10 cm at the point of intersection. Volume was calculated as described by Van Wagner (1968).

Precipitation and soil moisture may influence salamander surface activity; therefore, we obtained records of daily precipitation from National Oceanic and Atmospheric Administration cooperative stations (Martinsville 2SW, ID#125407 for Units 1-4; Nashville 2NNE, ID#126056 for Units 5-9) and determined the amount of precipitation that occurred during the 48 hours before each sampling occasion. We sampled soil moisture each time quadrat surveys were conducted in spring 2008 by taking five samples, one at each of the four corners and one at the center of each grid, with a soil probe to a depth of 10 cm. Soil samples were weighed wet, dried in an incubator at 40 °C for 5 days, and weighed dry to determine percentage moisture ([1 – (dry weight/wet weight)] x 100 percent). We averaged the values from the five samples taken at each grid to find mean percentage soil moisture.

Data Analysis

We standardized captures by sampling effort and present them as mean encounters per sampling occasion, where one sampling occasion is one check of a single grid. In determining the number of sampling occasions, we adjusted for missing or disturbed objects by subtracting the total proportion of disturbed grids from the total number of sampling occasions. Sample sizes (number of grids) differed by management type (e.g., unit 4 had 2 grids; unit 3 had 12 grids), so relatively high or low mean encounters presented herein may not reflect statistically significant trends.

In comparisons of slope aspects, grids were categorized as either northeast- or southwest-facing based on an azimuth taken in the middle of the top row of objects. Grids with an azimuth between 345° and 105° were categorized as "northeast" (N = 24); grids with an azimuth between 165° and 285° were categorized as "southwest" (N = 20). These categories include the compass range from north to east (0-90°) and from south to west (180°-270°), plus a buffer of 15° on either side. These wide ranges, though not strictly northeast and southwest, allowed us to include a similar number of grids within each category while excluding grids that more directly faced northwest and southeast. We excluded these "intermediate" (i.e., northwest and southeast) grids in slope comparisons because we expected they would mask slope effects between northeast and southwest slopes. This method resulted in 44 grids being included and 22 grids being omitted from slope comparisons.

RESULTS

The results presented in this paper are chiefly totals and averages. Detailed statistical comparisons of pre- and post-harvest data are presented elsewhere (MacNeil 2011).

During fall 2007-spring 2008, we encountered 5,092 salamanders of 5 species at the 66 cover-object grids (Table 1). Two species, eastern red-backed salamander (*Plethodon cinereus* Green), and northern zigzag salamander (*P. dorsalis* Cope), were encountered on every site in both seasons and collectively accounted for 89 and 97 percent of all encounters in fall 2007 and spring 2008, respectively. Mean encounters under cover objects for all species pooled ranged among units from 6.6 to 11.1 per sampling occasion (Table 1).

During spring 2008, we encountered 464 salamanders of 4 species in quadrat surveys (Table 2). Red-backed and zigzag salamanders were detected in all units and made up 98 percent of all encounters. Quadrat surveys did not detect southern two-lined (*Eurycea cirrigera* Green) or spotted salamanders (*Amybystoma maculatum* Shaw) but did detect eastern newts (*Notophthalmus viridescens* Rafinesque), which were not found under cover objects during the pre-harvest period. Mean encounters from quadrat surveys were much lower than those found by cover objects, ranging among units from 1.5 to 7.3 per sampling occasion (Table 2).

Table 1.—Total salamander encounters under wood cover objects at HEE units 1-9 during the pre-treatment period (fall 2007-spring 2008). Values in parentheses are mean encounters per sampling occasion.[a]

	Species[b]					
HEE unit	REBA	ZIZA	NOSL	SOTW	SPSA	Total
1	377 (5.2)	161 (2.2)	36 (0.5)	1 (0.0)	0	575 (8.0)
2	126 (7.0)	26 (1.4)	13 (0.7)	0	0	165 (9.2)
3	762 (7.1)	248 (2.3)	68 (0.6)	15 (0.1)	0	1,093 (10.1)
4	125 (6.9)	64 (3.6)	11 (0.6)	0	0	200 (11.1)
5	114 (6.3)	52 (2.9)	1 (0.1)	2 (0.1)	0	169 (9.4)
6	530 (4.9)	314 (2.9)	48 (0.4)	0	0	892 (8.3)
7	463 (6.4)	187 (2.6)	8 (0.1)	13 (0.2)	0	671 (9.3)
8	385 (5.3)	189 (2.6)	17 (0.2)	43 (0.6)	0	634 (8.8)
9	438 (4.2)	209 (2.0)	17 (0.2)	27 (0.3)	2 (0.0)	693 (6.6)
Total	3,320	1,450	219	101	2	5,092

[a] Sampling occasions are the product of the number of grids and the number of times grids were sampled within each unit. Numbers of sampling occasions corrected for missing and disturbed objects were as follows: 72 for units 1, 7, and 8; 18 for units 2, 4, and 5; 107.8 for unit 3; 108 for unit 6, and 104.7 for unit 9.

[b] REBA=eastern red-backed salamander; ZIZA=northern zigzag salamander; NOSL=northern slimy salamander; SOTW=southern two-lined salamander; SPSA=spotted salamander

Table 2.—Total salamander encounters in quadrat surveys at HEE units 1-9 during the pre-harvest period (spring 2008). Values in parentheses are mean encounters per sampling occasion.[a] Abbreviations and methods of computation follow Table 1.

	Species				
HEE unit	REBA	ZIZA	NOSL	NEWT[b]	Total
1	28 (1.8)	6 (0.4)	2 (0.1)	0	36 (2.3)
2	13 (3.3)	2 (0.5)	0	0	15 (3.8)
3	70 (2.9)	39 (1.6)	2 (0.1)	1 (0.0)	112 (4.7)
4	15 (3.8)	14 (3.5)	0	0	29 (7.3)
5	9 (2.3)	4 (1.0)	0	0	13 (3.3)
6	69 (2.9)	51 (2.1)	5 (0.2)	0	125 (5.2)
7	29 (1.8)	4 (0.3)	0	0	33 (2.1)
8	17 (1.1)	7 (0.4)	0	0	24 (1.5)
9	51 (2.1)	26 (1.1)	0	0	77 (3.2)
Total	301	153	9	1	464

[a] Numbers of sampling occasions were as follows: 16 for units 1, 7, and 8; 4 for units 2, 4, and 5; 24 for units 3, 6, and 9.
[b] NEWT=eastern newt (red eft)

We pooled encounters in the nine units by treatment to determine if salamander populations differed among treatment types before timber harvests. Data from cover objects showed few differences across treatment in mean encounters of all species (both individually and pooled; Fig. 2a). Quadrat surveys found higher encounter rates of all species pooled in control and even-aged units compared to uneven-aged units. This difference was driven largely by lower encounters of zigzag salamanders in uneven-aged units, although encounters of red-backed salamanders were also lower in uneven-aged units compared to the other treatment types (Fig. 2b).

Season affected salamander encounter rates under cover objects. For all species pooled within each unit, mean encounters under cover objects were greater in spring 2008 than in fall 2007 (Fig. 3). When considered individually, however, slimy (*P. glutinosus* Green) and southern two-lined salamanders did not follow this trend; mean encounters of these species were slightly lower in spring 2008 than in fall 2007.

Encounter rates also were affected by an interaction of season and slope aspect. In fall 2007, cover objects on northeast-facing slopes yielded higher mean encounter rates for all species pooled compared to those on southwest-facing slopes (Fig. 4). In spring

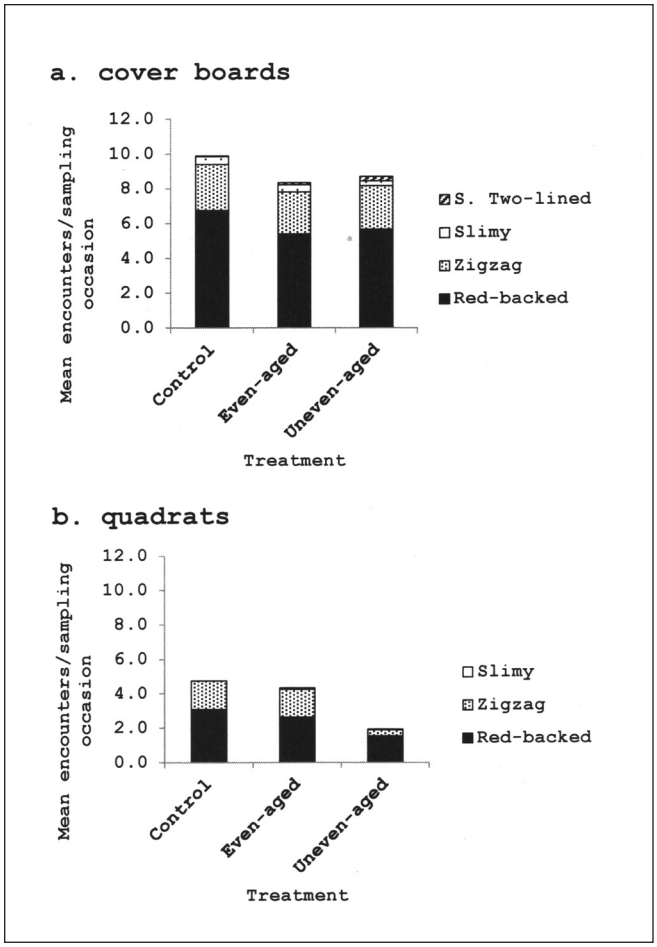


Figure 2.—Mean salamander encounters per sampling occasion found in a) cover-object grids and b) quadrat surveys in control (N=6), even-aged (N=36), and uneven-aged (N=24) HEE management units during the pre-harvest period (fall 2007-spring 2008).

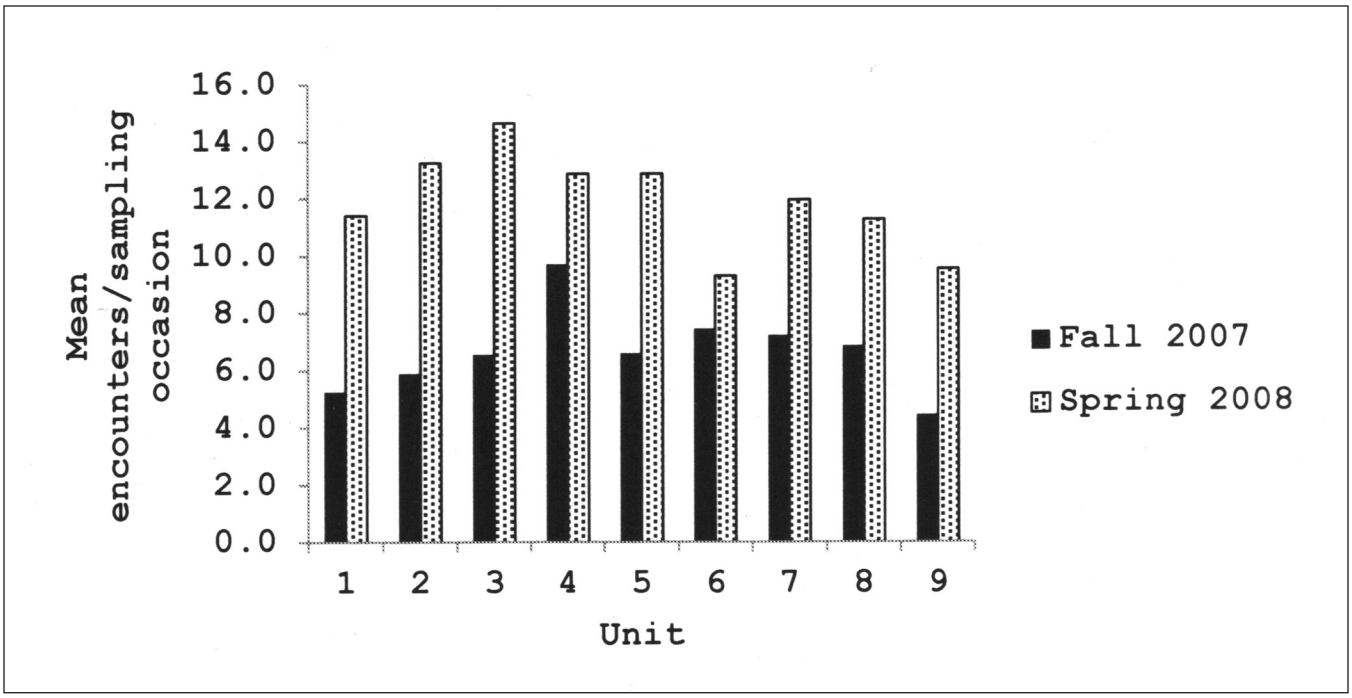


Figure 3.—Mean salamander encounters (all species pooled) per sampling occasion under cover objects in HEE sites 1-9 by season (pre-harvest). Units 2, 4, and 5 are controls; units 1, 7, and 8 are designated for uneven-aged management; units 3, 6, and 9 are designated for even-aged management.

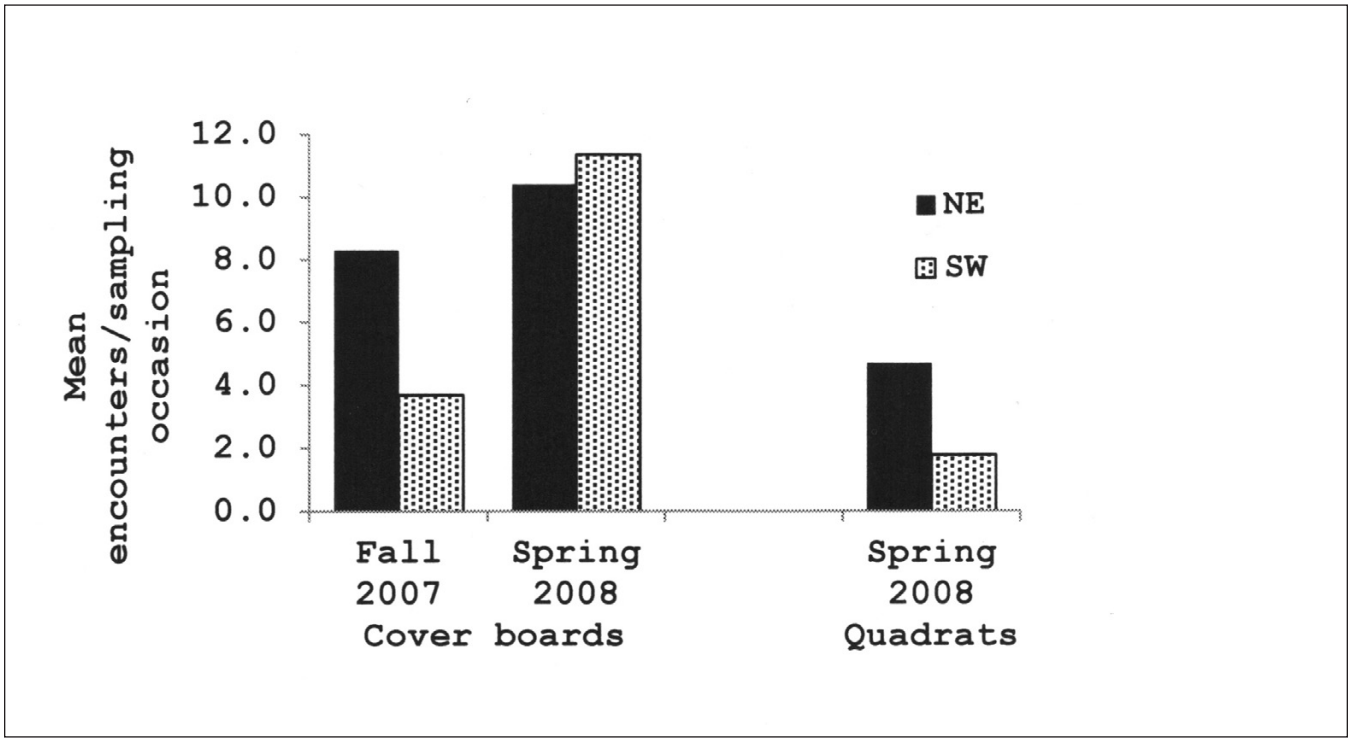


Figure 4.—Mean salamander encounters (all species pooled) per sampling occasion under cover objects and in quadrat surveys on northeast (NE, azimuth between 345° and 105°; N=24) and southwest (SW, azimuth between 165° and 285°; N=20) slopes in HEE sites in fall 2007 and spring 2008. Quadrat surveys were not conducted in fall 2007.

2008, this trend reversed such that the pooled mean encounters were slightly greater on southwest-facing slopes. Quadrat surveys on northeast-facing slopes yielded higher mean encounters of all species pooled compared to quadrat surveys on southwest-facing slopes (Fig. 4). Thus, the slope effect found by quadrats in spring 2008 resembled that found by cover objects in fall 2007, rather than that found by cover objects in spring 2008.

SUMMARY AND FUTURE WORK

Cover-object arrays detected five salamander species and quadrat surveys detected four, for a total of six unique salamander species detected in upland forest habitat on the HEE sites during the pre-harvest period. This total is 60 percent of the salamander species that use woody debris in upland, hardwood forests in south-central Indiana (Minton 2001, Williams et al. 2006). Encounter rates under cover objects did not vary greatly by unit or treatment. Quadrat surveys had lower encounter rates than cover objects, and units and treatments may differ according to the quadrat method, particularly with control and even-aged units having higher encounter rates compared to uneven-aged units.

Salamander encounters were greater in spring 2008 than fall 2007. In general, northeast slopes had greater encounter rates than southwest slopes for both cover-object sampling in fall 2007 and quadrat surveys in spring 2008. However, encounter rates were similar regardless of slope type for cover-object sampling in spring 2008, indicating a slope-by-season interaction for cover-object data.

Sampling continued in September-November 2008 on all cover-object grids not made inaccessible by logging activities, and continued on all grids each March-May and September-November from 2009 through 2011. Eighteen cover-object grids were added to control units in July 2009 to increase the total sample size in control units from 6 to 24. Quadrat surveys were conducted again in spring 2009 but were then discontinued due to low capture success. Down woody debris was measured as described above during each spring since harvests were implemented. Post-harvest sampling also involved the addition of 864 cover objects at 6 clearcuts to study the effects of silvicultural edges on terrestrial salamanders. This local-scale study included mass and SVL measurements of individual salamanders, as well as fine-scale habitat measurements such as canopy cover and leaf litter depth. Analysis methods for pre-harvest data are described fully by MacNeil (2011). To describe the analysis briefly, we used a mixed model analysis of variance in SAS 9.2 (SAS Institute Inc., Cary, NC) with fixed factors including unit, treatment, season, slope, and a random factor of grid nested in unit, with check as a repeated measure. "Check" represents a round of sampling (each 2-week period in which each grid was sampled once) and serves as a surrogate for time within a sampling season. Covariates include volume of d.w.d., amount of precipitation during the 48 hours before sampling, and, for quadrat data only, mean soil moisture. This model was used for each age class of each species captured in sufficient numbers.

This study offers several advantages over much of the previous research into the effects of timber harvests on salamanders. Advantages include sampling across the relatively large geographic area encompassed by the HEE, the collection of pre- as well as post-harvest data, the inclusion of both landscape and local scales, and the relatively long study duration (4 years). While this study investigates the immediate effects of harvests on salamanders, most of the sampling arrays will remain in place for continued monitoring in order to study long-term effects. The results of this study will be a valuable contribution to our understanding of how different forest management techniques impact terrestrial salamanders and the wider ecosystem.

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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.

A PRE-TREATMENT ASSESSMENT OF SMALL MAMMALS IN THE HARDWOOD ECOSYSTEM EXPERIMENT

Natasha A. Urban and Robert K. Swihart[1]

Abstract.—The Hardwood Ecosystem Experiment (HEE) is a 100-year, replicated experiment implemented in south-central Indiana to examine the impacts of multiple timber management regimes on forest ecosystems. A secondary objective of the HEE is to evaluate responses of small mammal assemblages. We trapped at 32 sites prior to silvicultural treatments to assess pre-treatment small mammal communities. Trapping at all sites in both years allowed for modeling of multi-season occupancy and relative abundance using environmental covariates while incorporating imperfect detection. Estimated occupancy probabilities and species richness were larger than naïve estimates. Species richness was not significantly different between treatments or years. Except for the abundance of eastern chipmunks (*Tamias striatus*), the probability of occupancy and relative abundance of species did not differ among proposed treatment units. Abundance of white-footed mice (*Peromyscus leucopus*) and short-tailed shrews (*Blarina brevicauda*) and survival of local populations of pine voles (*Microtus pinetorum*) were greater for sites with northeastern aspects. Abundance of short-tailed shrews and pine voles also increased with greater herbaceous ground cover. By incorporating detection probability, we were able to derive more accurate estimates of relative abundance and, when coupled with a Bayesian framework, estimate occupancy for uncommon species. The baseline responses reported here can be used by forest managers to determine impacts of even-aged and uneven-aged oak (*Quercus*) management on small mammals subsequent to timber harvest.

INTRODUCTION

The most accurate depiction of species responses to forest management involves longitudinal studies that monitor focal taxa at a site from the time of timber harvest until the site is subjected to harvest again. Due to the cost and long-term commitment required of broad-spectrum longitudinal studies, most studies restrict their focus to responses over brief periods before and after a forest management activity (Fantz and Renken 2005, Ford and Rodrigue 2001, Kirkland 1990, Potvin et al. 1999, Yahner 1992) or to responses inferred from chronosequences sampled at a point in time (Urban and Swihart 2011). Although valuable, these short-term studies fail to portray wildlife response to forest regeneration beyond the earliest stages. Further, it is important to observe population dynamics across landscapes, not just within stands (Gram et al. 2001). With the notable exception of the Missouri Ozark Forest Ecosystem Project (MOFEP) (Gram et al. 2001,2003), few studies have been devoted to a complete landscape-level, longitudinal assessment of small mammal responses.

The Hardwood Ecosystem Experiment (HEE) was established in south-central Indiana to examine the ecological and social impacts of alternative forest

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management regimes on public land. Objectives of this 100-year, landscape-level, replicated experiment are reviewed by Kalb and Mycroft (this publication).

Secondary objectives for the HEE include determination of the effect of alternative forest management practices on the long-term distribution, diversity, and abundance of small mammals at local and landscape scales. Within this study, we sought to characterize the small mammal communities and populations of the HEE forest stands before silvicultural treatments were applied.

STUDY AREA

Sampling sites were located in three different publicly owned tracts, all located in the Brown County Hills region between Bloomington, Martinsville, and Nashville, IN. Two of the tracts, Morgan-Monroe State Forest (MMSF) and Yellowwood State Forest (YWSF), are part of the HEE and are discussed in more detail by Kalb and Mycroft (this publication). The third location, Brown County State Park (BCSP), was established in 1920 and is the largest state park in Indiana at 64 km^2 (lat. 39°16' N long. 86°22' W). All three sites have a similar history inasmuch as they all consist of land that was farmed in the 1800s and early 1900s and then reverted to stands of native oak-hickory (*Quercus-Carya*), American beech-maple (*Fagus grandifolia-Acer*), and mixed mesophytic forest with scattered pine (*Pinus*) plantations established in the early half of the 20th century (Carman, this publication). BCSP has not been subjected to timber harvest since its inception, whereas the State Forests have been harvested, principally via uneven-aged management, over the past several decades.[2] Sample site selection and descriptions for MMSF and YWSF are described by Kalb and Mycroft (this publication). Four small-mammal trapping grids were established in each of the 6 treatment research cores, resulting in 12 grids per management treatment (Fig. 1). Along with the three HEE control trapping sites at MMSF and YWSF, a fourth was located in BCSP. Two trapping grids were established at each of the four sampling sites, resulting in a total of eight control grids for sampling of small mammals (Fig. 1).

MATERIALS AND METHODS

Mammal Sampling

Each grid consisted of 27 trapping stations arranged in a 3 x 9 lattice (Fig. 2). The three long transect lines were laid out 50 m apart across the slope with 20- to 50-m spacing between stations, depending on the size of the harvest treatment (Figs. 2a,b). As with the

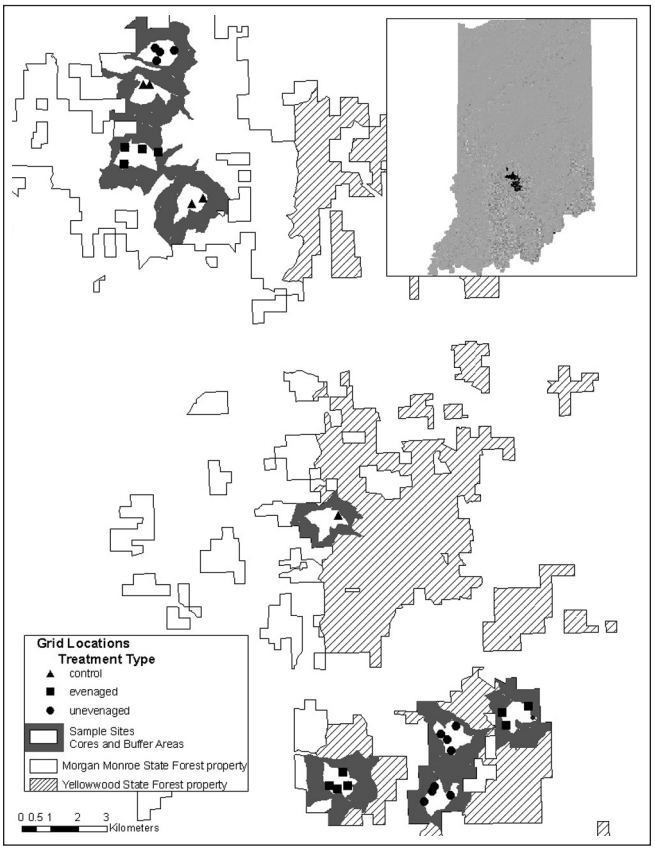


Figure 1.—Sampling sites and small mammal grid locations for the Hardwood Ecosystem Experiment. These nine sampling sites are located in Morgan-Monroe and Yellowwood State Forests. A tenth sampling site is located at Brown County State Park and contains a control grid.

[2] Duane McCoy, Indiana Department of Natural Resources-Division of Forestry, and Mike Mycroft, Indiana Department of Natural Resources-Division of State Parks and Reservoirs, personal communication

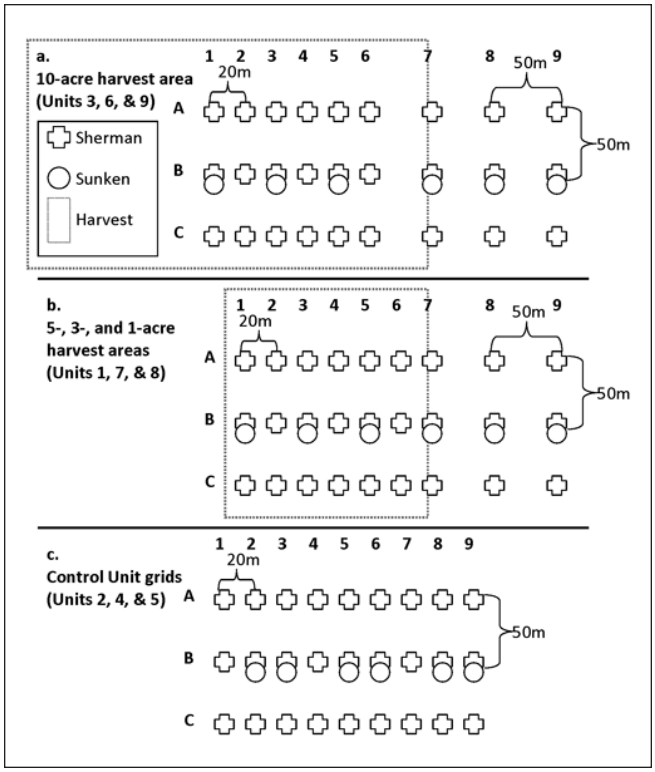


Figure 2.—Layout of Hardwood Ecosystem Experiment small mammal trapping grids. Several grid orientations were used depending on the size and treatment of the potential harvest area.

treatment grids, the control grids contained 27 stations allocated among three transects that were spaced 50 meters from each other and were placed across slope (Fig. 2c).

Every station received a Sherman trap (H.B. Sherman Trap, Inc., Tallahassee FL), and six sunken traps were placed within the harvest area on the center transect (Fig. 2). The sunken traps were composed of 16.5cm x 15.9cm aluminum cans with small perforations for drainage. Sherman and sunken traps were placed within 1 and 1.5 m, respectively, of the center of a trap station. Sunken traps were situated whenever possible along a natural drift fence such as a fallen log. Due to disturbance caused by raccoons (*Procyon lotor*) and opossums (*Didelphis virginiana*), Tomahawk traps (Tomahawk Live Traps Co., Tomahawk, WI) were placed in each corner of a trapping grid and baited with cat food. All captured raccoons and opossums were relocated at least 10 km away from a sample site to discourage their immediate return to the site.

Sherman traps were pre-baited for the 3 days preceding trapping. Small mammals were habituated to sunken traps during the pre-baiting period by placing a plastic lid over the trap and then covering it with leaf litter. Following pre-baiting, traps were set and checked for 5 consecutive days, both in the morning and evening. Sherman traps were baited with a mixture of sunflower seeds and rolled oats, and sunken traps were provisioned with earthworms.

For each captured individual, species, weight, sex, and reproductive status were recorded. Rodents were considered reproductively active if they were lactating or had an enlarged pubic symphysis (females) or descended testes (males). Shrews were toe clipped for identification. All trapping and handling procedures were approved by the Purdue Animal Care and Use Committee (protocol #07-045).

Vegetation Sampling

For each harvest site, a short site description was recorded. At the center of the harvest area, percent slope was estimated using a clinometer, and plot aspect was estimated in degrees using a compass. Slope position and shape also were determined using pre-defined classifications (Ruhe 1975). Within an 11.4-m radius (0.04 ha) from plot center, all trees >5 cm diameter at breast height (d.b.h.) were identified to species and d.b.h. was measured. For all coarse woody debris >10 cm diameter at its midpoint, length, midpoint diameter, and decay class (Maser et al. 1979) were recorded.

Vegetative structure also was recorded at the center of each harvest area. At the cardinal points of a circle of radius 3.6 m (0.004 ha) from the center of the harvest area, an ocular estimate of cover of herbaceous plants and seedlings <1 m tall was made within a 1-m^2 quadrat. All saplings <5 cm d.b.h. and >1 m tall were identified to species and counted within the 0.004-ha circle. Each stem was counted for multi-stemmed vegetation.

Canopy structure was sampled at 2-m intervals along four 10-m transects oriented up slope, down slope, and perpendicular to slopes from the center of each harvest area. At each sampling point, a density pole was used to measure vegetation density in four vertical strata with 1-m height increments. At each decimeter on the density pole, vegetation within 5 cm of the pole was counted as a contact, and the dominant species was recorded. The density was then determined by summing the number of decimeter contacts (Mills et al. 1991). At the end of each transect, a spherical densiometer was used to estimate canopy cover.

Microsite sampling was done for each small mammal trapping station. At each station, a 1-m radius circle was placed adjacent to the Sherman trap. The percent herbaceous and woody cover for plants <50 cm tall was recorded. At the circle center, depth of the leaf litter was measured. The length of all coarse woody debris >5 cm also was sampled.

Analysis Based on Occurrence

All sites were sampled for 2 consecutive years, thus permitting use of a multi-season, Bayesian hierarchical multi-species model of site occupancy that incorporated imperfect detection (Kéry and Royle 2008, Royle and Dorazio 2008: 390-393). The model enables estimation of both species- and community-level attributes, including survival and colonization parameters.

The general structure of the hierarchical model has been described in detail elsewhere (e.g., Royle and Dorazio 2008). Briefly, each of R (=32) harvest sites is visited on J (=5) occasions for T (=2) years, and for each occasion a record is made of species detected. For each of the n species, a count, y_{kt}, denotes the number of detections of the species in J visits to site k in year t. Conditional on the target species' occurrence at site k, the corresponding y_{kt} can be modeled as a binomial random variable with J trials consisting of detection probability p_{kt}. The detection frequencies for each of the ith observed species at the R sites can be summarized conveniently in a matrix $\mathbf{Y}_{n*R}$. For the initial year (t=1) and for the ith species and kth site, the species occurs with probability ψ_{ik}. A latent state variable z_{ikt} represents whether the species occurs at the site (z_{ikt}=1) or not (z_{ikt}=0). Occupancy state of the subsequent year (t+1) and survival ($\varnothing_t$) and colonization (γ_t) probabilities are dependent on the initial occupancy state (Royle and Dorazio 2008):

$$z(i,k,t+1) \mid z(i,k,t) = \text{Bernoulli}(\pi(i,t))$$

where

$$\pi(i,t) = z(i,k,t)(\varnothing_i)+[1 - z(i,k,t)]\gamma_i .$$

In words, the occupancy state for species i at site k in year t+1 is treated as a Bernoulli random variable that is conditioned on the prior year's occupancy state. If the site was occupied by the species in year t, the probability of occupancy in t+1 is determined by the probability of survival from t to t+1. But if the site was unoccupied in year t, its probability of occupancy is determined by the probability of colonization during the period t to t+1. Thus, the hierarchical model includes a matrix $\mathbf{Z}_{ikt}$ of state variables that are only partially observed because detection of a species at a site signifies occurrence there, but failure to detect a species at a site does not imply its absence at the site. Occurrence and detection were modeled as hyperparameters. Specifically, we defined u_i = logit(ψ_{ik}) and v_i = logit(p_{ik}), where logit(x) = ln(x/(1-x)). Interspecific heterogeneity in ψ and p were modeled with bivariate normal distributions, i.e., $u_i \sim N(\beta,\sigma_u^2)$, $v_i \sim N(\alpha, \sigma_v^2)$, and covariance σ_{uv}. The parameters β and α represent the mean probabilities of occurrence and detection, respectively, on a logit scale when considering all observed species in the community (Royle and Dorazio 2008: 382).

The hierarchical model is easily extended to incorporate effects of environmental covariates on ψ_{ikt}, the probability of occurrence of species i at site k in year t. Using the notation above, $z_{ikt} \mid \psi_{ikt} \sim$Bernoulli($\psi_{ikt}$), where logit($\psi_{ikt}$) is a linear function of site-specific covariates. Planned treatment type,

basal area, and aspect were chosen as site covariates of occurrence. Incorporating covariates into the observation portion of the model required modified notation because detection may differ among species, sites, sampling occasions, and years. Thus, y_{ijkt} is a binary observation indicating detection (y_{ijkt}=1) or not (y_{ijkt}=0) of species *i* during the *j*th visit to site *k* in the *t*th year. Then $y_{ijkt}|p_{ijkt},z_{ikt}$~Bernoulli($p_{ijkt},z_{ikt}$), where logit($p_{ijkt}$) is a linear function of site- or time-specific covariates that could influence detection. We considered average daily temperature (Temp), precipitation (Precip), year, sampling effort (Effort), Julian day (JD), and squared Julian day centered around the mean (JD^2) as possible covariates of detection. Detection covariate data are summarized in Appendix 1. We determined the best combination of detection covariates with the smallest deviance information criterion (DIC) value (Spiegelhalter et al. 2002). We then incorporated the chosen site-specific occupancy covariates: even-aged treatment type (Trt 1), uneven-aged treatment type (Trt 2), aspect, and basal area (BA). Management treatment types were grouped according to management system: even-aged or uneven-aged. All continuous occupancy and detection covariates were standardized prior to analysis.

The multi-season hierarchical multi-species site-occupancy model was implemented within a Bayesian framework (Appendix 2). We chose non-informative priors to ensure that inference was driven by data collected during our study. Specifically, we selected priors for the inverse logit of α and β from uniform distributions over the interval [0, 1], priors for σ_u and σ_v from uniform [0, 10], and priors for the correlation of *u* and *v*, $\rho=\sigma_{uv}/\sigma_u\sigma_v$, from uniform [-1, 1]. The model was implemented in the software package R 2.6.1 (R Development Core Team, Vienna, Austria) using the add-on package R2WinBUGS (Sturtz et al. 2005), which calls the software package WinBUGS (version 1.4.3; Lunn et al. 2000, Spiegelhalter et al. 1996). WinBUGS uses Markov chain Monte Carlo techniques to derive posterior distributions for model parameters.

For each run, we used five parallel chains of length 55,000 and discarded the first 5,000 to avoid effects due to random starting values (Kéry and Royle 2008). A thinning rate of 50 was used to reduce the likelihood of dependent samples (Ntzoufras 2009). Gelman-Rubin diagnostics were used to assess convergence (Brooks and Gelman 1998).

Community-level attributes were derived from the elements of **Z**. Species richness was estimated for site *k* as $s_k = \sum_{i=1}^{n} z_{ikt}$. An estimate of average species richness for category a of harvest sites was derived by summing across all sites in the category and then dividing by the A sites in the category; i.e., $\overline{S}_a = \sum_{k \in a} S_k / A$. Estimates of average species richness were thus obtained for each potential treatment type of harvest sites. The same procedure was used to derive estimates of the average similarity between pairs of categories for each year. We used Jaccard's coefficient (Jaccard 1912) to measure similarity in species richness between categories a and b:

$$C_{ab} = \frac{S_{a \cap b}}{S_a + S_b - S_{a \cap b}}$$

$S_{a \cap b}$ represents the number of species shared by classes *a* and *b*, whereas S_a and S_b are the total number of species in categories *a* and *b*, respectively. Thus, total overlap in species yields C_{ab}=1, and no shared species yields C_{ab}=0.

Analysis Based on Relative Abundance

For the four most commonly captured species, we implemented a multi-season version of the Royle and Nichols (2003) model, incorporating heterogeneity in *p* with finite mixtures. For N_{kt} animals at trap location *k*, replicate sampling yields a record of the number of animals detected there for each year *t*. Then the probability of detecting at least one animal at the location, given that the species occurs there, is $p_{kt} = 1-(1-r)^{N_{kt}}$, where *r* is the probability of capturing an individual (Royle and Nichols 2003).

Considering abundance as a Poisson random variable, i.e., $Pr(N\text{-}n)=e^{-\lambda}\lambda^{n}/n!$ where λ is the mean of the Poisson distribution on N, permits estimation of r and λ by the method of maximum likelihood (Royle and Nichols 2003).

We implemented a multi-season version of the Royle-Nichols model using PRESENCE 2.4 (Hines 2006) with species-level detection histories developed for each trap location. Because a single detection model is fitted to observations from all harvest sites, site-specific estimates of density are not independent. Thus, we used the two-stage bootstrap method of Buckland et al. (2009) to quantify precision. Specifically, for each species, we determined the combination of covariates for r resulting in the smallest Akaike's Information Criterion (AIC) value (Burnham and Anderson 2002). We then incorporated site-specific covariates for λ and chose the model with the lowest AIC.

Detection covariates considered for analysis were JD, JD^2, year, Effort, Temp, and Precip. Site-specific covariates for λ were coarse woody debris at the trap level (Micro CWD), percentage of herbaceous cover at the trap level (Herb), percentage of woody cover at the trap level (Wood), depth of leaf litter at trap level (Litter), Trt 1, Trt 2, BA, and aspect. Confidence intervals and standard errors for each parameter were estimated using bootstrap re-sampling of trap locations in each harvest site (Urban 2010). The selected Royle and Nichols (2003) model was fitted in R 2.6.1 using at least 200 bootstrap samples for each species, resulting in 200 x 1,728 trap-specific abundance values. These estimates were used to derive standard errors of N for each harvest site. Trap spacing was likely to result in multiple trap stations within a home range, so we suspect that our estimates of λ are biased high and are more likely to serve as estimates of relative abundance (Urban and Swihart 2011).

RESULTS

Trapping Results

Trapping in 2007 occurred from 25 June through 3 August with 8,936 trap nights and 6 species captured. In 2008 trapping took place from 16 June to 25 July for a total of 8,978 trap nights and 7 species captured. Total captures consisted of 2,101 white-footed mice (see Table 1 for list of common and scientific names), 1,143 eastern chipmunks, 96 pine voles, 71 short-tailed shrews, 4 smoky shrews, 4 southeastern shrews, and 3 long-tailed weasels.

Occupancy Analysis

All species were used for the multi-season, multi-species occupancy models. Average temperature and Julian day were the best DIC detection covariates for both years of data. Pine voles were more detectable during later trapping dates (Table 2). Conversely, eastern chipmunks had higher detections during earlier trapping dates. Mean estimates of occupancy for all species and for both years increased from naïve occupancy after the incorporation of detection probability (Fig. 3). Ninety-five-percent credible intervals for all treatment covariate by species combinations contained zero, indicating that they were not useful predictors of occupancy for any of the seven species. Likewise, treatment did not affect the survival or colonization of local populations, with one exception (Table 3). Survival of local pine vole populations was positively affected by northeastern aspects.

The incorporation of probability of detection increased species richness values. The mean estimated species richness for all sites in 2007 (4.6) was 47 percent higher than the naïve mean species richness (3.1). In 2008, the mean estimated species richness (4.8) was 60 percent higher than naïve richness (3.0). Mean estimated species richness did not differ significantly among treatment types ($F = 0.47$, $p = 0.63$) or years ($F = 0.60$, $p = 0.44$).

Table 1.—Trapping results for the Hardwood Ecosystem Experiment pre-treatment sampling period. Seven species were captured during the 2007 and 2008 sampling period. All seven species were used for the occupancy analysis, and the four most common were used for the relative abundance analysis.

Species	Scientific name	2007	2008	Total
White-footed mouse	*Peromyscus leucopus*	1,152	949	2,101
Eastern chipmunk	*Tamias striatus*	453	690	1,143
Pine vole	*Microtus pinetorum*	47	49	96
Short-tailed shrew	*Blarina brevicauda*	39	32	71
Smoky shrew	*Sorex fumeus*	3	1	4
Southeastern shrew	*Sorex longirostris*	3	1	4
Long-tailed weasel	*Mustela frenata*	0	3	3

Table 2.—Results from the multi-season, hierarchical multi-species occupancy model analyzed within a Bayesian framework. Asterisk (*) indicates parameters for which 95-percent credible intervals excluded zero.

	White-footed mouse	Eastern chipmunk	Pine vole	Short-tailed shrew	Smoky shrew	Southeastern shrew	Long-tailed weasel
Covariate	β_i SE	β_i SE	β_i SE	β_i SE	β_i SE	β_i SE	β_i SE
p	-0.02	-0.29*	0.39*	-0.10	-0.49	-0.68	0.19
Julian day	0.14	0.17*	0.21*	0.20	0.47	0.52	0.45
	-0.04	-0.07	-0.13	0.06	-0.01	0.02	0.00
Temperature	0.12	0.14	0.16	0.17	0.22	0.23	0.24
Ψ	0.98	0.38	0.36	0.91	1.18	0.16	0.18
Uneven-aged	1.31	1.10	0.94	1.26	1.42	1.57	1.54
	0.78	-0.01	0.35	-0.85	-0.74	1.22	-0.52
Even-aged	1.50	1.18	1.05	1.27	1.83	1.67	1.68
	-0.14	-0.24	-0.12	-0.09	-0.38	-0.12	-0.10
Basal area	0.61	0.50	0.38	0.47	0.75	0.87	0.80
	0.97	-0.32	1.04	0.73	0.49	0.45	0.00
Aspect	1.30	1.13	0.91	1.10	1.46	1.48	1.53

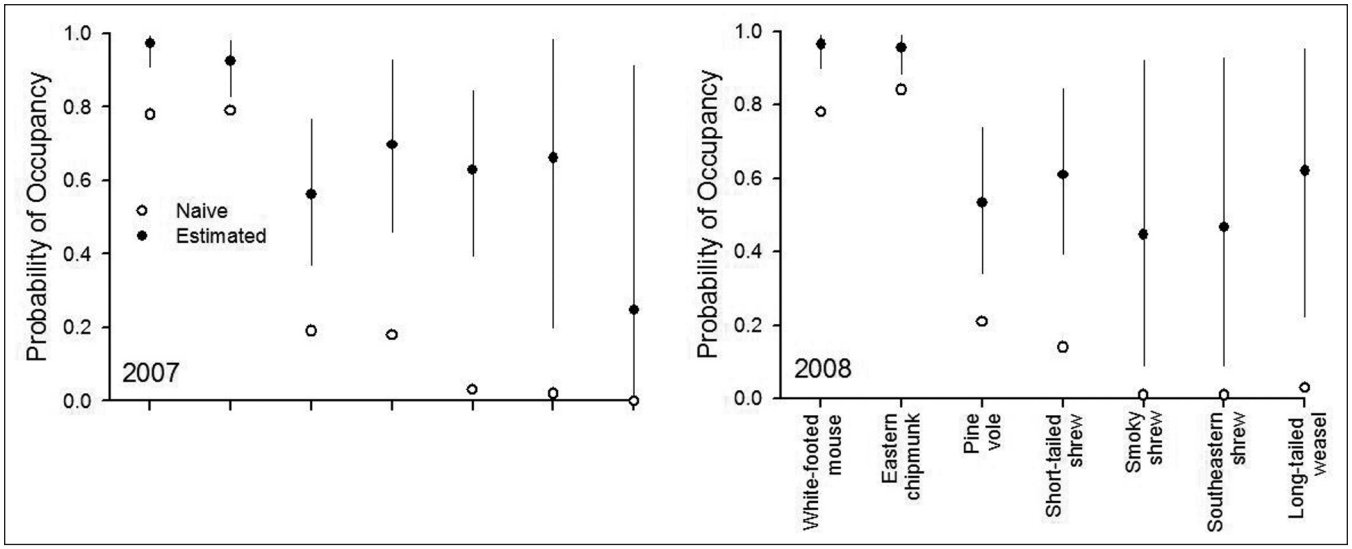


Figure 3.—Naïve and estimated (95-percent credible interval) values for probability of occupancy from the multi-season, multi-species occupancy model. All estimated occupancy values were higher than the naïve values.

Table 3.—Survival and colonization parameters from the multi-season, hierarchical multi-species occupancy model analyzed within a Bayesian framework. Models included even-aged or uneven-aged proposed treatment type, aspect, and basal area occupancy covariates as explanatory variables. The best detection probability covariates included were Julian day and average temperature, as determined by the lowest Deviance Information Criterion. Asterisk (*) indicates parameters for which 95-percent credible intervals excluded zero.

	White-footed mouse	Eastern chipmunk	Pine vole	Short-tailed shrew	Smoky shrew	Southeastern shrew	Long-tailed weasel
Covariate	β_j SE	β_j SE	β_j SE	β_j SE	β_j SE	β_j SE	β_j SE
Uneven-aged							
Colonization	0.52	0.88	1.22	0.60	0.32	0.44	0.43
	1.95	1.79	1.67	1.87	1.98	1.92	2.00
Survival	1.25	1.22	0.75	1.18	0.16	0.69	0.74
	1.45	1.47	1.46	1.43	1.89	1.73	1.76
Even-aged							
Colonization	-0.96	-0.45	-1.71	-1.35	-1.23	-1.04	-1.45
	1.96	1.90	1.49	1.63	1.88	1.91	1.87
Survival	1.17	1.17	1.13	0.63	0.52	0.45	0.64
	1.44	1.45	1.34	1.57	1.80	1.81	1.78
Basal area							
Colonization	-0.09	0.37	0.59	-1.09	-0.19	-0.50	-0.01
	1.88	1.86	1.50	1.74	1.86	1.84	1.60
Survival	-0.09	-0.09	-0.23	0.02	-0.03	-0.29	-0.06
	0.83	0.84	0.88	0.71	1.16	1.16	1.19
Aspect							
Colonization	0.15	1.13	-0.34	0.31	-0.19	-0.16	0.40
	1.99	1.77	1.66	1.66	1.97	1.99	1.82
Survival	1.85	1.82	2.26*	1.99	0.74	0.80	1.36
	1.44	1.47	1.29*	1.37	2.10	2.07	1.86

Jaccard's similarity coefficients for 2007 and 2008 indicated that all pairs of projected treatments had nearly two-thirds of their species in common (Table 4). This level of similarity was consistent across years.

Relative Abundance Analysis

White-footed mice, eastern chipmunks, pine voles, and short-tailed shrews were captured in sufficient numbers for the multi-season, single-species relative abundance analysis. All species except pine voles were more detectable during earlier trapping (Table 5). Trapping effort positively influenced probability of detection of mice and pine voles. Increased trapping effort and occurrence of precipitation negatively influenced eastern chipmunk detection probability, whereas increased temperature positively influenced detection probability. Occurrence of precipitation also negatively affected white-footed mice.

Relative abundance of white-footed mice and short-tailed shrews was greater on sites with northeastern aspects versus southwestern aspects. Short-tailed shrew and pine vole relative abundance was greater on sites with increased herbaceous cover. Pine voles also had higher relative abundance with greater amounts of coarse woody debris at the trap level. Relative abundance of eastern chipmunks was lower on sites proposed for even-aged treatment.

Table 4.—Jaccard's similarity coefficients (95-percent credible interval) comparing species richness values between the different proposed treatment types. Comparisons were made for each year separately. A similarity coefficient value of 1 indicates total overlap in species, whereas a value of 0 indicates no shared species. Comparisons for 2007 are located above the diagonal and comparisons for 2008 are below.

	Treatment type		
Treatment type	Even-aged	Uneven-aged	Control
Even-aged	1	0.61 (0.54, 0.76)	0.63 (0.55, 0.77)
Uneven-aged	0.61 (0.55, 0.76)	1	0.62 (0.56, 0.76)
Control	0.63 (0.55, 0.78)	0.63 (0.55, 0.77)	1

Table 5.—Results for the multiple-season, single-species relative abundance analysis for the four most abundant species. Bold-faced type indicates significance (P < 0.05). Asterisk (*) indicates parameters for which 95-percent credible intervals excluded zero.

Parameter	White-footed mouse		Eastern chipmunk		Pine vole		Short-tailed shrew	
Covariate	β_j	SE	β_j	SE	β_j	SE	β_j	SE
p								
Intercept	-1.75*	0.12*	-1.61*	0.41*	-2.55*	0.46*	-2.40*	0.48*
Julian day	-0.73*	0.12*	-0.39*	0.17*			-1.53*	0.77*
Julian day^2			0.06	0.10	0.64	0.46		
Trapping effort	0.59*	0.12*	-0.55*	0.12*	1.49*	0.57*	0.02	0.28
Temperature			0.58*	0.11*				
Precipitation	-1.17*	0.12*	-0.38*	0.13*			-0.28	0.60
N								
Intercept	0.27*	0.11*	0.10	0.14	-2.96*	0.34*	-4.32*	0.66*
Uneven-aged								
Even-aged	-0.15	0.10	-0.82*	0.14*				
Aspect	0.38*	0.09*	0.18	0.10	0.51	0.37	1.62*	0.63*
Basal area							0.81	0.42
Herb					0.31*	0.15*	1.62*	0.63*
Wood			0.11	0.05				
Micro CWD							0.42*	0.21*

DISCUSSION

Probability of Detection

The incorporation of probability of detection increased occupancy probabilities for all species in 2007 and 2008 relative to naïve estimates. Large credible intervals characterized the occupancy analysis, but naïve estimates for all species in both years were still smaller than the 2.5-percent lower credible intervals. Larger credible intervals and elevated occupancy probabilities were produced for the rarer species due to their lower estimated probabilities of detection. Occupancy estimates for species with a probability of detection less than 0.15 should be viewed with caution when sampling occasions are less than seven, as models may not properly distinguish between a truly absent species and a non-detection (MacKenzie et al. 2002). Researchers conducting small mammal studies at our sampling sites may wish to follow minimum sampling guidelines developed by MacKenzie et al. (2002) to better estimate minimum number of sites based on detection probabilities and number of sampling occasions.

Treatment Effects

The planned forest management practices for the HEE have the potential to alter small mammal populations and assemblages. Our study revealed few pre-harvest differences among the sites chosen for the various treatments. Especially encouraging to future small mammal work on the HEE, treatment type was not a significant predictor of probability of occupancy or abundance, except for lower relative abundance of eastern chipmunks on sites slated for even-aged management. Species richness and similarity also did not differ greatly among treatment types. These similar pre-treatment levels will enable easier interpretations of treatment effects in subsequent analyses.

Site descriptors of forest structure, basal area, and aspect failed to influence probability of occupancy for most species. One exception was higher survival of pine vole populations occupying northeastern aspects. For most species relative abundance was affected by site descriptors of forest structure and aspect. Abundance of white-footed mice and short-tailed shrews was higher on northeastern facing sites, which are often cooler and more mesic. Along with showing a relationship with northeastern aspect, short-tailed shrews were found on sites with increased herbaceous cover and trap-level coarse woody debris. Pine voles also were more abundant with increased herbaceous cover.

Similar reactions from these species have been documented in multiple studies. Higher occupancy probability has been seen for short-tailed shrews on northeastern sites in south-central Indiana (Urban and Swihart 2011). Miller and Getz (1977) also found that short-tailed shrews were more common at New England sites with higher herbaceous cover. Pine voles are associated with habitats containing thick litter and herbaceous cover (Smolen 1981).

Previous habitat studies of short-tailed shrews in Michigan noted that deep litter cover is important to protect them from dessication and high temperatures (Pruitt 1953, 1959). Getz (1961) concluded that short-tailed shrews avoid areas with little vegetative cover and extreme temperatures and moisture levels in Illinois. Schmid-Holmes and Drickamer (2001) reported higher abundance for short-tailed shrews at Illinois sites with less extreme temperature fluctuations and higher tree densities, indicative of older seral stages.

The relationships between these species and increased herbaceous cover, coarse woody debris, and northeastern aspects indicate that they may have an initial negative response to proposed forest management treatments. Schmid-Holmes and Drickamer (2001) found greater abundance of white-footed mice in older forests that were subjected to no harvest or uneven-aged harvesting. A shift from young post-harvest openings of small mammal assemblages dominated by eastern chipmunks to old harvest sites dominated by white-footed mice occurred along a chronosequence in south-central Indiana (Urban and Swihart 2011). Pine voles and short-tailed shrews also were more likely to be found in older sites along the chronosequence. However, other studies have indicated that both even-aged and uneven-aged management may positively influence *Peromyscus* abundance, and short-tailed shrews may have no change in abundance (Fantz and Renken 2005, Ford and Rodrigue 2001).

Community-Level Responses

Our pre-treatment analysis described the baseline community structure and population dynamics for the HEE. Subsequent studies should use analytical methods similar to the ones described in this chapter in order to accurately describe and compare post- and pre-treatment data. However, we suggest that a more intensive sampling scheme should be used to increase captures of less common species. Increased trapping effort and subsequent higher captures of rare and elusive species could improve future analyses

using multi-season, multi-species occupancy models. Specifically, focusing trapping effort to sample existing sites more often instead of surveying an increased number of sites less often likely would improve the precision of these occupancy estimates (MacKenzie and Royle 2005).

If detection probabilities remain low, a removal sampling design may produce more precise estimates (MacKenzie and Royle 2005). Increased capture rates would help reduce the large credible intervals associated with *Sorex* occupancy estimates. Increased capture rates also could allow more precise estimation of extinction and colonization rates between post- and pre-treatment data, which could be informative in understanding treatment effects.

Potential Effects of Oak Management

Results from a chronosequence in similar habitat with similar management practices can provide an idea of what changes to expect in the HEE small mammal community following treatment. Based on our analysis of a chronosequence at Crane Naval Surface Warfare Center (NSWC) in 2007 and 2008 (Urban and Swihart 2011), we offer the following predictions for responses of small mammals to the HEE treatments.

The most frequently captured species at the HEE was the white-footed mouse, which was ubiquitous but most strongly associated with older seral stages found at NSWC sample sites. In contrast, the eastern chipmunk was the most frequently captured species on NSWC, where forest management practices have been ongoing since the 1950s. The early successional openings at NSWC were dominated primarily by eastern chipmunks. In general, NSWC was characterized by a shift from chipmunk-dominated early successional openings to more diverse white-footed mouse-dominated late-successional openings. Pine voles and short-tailed shrews also were associated with older sites at NSWC. Thus, we predict increases in eastern chipmunks on recently harvested even-aged and uneven-aged stands. In contrast, white-footed mice, shrews, and pine voles, and hence species richness, should decrease at these early successional sites. As stands mature, a reversal of these trends is likely to occur.

Short-tailed shrews were positively associated with northeastern aspects at both NWSC and the HEE, whereas white-footed mice were positively associated with northeastern aspects at the HEE and eastern chipmunks were negatively associated with this aspect at NSWC. *Blarina* and *Sorex* are sensitive to dry environments (Brannon 2002; Getz 1961; McShea et al. 2003; Pruitt 1953, 1959); hence, we predict increases in shrews on northeastern aspects, especially as successional changes lead to increased litter layer development. Eastern chipmunks should be most prevalent at drier early successional sites. No effect of opening size created by even-aged and uneven-aged harvesting was found on small mammal populations at NSWC sites, although species richness was greater at smaller openings. Consequently, we predict that small mammal populations at the HEE will not respond to the size of harvest openings, although species richness may increase at smaller sites as an artifact of sampling species associated primarily with adjacent habitats.

White-footed mice at NSWC exhibited greater relative abundance in openings with high levels of basal area and coarse woody debris. In contrast, eastern chipmunks at NSWC exhibited greater relative abundance in openings with low basal area and high canopy closure. Based on these results, we predict that white-footed mice will be more abundant in sites with greater tree retention and where coarse woody debris has been retained. These conditions could occur in more mature stands, greater than 27 years, or in harvest areas that remove fewer trees.

Eastern chipmunks should have larger numbers at young sites that still retain some canopy cover, such as 4- to 12-year-old sites. They should also

have greater numbers at sites with more intensive harvesting techniques that retain some canopy cover, such as shelterwood, seed tree, and group selection methods. However, these species are likely to have varying numbers within a site as well. White-footed mice likely will exhibit higher abundance around coarse woody debris or patches of trees within a site. At NSWC, relative abundance of chipmunks and mice was negatively affected by proximity to water, whereas the opposite was true for short-tailed shrews. Following harvest, we predict that these associations will emerge at the HEE sites.

We did not consider the importance of mast to the population dynamics of the small mammal assemblages at the HEE. The two dominant species in the small mammal system at the HEE, white-footed mice and eastern chipmunks, are important mast consumers whose abundances are affected by mast production (McShea 2000, Wolff 1996). The amount of oak basal area, oak as a percent of the total basal area, and mast production data may explain spatial or temporal variation in abundance of these species. These data are currently being collected at our study sites (Kellner et al., this publication).

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APPENDIX 1.

Sampling data for the probability of detection covariates for the 2 years of trapping at HEE. JD is the first Julian day of trapping for a season at a site. Effort is the number of trapping occasions, Temp is the average daily temperature, and Precip is the presence (= 1) or absence (= 0) of precipitation during and up to 3 hours before trapping.

				Temp (°C) for each day					Precip for each day				
Site	Year	JD	Effort	1	2	3	4	5	1	2	3	4	5
1001	2007	211	264	29	28	27	28	24	0	0	0	0	0
1001	2008	167	282	18	18	20	19	21	1	0	0	0	0
1002	2007	211	287	29	28	24	28	29	0	0	0	0	0
1002	2008	167	273	18	19	18	20	18	1	0	0	0	0
1105	2007	204	274	27	24	23	21	26	0	0	0	0	0
1105	2008	174	283	27	25	26	28	27	0	0	0	0	0
1113	2007	204	295	27	24	25	21	24	0	0	0	1	0
1113	2008	174	283	27	25	26	28	27	0	0	0	0	0
1121	2007	204	296	28	24	23	21	24	0	0	0	1	0
1121	2008	174	246	27	25	26	28	18	0	0	0	0	0
1125	2007	204	292	27	24	24	25	21	0	0	0	0	0
1125	2008	174	291	27	25	26	28	27	0	0	0	0	0
1202	2007	190	292	32	24	24	24	25	0	1	1	0	1
1202	2008	188	277	24	25	26	21	27	0	1	1	0	0
1204	2007	190	277	32	27	25	21	24	0	0	1	0	1
1204	2008	188	283	24	27	24	21	26	0	1	1	0	0
1312	2007	190	254	32	25	26	24	21	0	1	1	0	1
1312	2008	188	280	27	27	25	24	27	1	1	1	0	0
1317	2007	190	252	32	29	26	26	24	0	1	0	0	1
1317	2008	188	214	29	26	27	23	18	1	1	1	0	0
1321	2007	190	279	32	27	26	24	24	0	1	0	0	1
1321	2008	188	258	24	27	26	24	24	1	1	1	0	0
1326	2007	190	242	29	28	26	23	27	0	0	0	0	1
1326	2008	188	233	27	26	24	23	24	1	1	1	0	0
1401	2007	204	290	28	22	22	23	21	0	0	0	0	1
1401	2008	202	283	32	23	24	23	24	1	1	0	0	0
1404	2007	204	286	28	23	23	23	26	0	0	0	0	1
1404	2008	202	283	29	24	21	23	26	0	1	0	0	0
1502	2007	197	292	29	26	27	27	23	0	1	1	1	0
1502	2008	202	280	29	27	25	25	25	0	1	0	0	0
1504	2007	197	281	29	26	24	27	20	0	1	1	1	0
1504	2008	202	287	32	26	25	25	25	1	1	0	0	0

(Appendix 1 continued on next page)

APPENDIX 1 (continued).

Sampling data for the probability of detection covariates for the 2 years of trapping at HEE. JD is the first Julian day of trapping for a season at a site. Effort is the number of trapping occasions, Temp is the average daily temperature, and Precip is the presence (= 1) or absence (= 0) of precipitation during and up to 3 hours before trapping.

				Temp (°C) for each day					Precip for each day				
Site	Year	JD	Effort	1	2	3	4	5	1	2	3	4	5
1602	2007	176	253	31	27	24	26	24	0	0	0	1	1
1602	2008	180	287	21	21	23	24	21	0	0	0	0	1
1622	2007	176	252	24	24	26	26	26	1	0	0	1	0
1622	2008	195	288	27	27	26	27	27	0	0	0	0	0
1624	2007	176	266	27	27	27	25	27	1	0	0	1	0
1624	2008	195	293	27	27	27	27	25	0	0	0	0	0
1627	2007	176	281	27	27	24	27	24	1	0	0	1	0
1627	2008	180	293	21	23	18	21	24	0	0	0	0	1
1703	2007	183	277	29	24	24	26	28	0	0	1	1	0
1703	2008	167	285	18	21	21	20	18	1	0	0	0	0
1705	2007	183	278	27	25	26	25	26	0	0	1	1	0
1705	2008	167	292	18	21	19	19	22	1	0	0	0	0
1716	2007	183	288	27	25	25	28	28	0	0	1	1	0
1716	2008	167	292	18	19	19	19	22	1	0	0	0	0
1728	2007	183	289	29	21	26	24	24	0	0	1	1	0
1728	2008	167	291	18	21	21	20	19	1	0	0	0	0
1803	2007	197	288	29	25	24	26	18	0	1	1	1	0
1803	2008	180	290	21	20	23	18	21	1	0	0	0	1
1820	2007	197	292	29	24	21	18	18	0	1	0	0	0
1820	2008	180	280	21	19	23	24	26	0	0	1	1	1
1821	2007	197	276	29	23	22	27	20	0	0	0	1	0
1821	2008	180	292	21	20	23	24	26	0	0	0	0	1
1830	2007	197	283	29	24	25	24	24	0	1	0	0	0
1830	2008	180	296	21	20	24	19	22	0	0	0	0	1
1907	2007	211	289	29	24	28	28	29	0	0	0	0	0
1907	2008	195	294	29	21	28	27	24	0	0	0	0	0
1908	2007	211	293	29	20	20	21	21	0	0	0	0	0
1908	2008	195	296	29	27	24	28	28	0	0	0	0	0
1917	2007	211	292	28	21	26	27	28	0	0	0	0	0
1917	2008	195	283	29	21	24	27	23	0	0	0	0	0
1919	2007	211	280	27	26	24	27	28	0	0	0	0	0
1919	2008	195	291	29	27	27	26	26	0	0	0	0	0

APPENDIX 2.

Code for the multi-season hierarchical multi-species site occupancy model implemented within a Bayesian framework. This code includes the incorporation of survival and colonization parameters. It was run using software packages R 2.6.1 and WinBUGS using add-on package R2WinBUGS.

```
##data files
##encounter history data from 2007
##7 rows of species encounter histories
##columns of sites
my.2007=read.table(file="HEE_psi_2007.csv", header=T, sep=",",
row.names=1)
##encounter history data from 2008
##7 rows of species encounter histories
##columns of sites
my.2008=read.table(file="HEE_psi_2008.csv", header=T, sep=",",
row.names=1)
##covariate data from 2007
##14 rows of covariate data
##columns of sites
my.cov07=read.table(file="HEE_cov.csv", header=F, sep=",",
row.names=1)
##covariate data from 2008
##14 rows of covariate data
##columns of sites
my.cov08=read.table(file="HEE_cov2.csv", header=F, sep=",",
row.names=1)
##organizing encounter history data into an array
my.data2<-array(data=NA, dim=c(7,32,2))
my.year1<-as.matrix(my.2007)
my.year2<-as.matrix(my.2008)
my.data2[,,1]<-my.year1
my.data2[,,2]<-my.year2
my.data<-array(data=NA, dim=c(7,32,2,1))
my.data[,,,1]<-my.data2
##organizing even-aged treatment covariate into array for
input
my.T1_1<-array(data=NA, dim=c(32,2))
cov.T1_07<-as.matrix(my.cov07[2,])
cov.T1_08<-as.matrix(my.cov08[2,])
my.T1_1[,1]<-cov.T1_07
my.T1_1[,2]<-cov.T1_08
my.T1<-array(data=NA, dim=c(32,2,1))
116
my.T1[,,1]<-my.T1_1
##organizing uneven-aged treatment covariate into array for
input
my.T2_1<-array(data=NA, dim=c(32,2))
cov.T2_07<-as.matrix(my.cov07[3,])
cov.T2_08<-as.matrix(my.cov08[3,])
my.T2_1[,1]<-cov.T2_07
my.T2_1[,2]<-cov.T2_08
```

APPENDIX 2 (continued).

```
my.T2<-array(data=NA, dim=c(32,2,1))
my.T2[,,1]<-my.T2_1
##organizing basal area covariate into array for input
my.ba_1<-array(data=NA, dim=c(32,2))
cov.ba_07<-as.matrix(my.cov07[14,])
cov.ba_08<-as.matrix(my.cov08[14,])
my.ba_1[,1]<-cov.ba_07
my.ba_1[,2]<-cov.ba_08
my.ba<-array(data=NA, dim=c(32,2,1))
my.ba[,,1]<-my.ba_1
##organizing aspect covariate into array for input
my.aspect_1<-array(data=NA, dim=c(32,2))
cov.aspect_07<-as.matrix(my.cov07[5,])
cov.aspect_08<-as.matrix(my.cov08[5,])
my.aspect_1[,1]<-cov.aspect_07
my.aspect_1[,2]<-cov.aspect_08
my.aspect<-array(data=NA, dim=c(32,2,1))
my.aspect[,,1]<-my.aspect_1
##organizing temperature detection covariate into array for
input
my.temp_1<-array(data=NA, dim=c(32,2))
cov.temp_07<-as.matrix(my.cov07[7,])
cov.temp_08<-as.matrix(my.cov08[7,])
my.temp_1[,1]<-cov.temp_07
my.temp_1[,2]<-cov.temp_08
my.temp<-array(data=NA, dim=c(32,2,1))
my.temp[,,1]<-my.temp_1
##organizing Julian day detection covariate into array for input
117
my.jd_1<-array(data=NA, dim=c(32,2))
cov.jd_07<-as.matrix(my.cov07[12,])
cov.jd_08<-as.matrix(my.cov08[12,])
my.jd_1[,1]<-cov.jd_07
my.jd_1[,2]<-cov.jd_08
my.jd<-array(data=NA, dim=c(32,2,1))
my.jd[,,1]<-my.jd_1
#treat occupancy covariates and p covariates the same until
#the logit step
Ymat=my.data #my presence/absence data
mycov=my.cov #my covariates data
nrepls=5 #days sampled, monday through friday
##monitor amount of time it took to run
start.time=Sys.time()
n = dim(my.data)[1] #number of species indicated by number of
rows
nsites = dim(my.data)[2] #number of sites indicated by number
of columns
nyear=dim(my.data)[3]
nfiller=dim(my.data)[4]
#create arguments for bugs()
##input data
```

APPENDIX 2 (continued).

```
data = list(n=n, R=nsites, J=nrepls, Y=my.data, T=nyear, Q=nfiller,
c.T1=my.T1, c.T2=my.T2, c.aspect=my.aspect, c.ba=my.ba,
c.jd=my.jd, c.temp=my.temp)
##parameters listed after model runs
params = list( 'alpha', 'beta', 'beta1','beta2',
##T1surv = survival dependent on even-aged treatment,
T1col=colonization dependent on uneven-aged trtment
'T1','T1surv','T1col',
'T2','T2surv','T2col',
'aspect','aspectsurv','aspectcol',
'ba','basurv','bacol',
'temp',
'jd',
118
'rho', 'sigma.u', 'sigma.v',
'sigma.u1', 'sigma.v1', 'sigma.u2', 'sigma.v2',
's.T1','s.T1surv','s.T1col',
's.T2','s.T2surv','s.T2col',
's.aspect','s.aspectsurv','s.aspectcol',
's.ba','s.basurv','s.bacol',
's.jd',
's.temp',
'phi', 'eta',
'pvec',
'lphi0', 'lgamma0',
'psivec',
'phisurv',
'gamma',
'phiyr','gammayr'
'Z')
## initials
inits = function () {
psi.meanGuess = runif(1, .25, 1)
psi.mean1Guess = runif(1, .25, 1)
psi.mean2Guess = runif(1, .25, 1)
p.meanGuess = runif(1, .25, 1)
## initial values for covariates
beta.T1 = runif(1, 0, 1)
beta.T1col = runif(1, 0, 1)
beta.T1surv = runif(1, 0, 1)
beta.T2col = runif(1, 0, 1)
beta.T2surv = runif(1, 0, 1)
beta.T2 = runif(1, 0, 1)
beta.aspect = runif(1, 0, 1)
beta.aspectcol = runif(1, 0, 1)
beta.aspectsurv = runif(1, 0, 1)
beta.ba = runif(1, 0, 1)
beta.bacol = runif(1, 0, 1)
beta.basurv = runif(1, 0, 1)
beta.temp = runif(1, 0, 1)
beta.jd = runif(1, 0, 1)
```

APPENDIX 2 (continued).

```
rhoGuess = runif (1, 0, 1)
sigma.uGuess = runif (1, 0, 1.5)
sigma.vGuess = runif (1, 0, 1.5)
sigma.u1Guess = runif (1, 0, 1.5)
sigma.v1Guess = runif (1, 0, 1.5)
sigma.u2Guess = runif (1, 0, 1.5)
119
sigma.v2Guess = runif (1, 0, 1.5)
sigma.T1 = runif(1, 0, 1)
sigma.T1surv = runif(1, 0, 1)
sigma.T1col = runif(1, 0, 1)
sigma.T2 = runif(1, 0, 1)
sigma.T2surv = runif(1, 0, 1)
sigma.T2col = runif(1, 0, 1)
sigma.aspect = runif(1, 0, 1)
sigma.aspectsurv = runif(1, 0, 1)
sigma.aspectcol = runif(1, 0, 1)
sigma.ba = runif(1, 0, 1)
sigma.basurv = runif(1, 0, 1)
sigma.bacol = runif(1, 0, 1)
sigma.temp = runif (1, 0, 1)
sigma.jd = runif (1, 0, 1)
list(psi.mean=psi.meanGuess, p.mean=p.meanGuess,
psi.mean1=psi.mean1Guess,
psi.mean2=psi.mean2Guess,
b.T1=beta.T1,
b.T1surv=beta.T1surv,
b.T1col=beta.T1col,
b.T2=beta.T2,
b.T2surv=beta.T2surv,
b.T2col=beta.T2col,
b.aspect=beta.aspect,
b.aspectsurv=beta.aspectsurv,
b.aspectcol=beta.aspectcol,
b.ba=beta.ba,
b.basurv=beta.basurv,
b.bacol=beta.bacol,
b.temp=beta.temp,
b.jd=beta.jd,
sigma.u=sigma.uGuess, sigma.v=sigma.vGuess,
rho=rhoGuess,
sigma.u1=sigma.u1Guess, sigma.v1=sigma.v1Guess,
sigma.u2=sigma.u2Guess, sigma.v2=sigma.v2Guess,
s.T1=sigma.T1,
s.T1surv=sigma.T1surv,
s.T1col=sigma.T1col,
s.T2=sigma.T2,
s.T2surv=sigma.T2surv,
s.T2col=sigma.T2col,
120
s.aspect=sigma.aspect,
```

APPENDIX 2 (continued).

```
s.aspectsurv=sigma.aspectsurv,
s.aspectcol=sigma.aspectcol,
s.ba=sigma.ba,
s.basurv=sigma.basurv,
s.bacol=sigma.bacol,
s.temp=sigma.temp,
s.jd=sigma.jd,
phi=rnorm(n*nfiller, log(psi.meanGuess/(1-psi.meanGuess)),
sigma.uGuess),
lphi0=rnorm(n, log(psi.mean1Guess/(1-psi.mean1Guess)),
sigma.u1Guess),
lgamma0=rnorm(n, log(psi.mean2Guess/(1-psi.mean2Guess)),
sigma.u2Guess),
eta=rnorm(n*nfiller, log(p.meanGuess/(1-p.meanGuess)), sigma.vGuess),
T1=rnorm(n*nfiller, log(beta.T1/(1-beta.T1)), sigma.T1),
T1surv=rnorm(n*nfiller, log(beta.T1surv/(1-beta.T1surv)), sigma.T1surv),
T1col=rnorm(n*nfiller, log(beta.T1col/(1-beta.T1col)), sigma.T1col),
T2=rnorm(n*nfiller, log(beta.T2/(1-beta.T2)), sigma.T2),
T2surv=rnorm(n*nfiller, log(beta.T2surv/(1-beta.T2surv)), sigma.T2surv),
T2col=rnorm(n*nfiller, log(beta.T2col/(1-beta.T2col)), sigma.T2col),
aspect=rnorm(n*nfiller, log(beta.aspect/(1-beta.aspect)), sigma.aspect),
aspectsurv=rnorm(n*nfiller, log(beta.aspectsurv/(1-beta.aspectsurv)),
sigma.aspectsurv),
aspectcol=rnorm(n*nfiller, log(beta.aspectcol/(1-beta.aspectcol)),
sigma.aspectcol),
ba=rnorm(n*nfiller, log(beta.ba/(1-beta.ba)), sigma.ba),
basurv=rnorm(n*nfiller, log(beta.basurv/(1-beta.basurv)), sigma.basurv),
bacol=rnorm(n*nfiller, log(beta.bacol/(1-beta.bacol)), sigma.bacol),
temp=rnorm(n*nfiller, log(beta.temp/(1-beta.temp)), sigma.temp),
jd=rnorm(n*nfiller, log(beta.jd/(1-beta.jd)), sigma.jd),
Z = array(rbinom(n*nsites, size=1, prob=psi.meanGuess),
dim=c(7,32,2,1))
)}
modelFilename <- "MSOMKN.txt"
###defining model and priors
cat("
model {
121
psi.mean ~ dunif (0,1)
psi.mean1 ~ dunif (0,1)
psi.mean2 ~ dunif (0,1)
beta <- log(psi.mean) - log(1-psi.mean)
beta1 <- log(psi.mean1) - log(1-psi.mean1)
beta2 <- log(psi.mean2) - log(1-psi.mean2)
b.T1 ~ dunif(0,1)
b.T1surv ~ dunif(0,1)
b.T1col ~ dunif(0,1)
b.T2 ~ dunif(0,1)
b.T2surv ~ dunif(0,1)
b.T2col ~ dunif(0,1)
b.aspect ~ dunif(0,1)
```

APPENDIX 2 (continued).

```
b.aspectsurv ~ dunif(0,1)
b.aspectcol ~ dunif(0,1)
b.ba ~ dunif(0,1)
b.basurv ~ dunif(0,1)
b.bacol ~ dunif(0,1)
b.temp ~ dunif(0,1)
b.jd ~ dunif(0,1)
mu.T1<-log(b.T1/(1-b.T1))
mu.T1surv<-log(b.T1surv/(1-b.T1surv))
mu.T1col<-log(b.T1col/(1-b.T1col))
mu.T2<-log(b.T2/(1-b.T2))
mu.T2surv<-log(b.T2surv/(1-b.T2surv))
mu.T2col<-log(b.T2col/(1-b.T2col))
mu.aspect<-log(b.aspect/(1-b.aspect))
mu.aspectsurv<-log(b.aspectsurv/(1-b.aspectsurv))
mu.aspectcol<-log(b.aspectcol/(1-b.aspectcol))
mu.ba<-log(b.ba/(1-b.ba))
mu.basurv<-log(b.basurv/(1-b.basurv))
mu.bacol<-log(b.bacol/(1-b.bacol))
mu.temp<-log(b.temp/(1-b.temp))
122
mu.jd<-log(b.jd/(1-b.jd))
sigma.u ~ dunif (0,10)
sigma.v ~ dunif (0,10)
sigma.u1 ~ dunif (0,10)
sigma.v1 ~ dunif (0,10)
sigma.u2 ~ dunif (0,10)
sigma.v2 ~ dunif (0,10)
s.T1 ~ dunif(0,10)
s.T1surv ~ dunif(0,10)
s.T1col ~ dunif(0,10)
s.T2 ~ dunif(0,10)
s.T2surv ~ dunif(0,10)
s.T2col ~ dunif(0,10)
s.aspect ~ dunif(0,10)
s.aspectsurv ~ dunif(0,10)
s.aspectcol ~ dunif(0,10)
s.ba ~ dunif(0,10)
s.basurv ~ dunif(0,10)
s.bacol ~ dunif(0,10)
s.temp ~ dunif(0,10)
s.jd ~ dunif(0,10)
tau.u <- pow(sigma.u,-2)
tau.v <- pow(sigma.v,-2)
tau.u1 <- pow(sigma.u1,-2)
tau.v1 <- pow(sigma.v1,-2)
tau.u2 <- pow(sigma.u2,-2)
tau.v2 <- pow(sigma.v2,-2)
tau.T1<-pow(s.T1, -2)
tau.T1surv<-pow(s.T1surv, -2)
tau.T1col<-pow(s.T1col, -2)
```

APPENDIX 2 (continued).

```
tau.T2<-pow(s.T2, -2)
123
tau.T2surv<-pow(s.T2surv, -2)
tau.T2col<-pow(s.T2col, -2)
tau.aspect<-pow(s.aspect, -2)
tau.aspectsurv<-pow(s.aspectsurv, -2)
tau.aspectcol<-pow(s.aspectcol, -2)
tau.ba<-pow(s.ba, -2)
tau.basurv<-pow(s.basurv, -2)
tau.bacol<-pow(s.bacol, -2)
tau.temp<-pow(s.temp, -2)
tau.jd<-pow(s.jd, -2)
rho ~ dunif(-1,1)
var.eta <- tau.v/(1.-pow(rho,2))
for(i in 1:n) {
phi[i] ~ dnorm(beta, tau.u)I(-5,5)
T1[i] ~ dnorm(mu.T1, tau.T1)I(-5,5)
T1surv [i] ~ dnorm(mu.T1surv, tau.T1surv)I(-5,5)
T1col [i] ~ dnorm(mu.T1col, tau.T1col)I(-5,5)
T2[i] ~ dnorm(mu.T2, tau.T2)I(-5,5)
lphi0 [i] ~ dnorm(beta1, tau.u1)I(-5,5)
T2surv [i] ~ dnorm(mu.T2surv, tau.T2surv)I(-5,5)
##T2 covariate for survival parameter
lgamma0 [i] ~ dnorm(beta2, tau.u2)I(-5,5)
##lgamma0 coefficient for intercept
T2col [i] ~ dnorm(mu.T2col, tau.T2col)I(-5,5)
##T2 covariate for colinization parameter
aspect[i] ~ dnorm(mu.aspect, tau.aspect)I(-5,5)
aspectsurv [i] ~ dnorm(mu.aspectsurv, tau.aspectsurv)I(-5,5)
aspectcol [i] ~ dnorm(mu.aspectcol, tau.aspectcol)I(-5,5)
ba[i] ~ dnorm(mu.ba, tau.ba)I(-5,5)
basurv [i] ~ dnorm(mu.basurv, tau.basurv)I(-5,5)
bacol [i] ~ dnorm(mu.bacol, tau.bacol)I(-5,5)
temp[i] ~ dnorm(mu.temp, tau.temp)I(-5,5)
jd[i] ~ dnorm(mu.jd, tau.jd)I(-5,5)
}
p.mean ~ dunif (0,1)
124
alpha <- log(p.mean) - log(1-p.mean)
for(i in 1:n){
mu.eta[i] <- alpha + (rho*sigma.v/sigma.u)*(phi[i] - beta)
eta[i] ~ dnorm(mu.eta[i], var.eta)I(-5,5)
}
# at logit(psi) you should add your occupancy covariates and
for logit(p) you should
#add your detection covariates
for (i in 1:n){
for (k in 1:R) {
for (t in 1:T){
for (z in 1:Q){
logit(p[i,k,t,z]) <-eta[i]+jd[i]*c.jd[k,t,z]+temp[i]*c.temp[k,t,z]
```

APPENDIX 2 (continued).

```
}}}}
#State Submodel
# initial state and likelihood for year = 1
#first year occupancy
for (i in 1:n){
for (k in 1:R) {
for (t in 1:1){
for (z in 1:Q){
logit(psi[i,k,t,z]) <-phi[i]
+T1[i]*c.T1[k,1,z]
+T2[i]*c.T2[k,1,z]
+aspect[i]*c.aspect[k,1,z]
+ba[i]*c.ba[k,1,z]
Z[i,k,1,z] ~ dbern(psi[i,k,1,z])
mu.p[i,k,1,z] <- p[i,k,1,z]*Z[i,k,1,z]
Y[i,k,1,z] ~ dbin(mu.p[i,k,1,z], J)
}}}}
##determining survival and colonization
##for year 1 to 2
for (i in 1:n){
for (k in 1:R) {
for (t in 2:2){
for (z in 1:Q){
logit(phisurv[i,k,t-1,z]) <- lphi0[i]
+T1surv[i]*c.T1[k,1,z]
+T2surv[i]*c.T2[k,1,z]
+aspectsurv[i]*c.aspect[k,1,z]
125
+basurv[i]*c.ba[k,1,z]
logit(gamma[i,k,t-1,z]) <- lgamma0[i]
+T1col[i]*c.T1[k,1,z]
+T2col[i]*c.T2[k,1,z]
+aspectcol[i]*c.aspect[k,1,z]
+bacol[i]*c.ba[k,1,z]
mu[i,k,t,z]<-Z[i,k,t-1,z]*phisurv[i,k,t-1,z]
+ (1-Z[i,k,t-1,z])*gamma[i,k,t-1,z]
Z[i,k,t,z] ~ dbern(mu[i,k,t,z])
mu.p[i,k,t,z] <- p[i,k,t,z]*Z[i,k,t,z]
Y[i,k,t,z] ~ dbin(mu.p[i,k,t,z], J)
}}}}
# Derived parameters: Compute annual occupancy and growth
rates
for (i in 1:n){
psivec[i,1]<-mean(psi[i,1:R,1,1])
pvec[i,1]<-mean(p[i,1:R,1:T,1])
phiyr[i,1]<-mean(phisurv[i,1:R,1,1])
gammayr[i,1]<-mean(gamma[i,1:R,1,1])
}
for(i in 1:n){
for(j in 2:2){
psivec[i,j] <- psivec[i,j-1]*phiyr[i,j-1]
```

APPENDIX 2 (continued).

```
+ (1-psivec[i,j-1])*gammayr[i,j-1]
growthr[i,j]<- psivec[i,j]/psivec[i,j-1]
turnover[i,j-1]<- ((1 - psivec[i,j-1]) * gammayr[i,j-1])/( (1 -
psivec[i,j-1]) * gammayr[i,j-1]
+ phiyr[i,j-1]*psivec[i,j-1])
eop[i,j-1]<- (gammayr[i,j-1])/( (gammayr[i,j-1])
+ (1-phiyr[i,j-1]))
}}}
", fill=TRUE, file=modelFilename)
#fit model to data using WinBUGS code, make sure to have
WinBUGS installed
#with the appropriate key decoded
library(R2WinBUGS)
fit = bugs(data, inits, parameters.to.save=params,
model.file=modelFilename,
n.chains=5, n.iter=55000, n.burnin=5000, n.thin=50,
bugs.seed=sample(1:9999,size=1), debug=T, DIC=F, coda=T)
126
end.time=Sys.time()
elapsed.time= difftime(end.time, start.time, units='mins')
cat(paste(paste('Posterior computed in ', elapsed.time, sep=''), '
minutes\n', sep=''))
```

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.

OAK MAST PRODUCTION AND ANIMAL IMPACTS ON ACORN SURVIVAL IN THE CENTRAL HARDWOODS

Kenneth F. Kellner, Jeffery K. Riegel, Nathanael I. Lichti, and Robert K. Swihart[1]

Abstract.—As part of the Hardwood Ecosystem Experiment we measured mast production in white (*Quercus alba*) and black (*Q. velutina*) oak, and quantified the impacts of seed predators on acorn survival over a 3-year period. Specifically, we measured the proportion of acorns of each species infested with weevils (*Curculio* spp.), and the probability of acorn removal by seed predators from a system of semipermeable exclosures. The 3 years of the study included 2 years of high mast production in both species (2006-07) and 1 year of mast crop failure (2008). Across all 3 years, 19 percent of acorns were infested. The rate of weevil infestation was slightly higher for black oak than for white oak in each year, but infestation peaked for both species during the year of mast failure (2008). Overall, 39 percent of acorns in the exclosures were removed by seed predators. The probability of acorn removal was lower when squirrels were excluded, providing support for additive effects of different seed predators. The probability of removal was highest in 2008 during the year of mast failure. In the future, these pre-harvest data will be compared to data obtained following timber harvests conducted in winter 2009.

INTRODUCTION

Oak (*Quercus* L.) is a dominant overstory species group in the Central Hardwoods Region. Oaks have been labeled as both a keystone and "foundation" species in eastern deciduous forests (Ellison et al. 2005, Fralish 2004), performing a wide variety of functions in forest ecosystems. For example, oak-dominated forests promote biodiversity in the herbaceous understory because oak branch structure allows a large amount of sunlight to reach the forest floor (Fralish 2004, Horn 1971). Oaks also are known to provide habitat for many species of insects, fungi, and vertebrates and play an important role in hydrology and nutrient cycling (Brändle and Brandl 2001, Johnson et al. 2002). Among the most important functions of oak in the Central Hardwoods is the production of hard mast, an important food source for at least 44 species of birds, small mammals, and larger vertebrates such as white-tailed deer (*Odocoileus virginianus* Zimmerman) and black bear (*Ursus americanus* Pallus) (McShea et al. 2007). The importance of oak as a source of hard mast has greatly increased in the past century due to the decline of American chestnut (*Castanea dentata* [Marsh.] Borkh.) (Dalgleish and Swihart [in press], Diamond et al. 2000, McShea et al. 2007).

Oaks generally are shade-intolerant species, requiring disturbance to regenerate effectively (Larsen and Johnson 1998). An active cycle of natural and/or anthropogenic fire disturbance is thought to have promoted oak dominance in the Central Hardwoods before European settlement (Abrams 1992). Following settlement, cycles of land clearing for agriculture and subsequent abandonment maintained oak presence in

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the overstory (Fralish 1997). However, the advent of fire suppression and the creation of protected national and state parks in the 20th century have greatly reduced the frequency of disturbance in eastern forests (Abrams 1992, 2003). The result has been regeneration failure of oaks across the Central Hardwoods Region, as well as shifts in species assemblages within the oak genus (Abrams 2003, Aldrich et al. 2005). This altered disturbance regime favors the establishment of shade-tolerant climax species such as maple (*Acer* L.) and American beech (*Fagus grandifolia* Ehrh.) (Fralish 2003).

Loss of oak as a canopy dominant would have important ecological and economic impacts (Johnson et al. 2002, McShea et al. 2007). As a result, researchers have begun to develop forest management (i.e., timber harvest) strategies to promote oak regeneration (Dey and Parker 1996, Dey et al. 2008, Loftis 1990). Examples of management include even-aged methods (clearcuts and shelterwood harvests) and uneven-aged methods (group-selection or single-tree selection harvests). Management efforts have met with mixed success; ecological variables such as soil quality and moisture likely play a role in the outcome (Dey et al. 2009). Oak regeneration success is also heavily influenced by animals, including acorn weevils, small mammal seed predators, and deer (Marquis et al. 1976).

Weevils (*Curculio* L.) infest the acorns of all oak species in the Central Hardwoods. In a given year, as many as 50-90 percent of all acorns produced may be infested (Gribko 1995, Lombardo and McCarthy 2008, Marquis et al. 1976, Riccardi et al. 2004). Infested acorns are both less likely to be dispersed (Steele et al. 1996) and less likely to germinate successfully (Andersson 1992, Lombardo and McCarthy 2009). The interaction of oaks with small mammal seed predators is more complex. A large percentage of fallen acorns are eventually removed by seed predators such as white-tailed deer, gray squirrels (*Sciurus carolinensis* Gmelin), eastern chipmunks (*Tamius striatus* L.) and mice (*Peromyscus* Gloger) (McShea 2000, McShea et al. 2007). However, some seed predators (e.g., *S. carolinensis*) make many small caches of acorns, which may promote germination success (Barnett 1977, Smallwood et al. 2001, Steele et al. 2006).

Regardless of animals' influence on oak regeneration success, little is known about the impact of timber harvest strategies on oak mast production and subsequent predation by insects and small mammals. The Hardwood Ecosystem Experiment (HEE), a long-term replicated study in Indiana of forest ecosystem responses to timber harvest, provides an excellent experimental framework to address this knowledge gap. Our objectives in this study were to compare mast production by black (*Q. velutina* Lamb.) and white (*Q. alba* L.) oaks at several sites in southern Indiana over a 3-year period, and subsequently to assess the impacts of acorn weevils and seed predators on the acorn crop. Following the final year of this preliminary study, the experimental sites were harvested under several different management strategies with the goal of improving oak regeneration. Ultimately, our results will serve as a baseline for identifying changes in mast production and seed predation by weevils and small mammals following the silvicultural treatments.

Prior to the application of silvicultural treatments, we tested several hypotheses. We expected that acorn production would vary among the 3 years of the study, and that production would be different between the two oak species. Oaks are a masting species group, synchronizing with other trees in a region to produce very large or very small mast crops (Janzen 1971), but often lacking interspecific synchrony (Abrahamson and Layne 2003, Lombardo and McCarthy 2008, Lusk et al. 2007). We also expected that black and white oaks may be affected differently by acorn weevils; Lombardo and McCarthy (2008) found that a higher proportion of acorns in the red oak section (Lobatae) were infested than those in the white oak section (Quercus), but few studies have compared infestation levels between oak species.

To address the impacts of predators on the fate of fallen seeds, we sought to isolate the individual contributions of several acorn predators on the total amount of removals observed, in order to assess if those contributions were additive or compensatory. We expected that gray squirrels would be most prolific at removing fallen acorns (Bellocq et al. 2005). In addition, we expected that less desirable acorns (i.e., acorns that were broken, germinated, or infested with weevils) would be less likely to be taken by small mammals (Smallwood et al. 2001, Steele et al. 1996).

STUDY AREA AND DESIGN

A summary of site characteristics and the selection of research areas can be found in a previous chapter of this report (Kalb and Mycroft, this publication). Following the delineation of the research core areas, we selected mature black and white oak trees to be included in the study. We chose this pair of oak species because they are among the most dominant tree species in the region (Jenkins and Parker 1998), and to ensure that both oak sections were represented. Individual trees of reproductive age were chosen based on their location relative to future harvests.

The number and location of selected trees depended on the assigned management treatment of the unit. In each of the three control units, six trees of each species were selected, with two near the center of the research core and four >100 m away near the core edge. In the three units assigned uneven-aged management, six trees of each species were selected, with all trees adjacent to (but outside) the proposed 0.40-, 1.21-, and 2.02-ha patch cuts. In the even-aged units, the arrangement of trees differed between harvest types. Each even-aged unit contained four trees of each species associated with shelterwood harvests; half were adjacent to the future shelterwoods and half were selected as overstory trees within the shelterwoods to be retained during the harvest. Each even-aged unit also contained two trees of each species located adjacent to clearcut areas.

Across all experimental units, 108 trees were sampled in 2006, divided evenly among the three harvest treatments. In 2007, an additional experimental unit was added in Brown County State Park; four additional control trees were selected in this unit for a total of 112 trees sampled. Due to active timber harvesting in uneven- and even-aged units in the fall of 2008, 20 trees could not be sampled, yielding a sample size of 96 in the third year of the experiment. Overall, black and white oaks had diameters at breast height (d.b.h.) of 20.0±4.4 cm (mean ± standard deviation) and 19.5±4.5 cm, respectively.

MATERIALS AND METHODS

Mast Production

At each tree, two mast collection traps were established. Traps consisted of a 52cm × 33cm × 32cm plastic bin mounted atop a 2-m high polyvinyl chloride pipe driven into the ground. Bins were covered with chicken wire to prevent pilferage of seeds by animals while still allowing mast to fall into the trap. Traps were placed midway between the trunk and canopy edge underneath a limb. For trees adjacent to proposed harvest areas, one mast trap was placed on the side of the tree facing the harvest and one on the opposite side. For all other trees, one mast trap was oriented to the north of the trunk and the other to the south.

In the first year of the study (2006), traps were set up in October and checked five times at weekly intervals until the beginning of December. In 2007 and 2008, traps were set up in late August and checked eight times at 1- to 2-week intervals until December. Some traps were not sampled in uneven-aged units in 2008 due to ongoing timber harvests in the area. At each trap check, all acorns in the collection buckets were counted, identified to species, and removed.

Weevil Infestation

The majority of acorns removed from mast collection bins were examined for weevil infestation. In 2007 and 2008, a small number of additional acorns

were removed from the forest floor underneath the mast trees to increase the sample size. Acorns were X-rayed to identify damage caused by weevils (Dixon et al. 1997, Lombardo and McCarthy 2009). Acorns were marked according to source tree and collection date and X-rayed in groups of 50 using a Specimen Radiography X-ray System (Faxitron X-ray Corporation, Lincolnshire, IL). Weevil damage is indicated on the X-ray film by characteristic dark patterns within the acorn (Fig. 1; see Lombardo and McCarthy 2009: Fig. 1). We classified an acorn as infested if any portion of the acorn interior exhibited weevil damage.

Acorn Removal

A set of four semipermeable exclosures was established underneath each mast tree to assess the impacts of several groups of acorn predators. In the study forests, acorn predators include white-tailed deer, gray squirrels (and a few fox squirrels, *S. niger* L.), eastern chipmunks, and white-footed mice (*P. leucopus*). Each exclosure was a 0.75m × 0.75m square. The first exclosure allowed access to all wildlife species (deer, squirrels, chipmunks, and mice or D+S+C+M) and simply consisted of four wooden stakes designating a 0.75m × 0.75m area on the ground. The second excluded deer only (allowing access to S+C+M) and consisted of four wooden stakes covered on the sides and top by 3.8-cm wide hexagonally meshed chicken wire. The wire began 15 cm above the ground to allow access to squirrels and smaller wildlife. The third exclosure excluded squirrels and larger wildlife but allowed access to mice and chipmunks (C+M) and was a 0.75m × 0.75m × 0.2m wooden frame covered in 3.8-cm chicken wire. The final control exclosure was designed to prevent access by all vertebrate wildlife (A) and was a wooden frame of the same size but covered instead with 0.6-cm mesh hardware cloth.

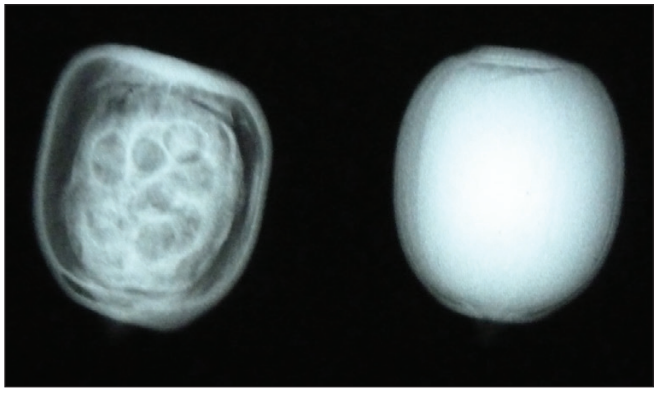


Figure 1.—X-ray of two black oak acorns; the acorn on the left is infested with weevil larvae and the acorn on the right is sound.

The four exclosures were arranged randomly on a north-to-south transect bisecting the trunk of the mast study tree. Two exclosures were placed to the north of the tree and two to the south. On each sampling occasion, acorns that had fallen into each exclosure were located and numbered with a black marker. It was impossible for acorns to fall naturally into the control exclosures, so the A exclosures were provisioned with a number of acorns equal to the average number inside the other three. On subsequent visits, the presence or absence of the marked acorns was determined. If an acorn was present, information about the acorn including the presence of weevil exit holes, germination status, and integrity of the acorn (broken or intact) was recorded.

ANALYSIS

Mast Production

The total number (count) of acorns collected at a given tree during a given sampling occasion was the response variable in this analysis. Overdispersion of these count data, with a variance ~5 times greater than the mean value, prevented analysis with simple Poisson regression. Instead, we fit a Poisson-lognormal model (Kéry 2010) which introduced an additional free parameter to allow the variance of the distribution to differ from the mean. Briefly, we modeled $C_{i,j,k}$, the observed acorn count at tree i during sampling period j of year k as $C_{i,j,k} \sim \text{Poisson}(\lambda_{i,j,k})$. The linear predictor for the Poisson intensity parameter λ was $\lambda_{i,j,k} = exp(\beta_q X_{q,ijk} + \varepsilon_{i,j,k})$ where β is a vector of slope parameters, X is a matrix of q covariate values,

and ε is distributed N(0, σ^2), to allow the variance to vary independently from λ. We included covariates for tree species, d.b.h., the length and date of sampling periods (in Julian day format), and proposed treatment in the linear predictor. In addition to the random error ε, we considered tree and year to be random effects.

Weevil Infestation

We estimated the effects of acorn-specific covariates on the infestation status z_i of individual collected acorns. Infestation z_i was modeled as $z_i \mid p_i \sim$ Bernoulli(p_i), where logit(p_i) is a linear function of covariates. The linear predictor included year, Julian day of sampling, Julian day of sampling centered squared, proposed management technique for the source site of acorn *i*, species of acorn *i* as fixed effects, and source tree as a random effect.

Acorn Removal

We related site- and acorn-specific covariates to the probability $\Phi_{i,j,k}$ that acorn *i* would be removed between sampling occasions *j-1* and *j* in year *k*. The presence $z_{i,j,k}$ of a given acorn *i* in an exclosure at sampling occasion *j* in year *k* took on a value of 0 or 1 and was modeled conditional on the acorn's state at the previous sampling occasion $z_{i,j-1,k}$ and probability of removal $\Phi_{i,j,k}$; that is, $z_{i,j,k} \mid z_{i,j-1,k}, \Phi_{i,j,k} \sim$ Bernoulli($\Phi_{i,j,k}$). Logit($_{\Phi i,j,k}$) was set equal to a linear predictor containing acorn- and site-specific covariates. Explanatory variables considered in this analysis were the fixed effects of Julian day of sampling for occasion *j*, Julian day of sampling centered squared, exclosure type for acorn *i*, proposed management treatment for acorn *i*, and the germination, weevil infestation, and shell status (i.e., broken or intact) of acorn *i* at time *j-1*. Length of time between sampling periods *j-1* and *j* was considered as a nuisance variable, and source tree and year were included as random effects.

Model Fitting

All regression models were fit in a Bayesian framework with non-informative priors. Slope parameters were assigned a normal prior distribution with mean 0 and variance 1,000, and variance parameters were assigned a lognormal distribution with mean 0 and variance 1 following Kéry (2010). All covariates were standardized by subtracting the mean and dividing by the standard deviation. Bayesian analyses were conducted using the software package WinBUGS (version 1.4.3; Cambridge, UK) (Spiegelhalter et al. 1996), which uses the Markov Chain Monte Carlo (MCMC) method to generate posterior distributions for the model parameters. WinBUGS was called from within R 2.10.0 (R Foundation for Statistical Computing, Vienna, Austria) using the R2WinBUGS library (Sturtz et al. 2005).

MCMC simulations were conducted with a minimum of 3 chains, 30,000 iterations, and a burn-in of 10,000 iterations. Convergence was assessed by calculating the Gelman-Rubin diagnostic Rhat (Brooks and Gelman 1998) for each estimate parameter. If we did not observe convergence in the distributions of all posterior parameters (Rhat > 1.1), MCMC chain length and burn-in were increased until convergence was achieved. Posterior predictive checks were performed on each regression model to assess goodness of fit; for the mast production model Pearson's chi-square residuals were compared between the actual data and data simulated based on the model, and for the other two analyses the sums of absolute residuals were compared. Bayesian p-values were calculated for each posterior predictive check. An individual covariate was considered to have an important effect if the 95-percent credible interval of its slope parameter did not contain zero.

RESULTS

Mast Production

We collected 1,140 acorns over the 3 years of the experiment (Table 1). The Poisson-lognormal model had acceptable fit; a posterior predictive check yielded a Bayesian p-value of 0.41 (0.5 is ideal).

Table 1.—Number of trees sampled and acorn sample sizes for the mast production, weevil infestation, and mast removal experiments in 2006-08.

Parameter	2006	2007	2008	Total
Trees sampled	108	112	96	
Total acorns collected in traps	355	670	115	1,140
Black oak (*Q. velutina*)	235	414	9	658
White oak (*Q. alba*)	120	256	106	482
Total acorns x-rayed for weevils	366	595	114	1,075
Black oak	265	423	8	696
White oak	101	172	106	379
Total acorns monitored for removal	1,041	1,827	240	3,108
Black oak	468	1,013	30	1,511
White oak	573	814	210	1,597

In general, white oaks produced fewer acorns than black oaks in our study area (Fig. 2), except in 2008, when there was a mast failure in black oak. The 95-percent credible interval for the species coefficient in the Poisson regression included zero (Table 2). However, 74 percent of the posterior distribution of the species coefficient was below zero, providing marginal support for lower production by white oak. Production by individual trees was positively correlated with d.b.h.; larger trees produced more acorns. Counts were negatively correlated with both Julian day of sampling and Julian day squared, indicating that production was highest early in the sampling period (September-October), declining later in the season. Proposed harvest treatment had no effect on acorn production among the trees we sampled.

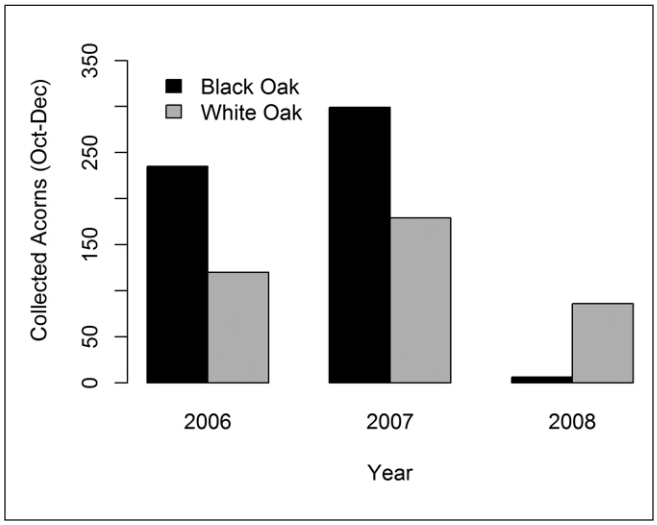


Figure 2.—Total acorns collected by species and year. Since sampling began in October 2006, only acorns collected in the months of October through December are included to allow direct comparisons between years.

Table 2.—Estimated parameter values for the Poisson-lognormal model of oak mast production. The value *f* represents the proportion of the posterior distribution for the parameter which has the same sign (positive or negative) as the mean. Parameters with 95-percent credible intervals which do not include zero are marked with an asterisk (*).

Parameter	Covariate type	Mean	SE	*f*
Intercept		-3.21	0.98	1.00
Tree species (WO=1)	indicator	-0.17	0.26	0.74
d.b.h.	continuous	0.33*	0.13	0.99
Sample day (Julian)	continuous	-1.06*	0.12	1.00
Julian day squared	continuous	-1.75*	0.13	1.00
Even-aged treatment effect	indicator	-0.46	0.31	0.93
Uneven-aged treatment effect	indicator	0.12	0.32	0.64

Weevil Infestation

A total of 1,075 acorns were X-rayed to identify weevil infestation (Table 1). The number of X-rayed acorns does not correspond exactly to the number removed from buckets because small numbers of acorns were lost following collection and/or added from the ground beneath the bucket. A posterior predictive check of the logistic regression model fit to the data yielded an acceptable Bayesian p-value of 0.39. Black oak acorns had a higher probability of infestation in all three sample years compared to white oak (Fig. 3, Table 3).

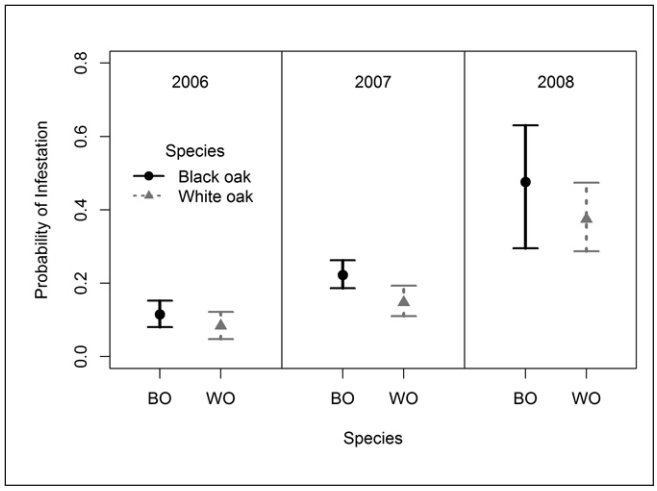


Figure 3.—Posterior distributions for the mean probabilities of acorn infestation, by acorn species and year. Error bars represent 95-percent credible intervals.

Infestation probability increased from 2006 to 2008 (Table 3). The high probabilities of infestation in both species in 2008 (Fig. 3) coincided with low mast production in white oak and a mast failure in black oak in 2008 (Fig. 2). In addition to the year effects, there was a negative correlation between Julian day of sampling and probability of infestation (Table 3); acorns collected later in the fall were less likely to have weevil damage. There were no differences in weevil infestation between areas selected to receive different silvicultural treatments.

Acorn Removal

Over the 3-year study period, 3,108 acorns were marked and monitored for removal (Table 1). Of these, 1,200 (39 percent) were removed from the exclosures at some point: 595 of 1,511 black oak acorns (39 percent) and 605 of 1,597 white oak acorns (38 percent). We used logistic regression to model the probability that an individual acorn was removed during a given time period [j-1, j]. Our model fit the data adequately, with a posterior predictive check of absolute residuals yielding a Bayesian p-value of 0.38.

There were no differences in the probability of removal between acorn species. However, acorn damage, weevil infestation, and germination were all negatively correlated with the probability of acorn removal (95-percent credible intervals excluding zero; Table 4). There were no differences in probability of removal between proposed treatment areas.

Table 3.—Estimated parameter values for the model of probability of acorn infestation. The value *f* represents the proportion of the posterior distribution for the parameter which has the same sign (positive or negative) as the mean. Parameters with 95-percent credible intervals which do not include zero are marked with an asterisk (*).

Parameter	Covariate type	Mean	SE	*f*
Intercept		-2.02	0.26	1.00
2007 year effect	indicator	0.72*	0.22	1.00
2008 year effect	indicator	1.96*	0.34	1.00
Acorn species (WO=1)	indicator	-0.52*	0.25	0.98
Sample day (Julian)	continuous	-0.18*	0.094	0.98
Julian day squared	continuous	0.015	0.084	0.58
Even-aged treatment effect	indicator	-0.084	0.27	0.62
Uneven-aged treatment effect	indicator	-0.071	0.26	0.62

Table 4.—Estimated parameter values for the model of probability of acorn removal by predators. The value f represents the proportion of the posterior distribution for the parameter which has the same sign (positive or negative) as the mean. Parameters with 95-percent credible intervals which do not include zero are marked with an asterisk (*).

Parameter	Covariate type	Mean	SE	*f*
Intercept		-4.34	0.60	0.99
Date and harvest treatment parameters				
Sample day (Julian)	continuous	0.049	0.055	0.82
Julian day squared	continuous	0.24*	0.065	1.00
Time between samples	continuous	-0.055	0.038	0.92
Even-aged treatment effect	indicator	0.12	0.26	0.67
Uneven-aged treatment effect	indicator	-0.011	0.27	0.52
Acorn characteristics				
Acorn species (WO=1)	indicator	-0.082	0.22	0.66
Acorn integrity (interact=1)	indicator	-0.48*	0.15	1.00
Germination status (germ=1)	indicator	-0.49*	0.15	1.00
Weevil status (infested=1)	indicator	-0.78*	0.21	1.00
Exclosure-type parameters (baseline = completely inaccessible exclosure)				
Squirrel and larger exclosure	indicator	3.39*	0.24	1.00
Deer and larger exclosure	indicator	4.09*	0.24	1.00
Open exclosure	indicator	3.91*	0.24	1.00

Acorns in all three exclosures accessible to at least some vertebrates (open, deer, and squirrel exclosures) had a higher probability of removal than acorns in the control, inaccessible exclosure (Table 4, Fig. 4). Differences between the three accessible exclosure types were more difficult to ascertain. The posterior distributions for the exclosure regression coefficients overlapped (Fig. 5), with the coefficient corresponding to squirrel exclosures having a smaller mean value, corresponding to a lower probability of removal.

In 2007 and 2008, the 95-percent credible interval for removal probability from squirrel exclosures did not overlap with the open or deer exclosures credible intervals, providing further evidence that excluding access to squirrels reduced the probability of acorn removal (Fig. 4). In the same years, probability of removal was higher from the exclosures that excluded only deer. In general, probabilities of removal were higher in 2008, corresponding to reduced mast production (Figs. 2 and 4).

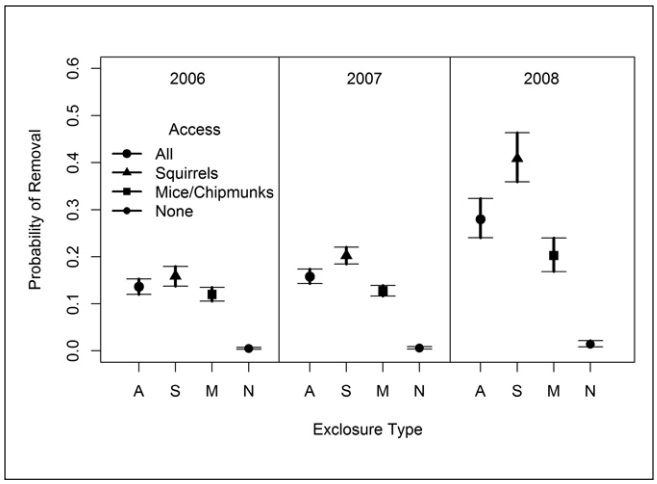


Figure 4.—Mean probabilities of removal from each exclosure type (for an individual sampling period) during the 3 years of the study. Exclosure O was open to all animals, D excluded deer, S excluded everything squirrel-sized and larger, and A (the control) excluded all vertebrates. Error bars represent 95-percent credible intervals.

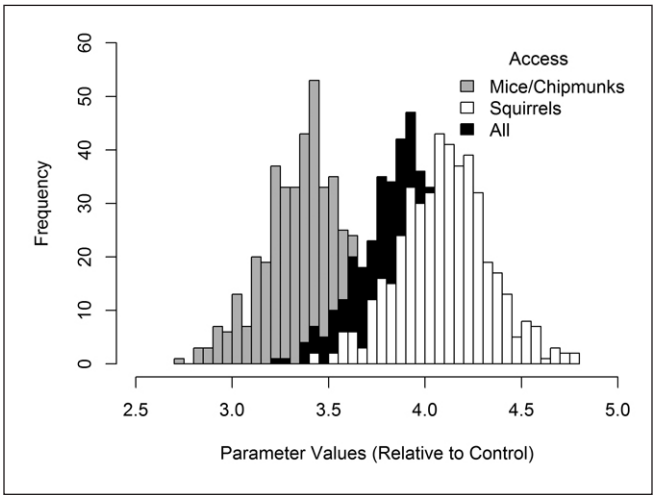


Figure 5.—Comparison of posterior distributions for the three exclosure-type parameters. The three parameter values are relative to removal from a control, i.e., a completely inaccessible exclosure.

DISCUSSION

Mast Production

This 3-year study included 2 years of abundant mast (2006-07) and 1 year of mast failure in black oak and partial mast failure in white oak (2008; Fig. 2). More black oak acorns were collected per sampling occasion (traps × sample periods) in the non-failure years, indicating that individual black oaks in our study region may be more valuable than white oaks for wildlife food production. This result is consistent with other studies that found black oak to generally be a larger producer of acorns than chestnut oak (*Q. prinus*) and white oak (Lombardo and McCarthy 2008, Sork and Bramble 1993, Sork et al. 1993).

Overall, however, our model did not identify a species effect that was statistically different from zero (Table 2). In the short duration of our study, production by black oak appeared to be more variable than white oak (Fig. 2), though the literature suggests it should be a consistent producer (Lombardo and McCarthy 2008, Sork and Bramble 1993, Sork et al. 1993). A longer record of production is necessary to accurately compare variability in production of acorns between these species. In addition, our study did not incorporate population-level information on oaks in the study area; consequently, we cannot make conclusions about the overall importance of the two oak species at our sites.

Though white and black oaks are common at our study sites, it is important to note that the total amount of mast available also depends on other hardwood species (especially chestnut oak, but also Northern red oak, *Q. rubra*, and the hickories, *Carya*). We did not monitor mast production by these species but because there is often a lack of interspecific synchrony in masting (Abrahamson and Layne 2003, Lombardo and McCarthy 2008, Lusk et al. 2007), they may buffer the total mast crop available for wildlife in the study area.

Weevil Infestation

We found higher rates of weevil infestation in black oak than in white oak in each year of the study (Fig. 3), and overall there was a significant species effect for white oak (Table 3)). Our results are consistent with the findings of Lombardo and McCarthy (2008), who found higher rates of infestation in black oak than in chestnut oak (which belongs to the white oak section) in southeastern Ohio. Black oak may be a more consistent mast producer than chestnut oak (Lombardo and McCarthy 2008) and white oak (Sork and Bramble 1993, Sork et al. 1993), which may allow the black oak population to support a larger number of weevils. A limitation of this study is that weevils were not identified to the species level, and different members of the genus *Curculio* appear to specialize on certain oak species (Gibson 1972, 1982). It is possible that the particular assemblage of individual *Curculio* species at our study sites is responsible for higher infestation in black oak. Nearly all acorns were collected from raised buckets, so we do not believe many of the acorns were infested with weevils in the genus *Conotrachelus* (Schonherr), which generally lays eggs in previously infested or broken acorns that have already fallen to the ground (Gibson 1964).

While there were year effects for both 2007 and 2008 (Table 3), infestation rates were highest for both species in 2008 (Fig. 3), corresponding to a mast failure in black oak and a reduced mast year in white oak (Fig. 2) across the study sites. The previous 2 years were good mast years for both white and black oak (Fig. 2), likely resulting in a large population of Curculio larvae undergoing diapause in the soil. When this population emerged from the soil in fall 2008, only a few acorns were available, resulting in very high rates of infestation. Lombardo and McCarthy (2008) identified a similar pattern in chestnut oak in southeastern Ohio. They found 86 percent of chestnut oak acorns were predated by weevils in a non-mast year, but only 26 percent were infested in a mast year. Unlike our study, however, they found black oak to have relatively consistent weevil predation regardless of the quantity of mast produced.

The mean probabilities of black oak acorn infestation that we observed across the 3 years of the study (0.12-0.48 for black oak, and 0.08-0.38 for white oak) were generally lower than in similar studies. For example, Lombardo and McCarthy (2008) reported that on average 79 percent of black oaks were infested over a 4-year period. Riccardi et al. (2004) found that 55 and 34 percent of black oaks were infested, respectively, in the 2 years of their study. The lower numbers we report may be due to differences in the assemblage of acorn weevil species between sites. Differences in host tree and weevil predator densities may also have an impact on weevil populations. For example, Anderson and Folk (1993) found that small mammal predators, especially white-footed mice and short-tailed shrews (*Blarina brevicauda* Say) reduced overwinter survival of acorn weevils in Indiana.

Acorn Removal by Predators

Overall, 39 percent of acorns were eventually removed from the exclosures across the 3 years of the study. Seed predators did not seem to preferentially remove either acorn species, but avoided acorns that were damaged, weevil-infested, or germinated (Table 4). Previous studies have demonstrated that gray squirrels, the primary dispersal agent in our system, are sensitive to acorn condition and are less likely to cache infested acorns (Steele et al. 1996). In addition, because white oak begins to germinate in the fall, squirrels are less likely to cache them for later use than acorns from the red oak section (Smallwood et al. 2001). The most likely explanation for this behavior is that broken, infested, and germinated acorns are more perishable and therefore less valuable to store for the winter months. Infested or broken acorns may still be removed and eaten immediately by the predator (Steele et al. 1996), but in years when mast is plentiful (2006-2007 in this study), predators may ignore less desirable acorns that are not intact. In the year of mast failure (2008), removal probability during a given sample period was very high (Fig. 4), and nearly all fallen acorns were removed by the end of the experiment regardless of their condition. The design of this experiment did not allow us to ascertain the fate of acorns post-removal, so we were unable to identify differences in the ultimate fate of acorns of varying species and conditions.

The system of semipermeable exclosures used in this study allowed us to identify the impacts of several groups of acorn predators on overall removal rates. When deer and turkeys (*Meleagris gallopavo*) were excluded, removal probabilities actually increased in each year (compare exclosures O and D in Figure 4), though differences between the two exclosures were not different from zero (Table 4). Posterior distributions for the slope parameters of the deer and open exclosure indicator variables overlapped almost completely (Fig. 5). Therefore, large animals like deer do not appear to be an important predator of acorns in this system as squirrels and smaller predators completely compensate for their effect when they are excluded. However, any predation by deer is a dead end for acorns, whereas removal by squirrels and other small mammals could result in either consumption or caching and later germination. Bellocq et al. (2005) also observed that removal rates were higher when deer were excluded than when they had access to the acorns, further evidence that deer are not an important

seed predator in our forest system. Increased rates of removal when deer are excluded appear to be counter-intuitive; fewer animals in total have access to the acorns. One possible explanation is that the deer exclosure itself provided shelter from airborne predators, increasing the probability that squirrels and mice would forage inside the exclosure.

We did not observe a statistically significant difference in removal probability when squirrels were excluded (Table 4), but the trend (Fig. 4) indicates that removal probability was reduced; the posterior distributions for the slope parameters corresponding to the deer and squirrel exclosures did not overlap greatly (Fig. 5). Differences in small mammal community composition and demographics (e.g., very high mouse and chipmunk populations) could be the reason we did not observe an additive effect of squirrel predation as strong as the one observed by Bellocq et al. (2005).

Treatment Effects and Predictions

One of the primary objectives of this study was to identify any differences in tree-level mast production and predation between HEE sites prior to applying the silvicultural treatments. In general, we did not observe many differences between trees in areas with different future management treatments. The parameters corresponding to proposed uneven- and even-aged treatment effects on probability of weevil infestation and probability of acorn removal were not statistically different from zero (Tables 3 and 4). Mast production was similar between trees in future control, uneven-, and even-aged management sites (Table 2). However, tree-level mast production was slightly lower at future even-aged sites (95 percent of the posterior distribution for the even-aged treatment parameter was less than zero).

Mast production in oaks can be highly variable from year to year (Sork et al. 1993), so it is possible that observed differences in production between trees are due to the short duration of the study. All sites at the HEE are relatively close together so it is unlikely that weather or site quality is responsible for lower production in trees at future even-aged treatment stands. Unfortunately, we are unable to scale our results up from tree-level to site-level comparisons because we did not account for population-level differences in oak between sites in this study. Still, these results provide a basis for attributing future observed differences in tree-level mast production and predation to silvicultural treatments, rather than to pre-existing site characteristics.

We anticipate that timber harvest with the goal of regenerating oak will increase acorn production. Harvesting at the clearcut and patch cut sites will increase light penetration to oaks on the edge of the cut, which will increase branch density and therefore acorn production (Johnson 1994, Verme 1953). However, these acorns play a minimal role in regeneration inside the clearcut (4.05 ha) and patch cut (0.40, 1.21, and 2.02 ha) harvest sites. In general, the primary dispersal agents in this system (small mammals and the blue jay, *Cyanocitta cristata* L.) do not move acorns deep into recently harvested areas (Nixon et al. 1980). More importantly, any seedlings that do develop from dispersed acorns following harvest will not be competitive without silvicultural intervention (Sander 1972). The primary source of regeneration in the harvested areas will be seedlings of sufficient size (>1.4 m tall) established prior to harvest and stump sprouts (Sander 1972, Sander et al. 1984).

Dominant and codominant oaks are retained in the overstory during the first phase of the shelterwood method for regeneration (Loftis 1990). Removal of the midstory and some competing trees should increase light availability for oaks in the overstory, increasing acorn production (Johnson 1994). Evidence in the literature is inconclusive. Thinning had a positive effect on red oak acorn production in New England (Healy 1997) but had minimal effects on chestnut and black oak production in Ohio (Lombardo and McCarthy 2008). Bellocq et al. (2005) found that production of red oak in Ontario appeared to increase when visual estimates were used, but no differences between treatments were observed in the number of

acorns collected in traps. Responses in production to shelterwood harvests may be species-specific (Lombardo and McCarthy 2008); in addition, any increases in production due to increased light may have begun after the time period following harvest covered by these studies.

Currently, little is known about the effects of harvest treatments on vertebrate and invertebrate acorn predators. Thinning and shelterwood harvests do not appear to change weevil infestation rates (Bellocq et al. 2005, Lombardo and McCarthy 2008). Weevil infestation of acorns produced by trees on the edge of clearcuts and patch cuts is also unlikely to be affected. Predation by small mammals following harvest likely depends on population levels relative to the amount of mast available. Bellocq et al. (2005) found no differences in rates of acorn removal by predators between shelterwoods and control stands, but there were also no treatment effects on the abundance of small mammals. A limitation of this and previous studies is that the ultimate fate of removed acorns was not determined. Future work should focus on identifying differences in seed fate between treatments, since removal could have either a positive (if the acorn is cached and forgotten) or negative (if the acorn is eaten) effect on seedling establishment.

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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.

BATS OF THE HARDWOOD ECOSYSTEM EXPERIMENT BEFORE TIMBER HARVEST: ASSESSMENT AND PROGNOSIS

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Abstract.—Before experimental harvest of the Yellowwood (YW) and Morgan-Monroe (MM) State Forests (Indiana) as part of the Hardwood Ecosystem Experiment, bats were sampled using mist nets at four locations in MM and five locations in YW during each summer 2006 through 2008. Netting locations were adjacent to forest stands scheduled for experimental manipulations following conclusion of netting in 2008. This effort produced 342 bats (in order of abundance): northern myotis (*Myotis septentrionalis*), eastern red bat (*Lasiurus borealis*), big brown bat (*Eptesicus fuscus*), Indiana myotis (*M. sodalis*), little brown myotis (*M. lucifugus*), tri-colored bat (*Perimyotis subflavus*), hoary bat (*L. cinereus*), and silver-haired bat (*Lasionycteris noctivagans*). These data provide a baseline to understand how bats are affected by short- and long-term forest manipulations initiated in summer 2008.

INTRODUCTION

Many bats rely on forested habitats for foraging and roosting (Barclay and Brigham 1991, Barclay and Kurta 2007). Although bats are a critical element of forests, an understanding of how bats are affected by many forest management practices remains elusive (Lacki et al. 2007). Most research has addressed short-term responses of bats to common management techniques, such as thinning (Fisher and Wilkinson 2005, Kurta and Kennedy 2002, Miller 2003, Tibbels and Kurta 2003), prescribed burning (Boyles and Aubrey 2006, Boyles et al. 2005, Carter et al. 2002), and clearcutting (Hogburg et al. 2002, Owen et al. 2004, Patriquin and Barclay 2003). However, effects of other commonly used techniques on bats, especially single-tree selection and patch cutting, remain relatively unknown. Further, little is known about long-term effects of forest management practices on bats. Finally, past research involving bats typically considered only one management activity at a time, even when conducted within forest ecosystems receiving manipulations across multiple temporal and spatial scales.

To better understand impacts of forest management techniques, the Indiana Department of Natural Resources (IDNR)-Division of Forestry (DoF) and Purdue University designed a series of forest manipulations within the context of the Hardwood Ecosystem Experiment (HEE), a 100-year multi-agency, multi-university experiment at Morgan-Monroe (MM) and Yellowwood (YW) State Forests (Kalb and Mycroft, this publication). Provided herein is an overview of the bat community at HEE research sites before timber harvest, and hypotheses on how each species will respond to these manipulations.

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STUDY AREA

MM and YW state forests are in Morgan, Monroe, and Brown Counties in south-central Indiana (Fig. 1). The forests are similar in composition and are managed as one unit by DoF. Both state forests consist mainly of high ridges with steep runoff streams and upland forests. YW has 9,459 ha and MM has >9,720 ha of forest subject to harvest.

MATERIALS AND METHODS

Bats were captured using four multi-tiered mist-net sets (Avinet, Inc., Dryden, NY) placed at two sites in each of the nine management units (see Kalb and Mycroft, this publication). Sites were netted twice per summer (15 May to 15 August) during 2006, 2007, and 2008 on nonconsecutive nights, resulting in 144 net-nights of sampling each year, for a total

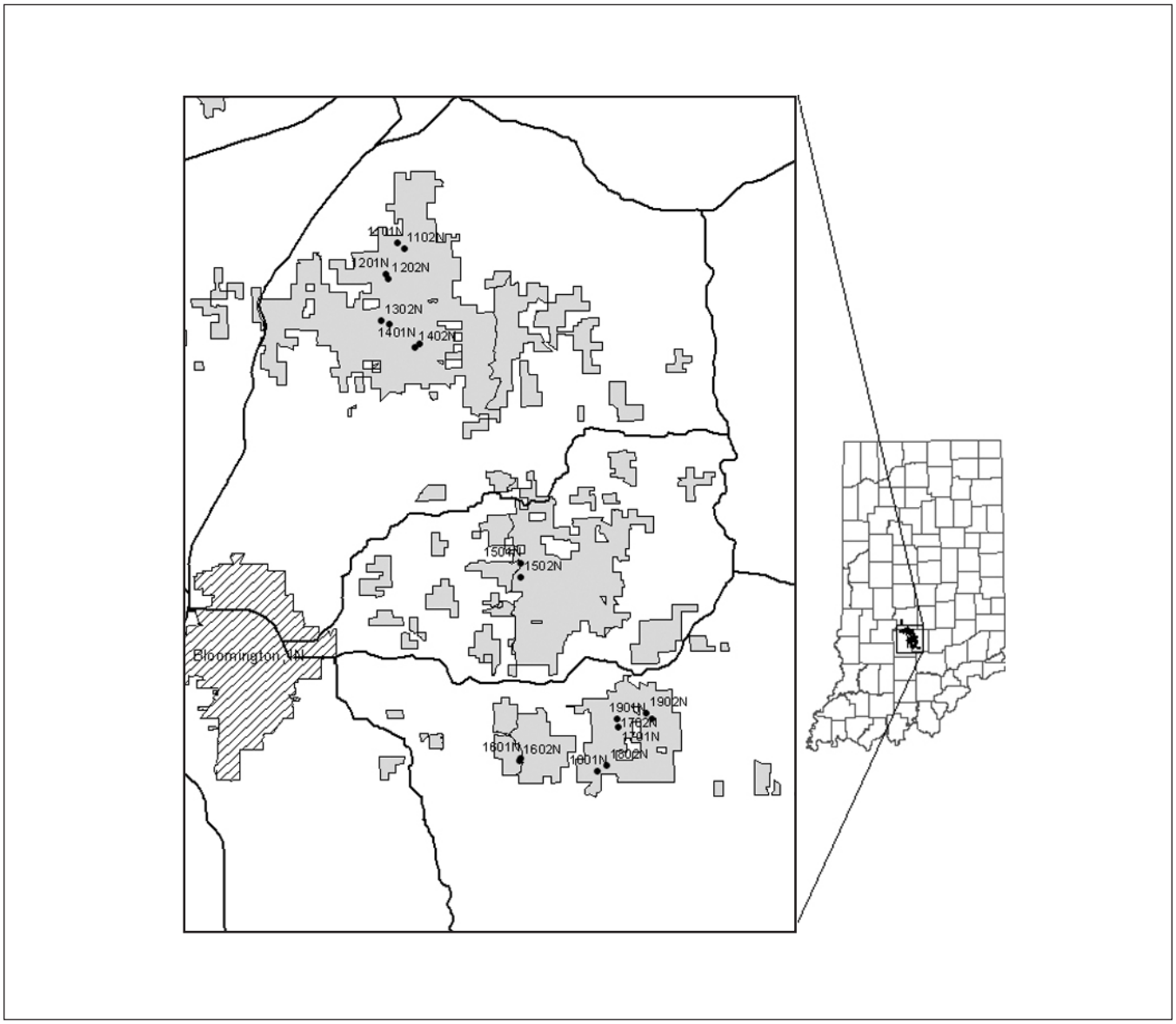


Figure 1.—Location of Morgan-Monroe (MM) and Yellowwood (YW) State Forests (gray areas) in relation to Bloomington, IN (hash mark). Solid circles denote net sites.

of 432 net-nights. At each site mist-net sets were arranged to maximize bat captures with most placed across logging roads. At each site we recorded habitat descriptions, including tree species, potential roost trees, undergrowth species, and distance to water sources. Weather conditions were monitored, including temperature (°C), estimated cloud cover, and wind speed. Mist nets were set by nightfall and left in place for 5 hours. Nets were checked at least every 10 minutes. Data collected on captured bats included species, sex, reproductive condition, right forearm length, body mass, and an estimate of age (juvenile or adult). Numbered metal bands (Prozana Ltd., Icklesham, East Sussex, UK) were fitted to the right (males) or left (females) forearm to allow identification of individual bats. Statistical analyses were conducted with SPSS 11.0 for Windows (SPSS Inc., Chicago, IL) with a rejection level of $\alpha = 0.05$ used throughout. A Kruskal-Wallis one-way analysis of variance test for non-parametric data was used to compare the capture rates of each species among years and MacArthur's (1972) diversity index across all years.

During 2006-08 24 net-nights did not meet pre-specified weather conditions of rain or temperature (because of rain for >45 minutes or ambient temperature <10 °C). Bats captured on those nights were not included in the analysis of standardized data, and the site was netted again. This data set was included to provide a more complete survey.

RESULTS

Over 3 summers 342 bats were captured consisting of 8 species (Table 1). In 2006, we captured 140 bats representing 6 species: northern myotis (*Myotis septentrionalis*), eastern red bat (*Lasiurus borealis*), big brown bat (*Eptesicus fuscus*), little brown myotis (*M. lucifugus*), Indiana myotis (*M. sodalis*), and tri-colored bat (*Perimyotis subflavus*)[2]. In 2007, 87 bats were captured, including an additional species, the hoary bat (*L. cinereus*). In 2008, 115 bats were captured, including the migratory silver-haired bat (*Lasionycteris noctivagans*).

A total of 432 net-nights resulted in 0.79 bats per net-night. Capture rates (bats per net-night) for each species were similar across the 3 years ($p = 0.091$). The total capture rate of bats was 0.83 per net-night (192 net-nights) in MM State Forest and 0.77 bats per net-night (240 net-nights) in YW State Forest. The MacArthur diversity index for MM and YW forests combined for each year was 3.5 in 2006, 3.0 in 2007, and 4.0 in 2008. The supplemental netting in 2007 captured 5 bats and in 2008 captured 11 bats (Table 2).

[2] Formerly eastern pipistrelle (*Pipistrellus subflavus*)

Table 1.—Numbers (n) of bats captured in mist nets in 2006, 2007, and 2008 (each with 144 net-nights, NN), within Morgan-Monroe and Yellowwood State Forests in Indiana, and results of the Kruskal-Wallis test comparing capture rates (n/NN) of species among years, 2006-08.

	2006		2007		2008		Total		Kruskal-Wallis	
Species	n	n/NN	n	n/NN	n	n/NN	N	n/NN	χ^2	P
Northern myotis	53	0.37	37	0.26	41	0.28	131	0.30	0.85	0.65
Eastern red bat	46	0.32	26	0.18	32	0.22	104	0.24	5.29	0.07
Big brown bat	27	0.19	12	0.08	21	0.15	60	0.14	3.59	0.17
Indiana myotis	5	0.03	3	0.02	8	0.06	16	0.04	0.30	0.86
Tri-colored bat	4	0.03	3	0.02	10	0.07	17	0.04	0.89	0.64
Little brown myotis	5	0.03	4	0.03	2	0.01	11	0.03	1.12	0.57
Hoary bat	0	0.00	2	0.01	0	0.00	2	0.00	4.08	0.13
Silver-haired bat	0	0.00	0	0.00	1	0.01	1	0.00	2.00	0.37
Total	140	0.97	87	0.60	115	0.80	342	0.79	4.78	0.09

DISCUSSION

The eight species of bats currently found in the HEE study area are typical of the region's forested areas (Mumford and Whitaker 1982, Whitaker and Hamilton 1998). The MacArthur diversity index increased each year because a new species was caught each year. A ninth species, the evening bat (*Nycticeius humeralis*), also may inhabit these forests, but is most often caught in lowland areas (Whitaker et al. 2007), which were not well represented in the sample.

Bat species capture rates reflect the structure of the HEE study sites, which historically have been managed using mostly uneven-aged harvest methods. Northern myotis, a clutter-adapted species (Loeb and O'Keefe 2006), was most commonly caught in both state forests. "Clutter" refers to physical objects in the environment into which a bat can fly or that may disrupt echolocation. Clutter-adapted species have high-frequency echolocation calls that produce more details about an environment and allow a bat to fly through areas of dense foliage, and body/wind shapes that allow for rapid maneuverability (Brooks and Ford 2006). Single-tree and group-selection management results in a mosaic of wooded areas and small openings around which northern myotis typically forage (Patriquin and Barclay 2003).

The tri-colored bat is an abundant species in other forests in southern Indiana (Brack et al. 2004, Brack and Whitaker 2004) but was rare in the HEE study area. Tri-colored bats are clutter-adapted but are known to forage in many types of forest habitats. The lack of diverse habitats in the HEE study area may explain why the tri-colored bat was rare. Indiana myotis use clutter-adapted echolocation calls, suggesting that they would be abundant in the HEE study area, but they were caught infrequently. Indiana myotis occur in more open woodland habitats with little or no understory and small openings (Menzel et al. 2001, Sparks et al. 2005a). Indiana myotis also roost in dead trees and require solar exposure for offspring (Kurta 2005), an attribute which might not be abundant under forest canopy. Little brown myotis were also infrequently captured and are less clutter-adapted than other myotis (Arita and Fenton 1997, Broders et al. 2004). Little brown myotis are known to forage over water, forest openings, and forest edges (Brack 2009), which were rare in the HEE study area.

Eastern red and big brown bats were commonly caught within the study area. Neither species is adapted to clutter, but most net sites were in uncluttered corridors where these bats can travel and forage. Hoary bats and silver-haired bats are not adapted to clutter and were not commonly captured. Hoary bats are rarely caught in mist nets, possibly because they are commonly active above the height sampled with mist nets. Silver-haired bats are usually caught in early spring during migration. Eastern red, hoary, silver-haired, and big brown bats are usually found foraging in low-clutter areas (i.e., old field, forest openings, and forest edges), which were rare in the HEE study area.

Our expectation is that all eight species will be present following timber harvest, but that the community composition may change at research sites as each species responds to changes in habitat. Over the short term, individual bats may be killed if they are roosting in a tree when it is harvested (Belwood 1979, Humphrey et al. 1977). To avoid mortality of bats, timber harvest should be conducted in the winter when bats are either hibernating or have migrated. In the long term, succession from dense forests (the current structure of the HEE study area) to more open forest (HEE study area after timber harvest) will benefit most species, especially the Indiana myotis. Below we summarize the foraging and roosting needs of the eight species captured at the study areas during the pre-treatment period, and based on these habitat needs, make predictions about how each species will respond to the various planned timber harvests in both the short term (i.e., immediately after harvest) and long term.

Northern Myotis

Northern myotis typically roost in tree cavities or under bark (Lacki and Schwierjohann 2001) and forage in forest interiors (Brack and Whitaker 2001, Owen et al. 2003, Patriquin and Barclay 2003). Northern myotis prefer interior forest (Owen et al. 2003, Patriquin and Barclay 2003) and so are less likely than other species to use recently clear-cut areas and other forest openings. For example northern myotis were captured less frequently following clearing of a large forest patch along Prairie Creek in Vigo County, IN (Sparks et al. 1998). Other types of timber harvest will not negatively impact this species as long as the availability of preferred roosts (hollow trees) remains unchanged in the surrounding landscape and the harvest protocol calls for the retention of snags. The species may be among the most tolerant of dense subcanopy (Owen et al. 2003), and thus as forest management practices create less cluttered forest, it may face competition with other species for foraging and roosting habitat within the HEE study area. Forest stands that are "solid walls" of vegetation provide little usable habitat for the northern myotis; however, single-tree selection and patch cuts can create a suitable matrix of habitats composed of different tree age classes over time.

Eastern Red Bat

Eastern red bats roost primarily in tree foliage (Mager and Nelson 2001) and forage in open habitats (Hutchinson and Lacki 1999, Walters et al. 2007). In the short term, timber harvests will remove potential roost trees but create open foraging habitat in a cluttered landscape (Elmore et al. 2004). We suspect that the overall availability of roost trees in the surrounding uncut blocks will provide a ready supply of suitable trees for roosting. As such, the greatest impact on eastern red bats may be the creation of uncluttered foraging habitat such as clearcuts, large patch cuts, logging corridors, and understory removal in shelterwoods. In the long term without continued timber harvests or management of past harvest areas, the openings will succeed to thick stands of saplings in which eastern red bats cannot forage.

Big Brown Bat

Big brown bats use anthropogenic structures for roosting during both summer and winter (Whitaker et al. 2007); however, some individuals roost in tree cavities after maternity colonies break up in late summer (Whitaker 1996, Duchamp et al. 2004). Big brown bats forage in early successional and forest openings and forests with little subcanopy (Loeb and O'Keefe 2006) and a wide variety of other non-forested habitats (Duchamp et al. 2004). Openings created by clearcuts, large patch cuts, and shelterwoods will provide increased foraging opportunities for big brown bats in the short term. Smaller patch cuts and single-tree selection may not provide enough space and may be too scattered among cluttered areas for big browns to forage. In the long term, big browns will require more areas where the subcanopy is removed and new forest openings are created.

Tri-colored Bat

Tri-colored bats roost in leaf clusters in trees (Veilleux et al. 2003), tree cavities (Yates and Muzika 2006), and anthropogenic structures (Whitaker 1998). Tri-colored bats forage in forest openings and early successional areas in South Carolina (Loeb and O'Keefe 2006) and in cluttered forests (Menzel et al. 2005), but at Indianapolis International Airport, closer to our project area, this species uses a variety of land classes, such as open fields, the space above saplings in a regeneration opening, and mature closed canopy forest (Helms 2010). As such, tri-colored bats may be a habitat generalist and will forage in habitats in different successional stages. The different timber harvests should benefit the tri-colored bat by creating a mosaic of different habitats through the forest.

Indiana Myotis

Indiana myotis typically roost under tree bark (Humphrey et al. 1977) and forage at the air-vegetation interface (Sparks et al. 2005a), including along logging roads, riparian streams, above and below the canopy, and at the edge of clearings. Foraging Indiana myotis

may make increased use of edge habitat created by harvests (Menzel et al. 2001). Further, regenerating clearcuts will closely resemble seedling plantations at Indianapolis International Airport, where Indiana myotis forage (Sparks et al. 2005a); if dominated by oak (*Quercus*) species (as planned), harvest areas may provide a source of Asiatic oak weevils (*Cyrtepistomus castaneus*), which are both an important consumer of young oaks and acorns and a common food item of several bat species including Indiana myotis (Brack 1983, Brack and LaVal 1985, Tuttle et al. 2006). As the understory removal stage of a three-stage shelterwood cut eliminates the subcanopy, the reduced clutter will allow movement through the forest. However, 30- to 80-year-old timber stands will be too cluttered for Indiana myotis.

A hypothetical comparison of high-quality Indiana myotis summer foraging habitat and successional stages of a forest is shown in Figure 2 (adapted from Sheets 2010). Through time, forest structure naturally increases in clutter during early successional stages and decreases in clutter in later successional stages. Timber harvests can simulate these natural stages. A disturbance event, such as a clearcut, provides uncluttered high-quality foraging habitat for Indiana myotis. Single-tree selection and patch cuts will also provide high-quality habitat because they can mimic a shifting mosaic. A shifting mosaic is a climax stage of forest succession before European settlement. The first cut of a shelterwood will decrease clutter under the canopy, and the overstory cut of the shelterwood will shift the successional stage to a clearcut-like condition that has seedlings already established (Sheets 2010).

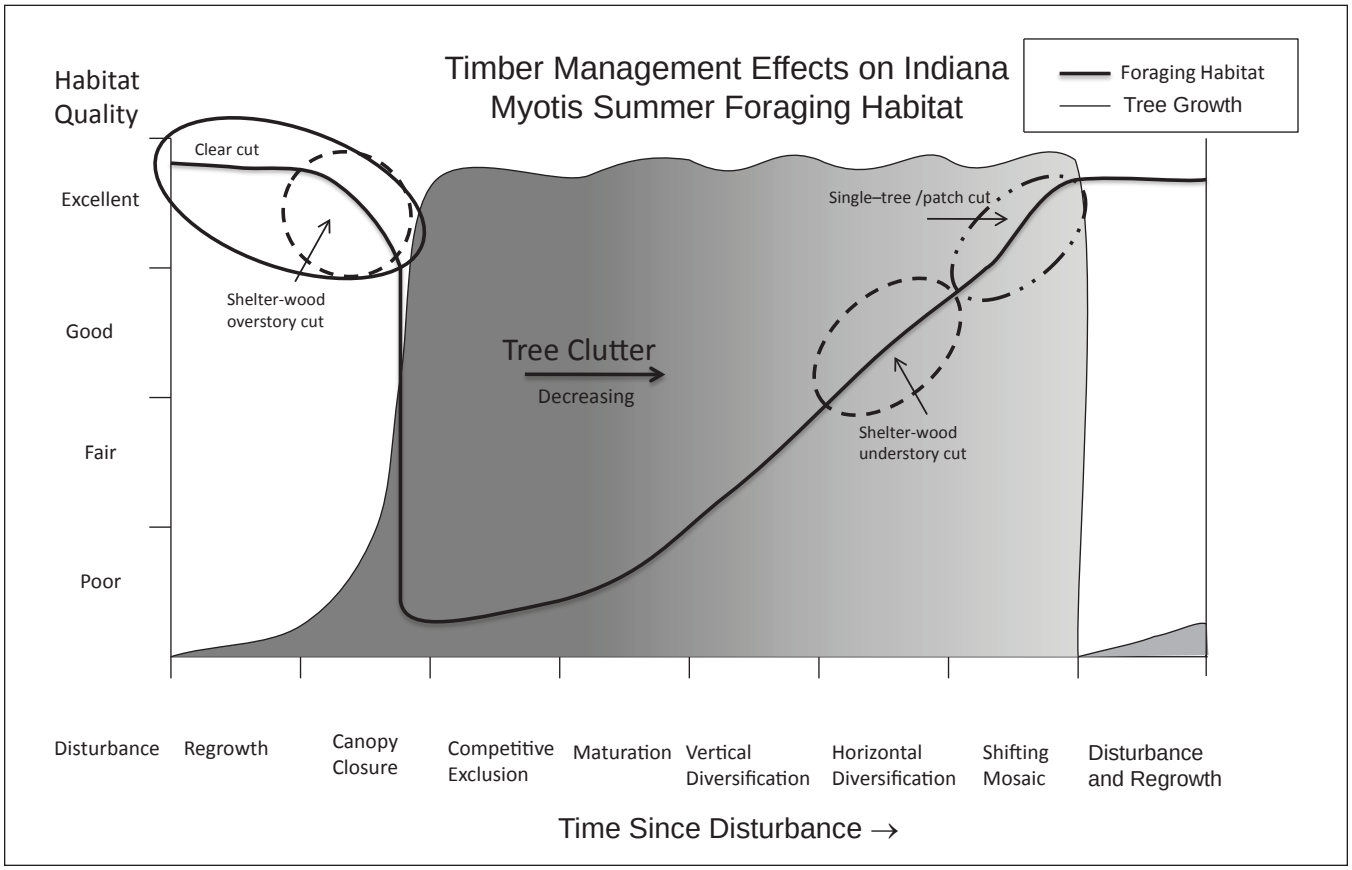


Figure 2.—Predicted changes in quality of summer foraging habitat for Indiana myotis as a function of time since disturbance and type of silvicultural treatment used. The horizontal axis depicts changes in forest structure over time. Ellipses represent harvest activity. The thin line bounding the upper portion of the shaded area mimics change in tree size following disturbance, whereas the thick line represents change in quality of foraging habitat associated with stand structure. The shaded area within the solid line denoting tree size depicts the decrease of clutter in a forest through time. Adapted from Sheets (2010).

It is possible that some Indiana myotis roost trees will be lost to harvest. Given the relatively small size of regeneration openings (≤ 4 ha), however, the availability of suitable roost trees in unharvested control units and buffer areas and retention/creation of snags in harvest areas (IDNR 2008), should ensure that much suitable roosting habitat will remain. Outside harvest areas, falling trees and logging equipment will damage some trees, which may help create suitable roosts for this species over the long term (Gumbert et al. 2002). Harvest methods that successfully yield mature oak-hickory (*Carya*) stands may benefit this species as it frequently roosts in shagbark (*C. ovata*) and shellbark (*C. lacinosa*) hickories and oaks (Callahan et al. 1997, Kurta 2005, Whitaker and Sparks 2008) during summer. Over the long term, the senescence and death of oaks and hickories during succession may provide valuable roost sites.

A hypothetical comparison of successional stages of a forest and high-quality Indiana myotis summer roosting habitat is shown in Figure 3 (adapted from Sheets 2010). Each type of timber cut, without additional girdling of trees, can provide a quality of roosting habitat different from the others. Clearcuts have the longest period of low-quality roosts because it takes time for the forest to produce large high-quality snags to serve as Indiana myotis roosts. Shelterwood cuts essentially resemble a clearcut harvest, but the effect is delayed because the overstory is cut later. Single-tree selection and patch cuts are likely to retain and have higher quality snags than the other timber harvests because trees will be damaged by the smaller cuts, thereby creating more snags; many snags may also be left standing (Sheets 2010).

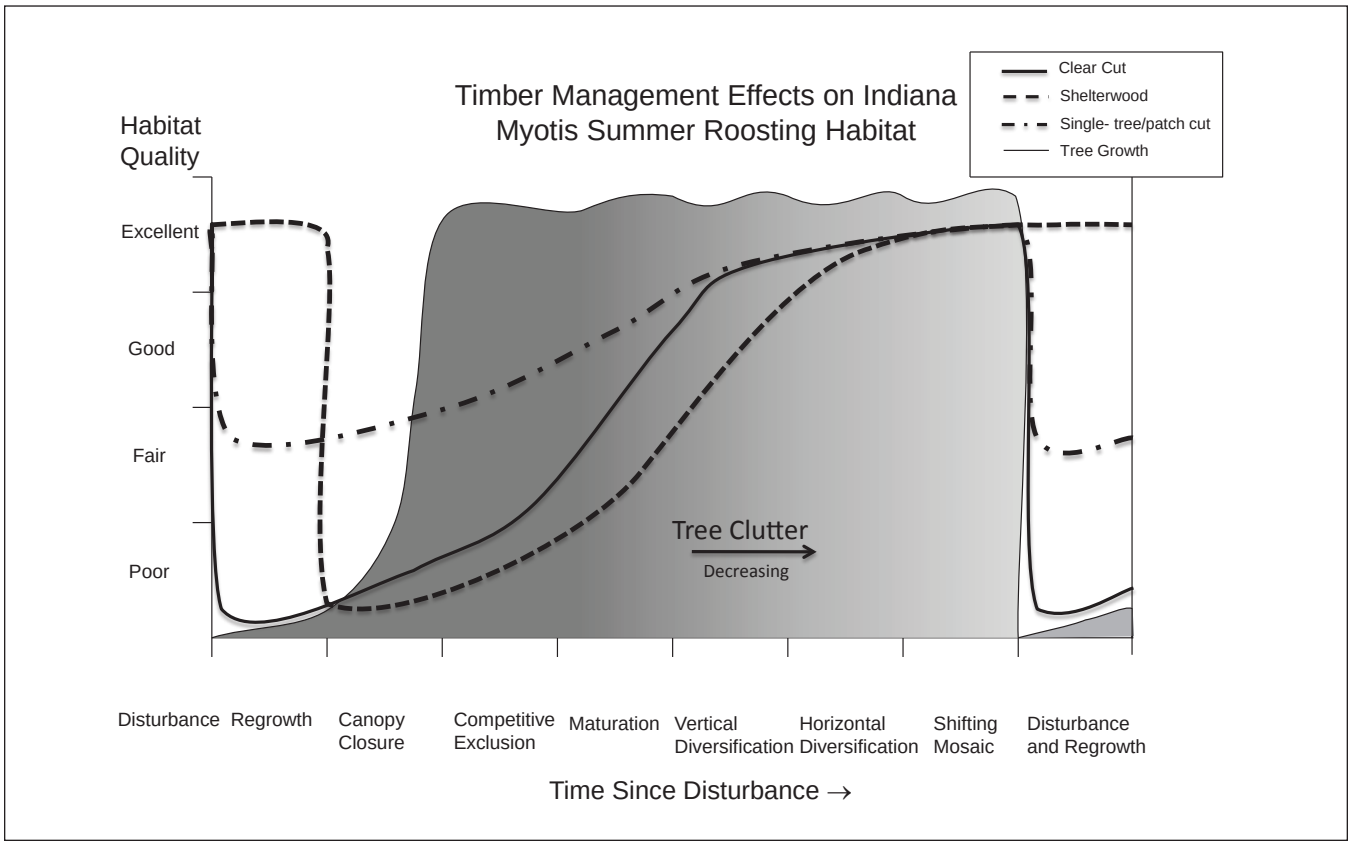


Figure 3.—Predicted changes in quality of summer roosting habitat for Indiana myotis as a function of time since disturbance and type of silvicultural treatment used. The horizontal axis depicts changes in forest structure over time. The thin line bounding the upper portion of the shaded area mimics change in tree size following disturbance and the shaded area within the solid line denoting tree size depicts the decrease of clutter in a forest through time. Adapted from Sheets (2010).

Little Brown Myotis

In Indiana, most documented summer roosts of little brown myotis have been in anthropogenic structures, especially attics of buildings and expansion cracks of bridges, although they occasionally roost under the exfoliating bark of dead trees (Whitaker et al. 2007). Little brown myotis forage along forest edges (Patriquin and Barclay 2003), in centers of clearcuts (Hogburg et al. 2002), and over aquatic habitats (Anthony and Kunz 1977). Being less clutter-adapted than other myotis, these bats are likely to benefit from larger clearcuts and shelterwoods, which have less clutter, but unlikely to benefit from smaller single-tree or patch cuts, which have more clutter in between openings. Corridors connecting water bodies or clearings around existing water sources will benefit little brown myotis. As harvest areas succeed, they are likely to become too cluttered for this species to use extensively.

Hoary Bat

Hoary bats roost mainly in tree canopies (Sparks et al. 2005b, Perry and Thill 2007) and based on wing size and call frequency, forage in open areas (Barclay et al. 1999, Elmore et al. 2004). Hoary bats will benefit from uncluttered foraging habitat in clearcut, large patch cut, and shelterwood harvest areas. Currently hoary bats are rarely encountered during mist-net surveys (we captured two during this study), but we suspect that the species roosts in these forests and forages outside the sampled area in surrounding unforested areas and above the canopy (Sparks et al. 2005b). Roosting habitat will remain abundant on the landscape as long as dominant mature trees are present (Perry and Thill 2007).

Silver-haired Bat

Silver-haired bats summer in the upper Midwest and provinces of Canada, where they roost in tree hollows (Parsons et al. 1986, Whitaker and Hamilton 1998) and forage in the interior of clearcuts (Hogburg et al. 2002). Silver-haired bats migrate through central Indiana during spring and autumn (Mumford and Whitaker 1982), although some hibernate in southern Indiana (Whitaker et al. 2007). Where bats, including the silver-haired bat, forage and roost during migration is not well known. As such, it is speculation that the silver-haired bats will forage in the HEE study area after harvest during migration. Silver-haired bats also feed extensively on caddisflies (Order Trichoptera), which live in streams and wetlands (Whitaker 1972), and may not typically forage in the upland areas of the HEE study area.

CONCLUSIONS

As the HEE progresses, the data presented herein will provide a valuable reference point to determine how the bat community responds to forest management techniques. With new openings created by timber harvest every 20 years, a mosaic of even-aged stands in varying successional states, uneven-aged stands, and unharvested control units will provide foraging habitats for species of bats throughout this 100-year project. Eventually, comparison of the bat community within these habitat types before and after timber harvests should provide substantial insight into ways to successfully manage bats and timber production on the same landscape.

ACKNOWLEDGMENTS

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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.

HABITAT USE BY BATS IN TWO INDIANA FORESTS PRIOR TO SILVICULTURAL TREATMENTS FOR OAK REGENERATION

Jeremy J. Sheets, Joseph E. Duchamp, Megan K. Caylor, Laura D'Acunto, John O. Whitaker, Jr., Virgil Brack, Jr., and Dale W. Sparks[1]

Abstract.—As part of a study examining the effects of silvicultural treatments for oak regeneration on habitat use by bats, we surveyed forest stands prior to the implementation of treatments in two state forests in Indiana. Interior forest sites corresponding to areas designated for silvicultural treatments were surveyed for 2 nights each during the summers of 2007 and 2008. Additionally, three types of existing forest openings (pre-existing harvest openings, forest edges, and open corridors) were surveyed during 2007 in the same area. We assessed bat habitat use by recording echolocation calls using ANABAT II bat detectors. We characterized a location as experiencing high levels of activity, experiencing low levels of activity, or being unused by a species of bat based on recorded call minutes. We used occupancy models to account for differences in detection probability.

Stands designated for uneven-aged harvest had a greater probability of experiencing high activity by a species of bat when compared to areas designated for other silvicultural treatments. Existing forest openings had probabilities of high activity levels that were similar to each other, but high activity levels in these habitats were less likely for Indiana myotis (*Myotis sodalis*) and northern myotis (*M. septentrionalis*) when compared to forest interior locations. At interior forest sites, northern myotis were more likely than other species to have high activity levels. These results provide some insight into the potential responses of bats to forest management practices and serve as a baseline for future experimental treatments.

INTRODUCTION

Bats are one of the most diverse mammals in both feeding habits and numbers of species, making up 20 percent of extant mammalian species (Nowak 1994, Altringham 1996). Being the only volant mammals, they provide important ecosystem services, such as regulating insect populations and distributing forest nutrients (Duchamp et al. 2010, Kunz and Fenton 2005). Because bats are widespread and can consume >50 percent of their body weight in insects during a night, they play an important role in reducing insect populations (Cleveland et al. 2006). Several studies have confirmed that bats can reduce the amount of insects in a landscape (Kalka et al. 2008, Williams-Guillén et al. 2008). The estimated annual value of bats to North American agriculture is $3.7 billion (Boyles et al. 2011).

Bats can also be important biological indicators. Studies have shown that bats respond rapidly to

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changes in the environment (Jones et al. 2009, Kunz et al. 2007, Lacki et al. 2007). Their ability to fly lets them move from less desirable habitats to more suitable ones. Thus, a presence, absence, or change in bat diversity and activity can help determine the environmental effects of human activities on ecological communities in a landscape. For example, one study showed that bat activity was reduced in areas of poor water quality as a result of sewage outfalls (Vaughan et al. 1996).

Bats also can accumulate heavy metals, pesticides, and other pollutants found in the environment. Researchers in Britain have found that bat activity and diversity increased on organic farms versus traditional farms that used pesticides (Wickramasinghe et al. 2003). Additionally, heavy metals such as mercury and lead have been found in high concentrations in bats, probably due to bioaccumulation from feeding on insects in polluted areas (O'Shea et al. 2001). Bats' ability to respond to landscape changes and pollution makes them valuable indicators of ecosystem health.

The suitability of forests for bats can be described by four factors: the density of forest structure or "clutter" and the availability of roosts, prey, and water (Hayes and Loeb 2007). All of these characteristics can be drastically affected by timber harvests, and previous studies have shown a strong relationship between timber harvests and forest use by resident bat species (Hayes and Loeb 2007, Patriquin and Barclay 2003). Bats' ability to maneuver through dense forest environments, and the efficiency with which they can use large forest openings, can be predicted to some extent by their wing morphology and body size. Larger-bodied bats with long, narrow wings tend to fly faster and more efficiently and make use of forest openings (Norberg and Rayner 1987). In contrast, smaller bats with a wing-tip shape that aids in their maneuverability can carefully pick their way through a cluttered forest environment (Norberg and Rayner 1987).

Bats' response to timber harvest is typically measured by use, either flight activity represented by recorded echolocation calls or selection of roosting habitat (Hayes and Loeb 2007). Although characteristics of a forest can affect the ability to record echolocation calls, with some exceptions (Duchamp et al. 2006, Yates and Muzika 2006) previous studies of bat activity have not typically estimated detection probability when measuring use. Additionally, to our knowledge, response to timber harvest has been measured only after a timber harvest. A true experimental manipulation involving the measurement of pre-harvest activity levels followed by long-term post-harvest monitoring of activity has not occurred.

The present study measures bats' pre-harvest use of intact forest stands based on echolocation calls and analyses that incorporate detection probability. Additionally, we monitored activity levels in a variety of existing openings for comparison with future activity levels in openings created by silvicultural treatments. Our study had two objectives: 1) compare habitat use by bats among existing forest openings prior to silvicultural treatment, and 2) compare habitat use by bats among our proposed treatment areas.

STUDY AREA

This study was conducted on Morgan-Monroe State Forest and Yellowwood State Forest in Morgan, Monroe, and Brown Counties in south-central Indiana (Fig. 1). Both forests (19,000 ha combined) were established in the 1920s after the high ridges, steep slopes, and narrow streams proved unsuitable for agriculture (Carman, this publication; Sheldon 2007). Today, these sites are covered in upland forests dominated by oaks (*Quercus* spp.) and hickories (*Carya* spp.), broken by occasional corridors (mostly logging roads) and previously harvested areas. Previous timber management consisted of single-tree and group selection in both forests (Sheldon 2007).

During the next 100 years, nine management units totaling 3,603 ha within these forests will be subjected to a variety of silvicultural treatments: even-aged (clearcuts and shelterwood cuts) and uneven-aged (single-tree selection and patch cuts) harvest, and control areas of no harvest (Kalb and Mycroft, this publication).

MATERIALS AND METHODS

Sampling

During 2007 and 2008, bats were acoustically sampled at locations relative to planned timber harvests within the nine experimental management units (Kalb and Mycroft, this publication). Within even-aged and control management units, the four areas designated

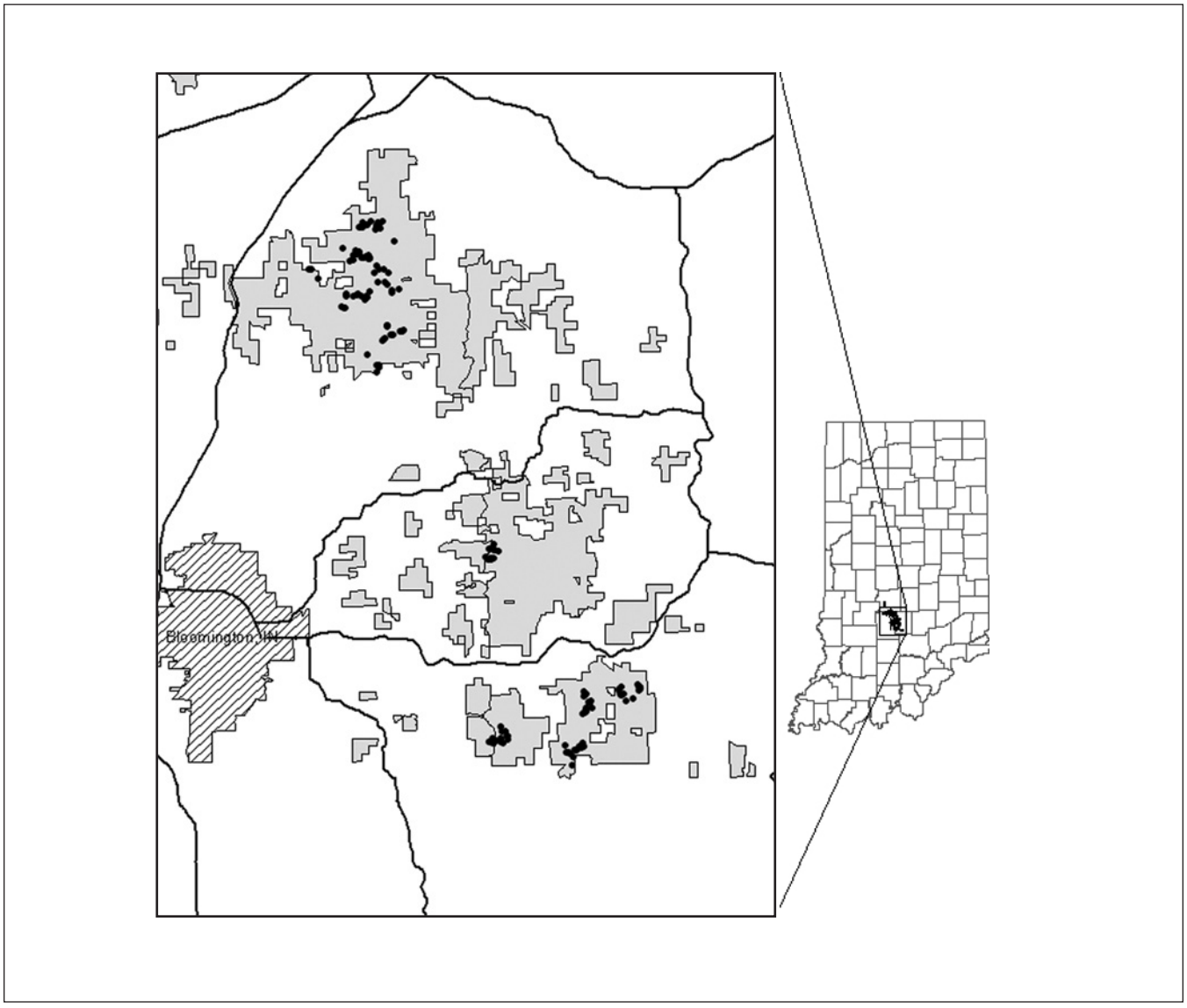


Figure 1.—Location of Morgan-Monroe (MM) and Yellowwood (YW) State Forests (gray areas) in relation to Bloomington, IN (hash mark). Solid circles denote sample sites.

for silvicultural treatments were sampled. Within the uneven-aged management units, four areas were randomly selected from the eight total areas designated for silvicultural treatments. For each sampled treatment area, detectors were placed at least 40 m apart at three locations relative to expected harvest area: inside, adjacent to, or outside the timber harvest. For control sites where no harvest was to occur, locations were still based on the area that potentially could have been harvested. Each detector location was considered to be an independent sample.

During 2007, in addition to the forest interior sites described above, we sampled three forest habitats that represented existing structural heterogeneity prior to timber harvests: forest edge, pre-existing harvest openings, and open corridors. Forest edge habitat was the interface of the forest with open lands, largely agricultural fields. Pre-existing harvest openings were non-linear openings (<2 ha) in the forest canopy. Corridors were linear openings within the forest such as fire and hiking trails, logging and access roads, and utility corridors. Locations for forest edge, pre-existing harvest openings, and open corridor samples were selected by identifying multiple, suitable sites for each habitat throughout the study area and then randomly selecting among these using a random numbers table. Additional corridor sites were chosen at established sites where bats were captured with mist nets along existing logging trails. Sampling occurred at 108 forest interior locations, 13 forest edges, 12 pre-existing harvest openings, and 24 corridors.

At each sample location, echolocation calls were digitally recorded to a CF card during 2 entire nights per year using an ANABAT II detector with a CF ZCAIM (Titley Electronics, Ballina, New South Wales, Australia). One sample occasion occurred during the early summer (15 May-8 July) and the other later in the summer (9 July-15 August). These time periods correspond to typical formation and then dispersion of maternity colonies in the area. Each ANABAT detector was set about 1 m from the ground on a polyvinyl chloride (PVC) pipe and housed within a plastic storage container with a 45° PVC elbow at one end (Duchamp et al. 2006). Microphones were placed 1 cm from the PVC elbow. Detectors were aligned on a random azimuth selected from a random numbers table, and areas of dense vegetation were avoided.

Bat Echolocation Call Identification

To identify recorded bat call sequences, we cleaned digital recordings with a filter in program ANALOOK (v. 4.8; Corben 2001) according to settings proposed by Britzke and Murray (2000). Call sequences were screened visually for irregular calls, and only regular search-phase calls were retained and measured in ANALOOK. We identified species by comparing measured parameters of a recorded call to the same parameters of calls in a reference library of echolocation calls collected from bat communities across the eastern United States (Britzke et al. 2011). We reduced the species included in the reference library to summer residents regularly captured during extensive mist-net surveys occurring within the study area: big brown bat (*Eptesicus fuscus*), eastern red bat (*Lasiurus borealis*), hoary bat (*L. cinereus*), little brown myotis (*Myotis lucifugus*), northern myotis (*M. septentrionalis*), Indiana myotis (*M. sodalis*), and tri-colored bat (*Perimyotis subflavus*)[2].

We then used the remaining call reference library to train 20 feed-forward, back-propagation neural networks, each with a single hidden layer of 24 nodes and skip layer connections (R v. 2.1; package ‘‘nnet’’) (Britzke et al. 2011, R Development Core Team 2011, Venables and Ripley 2002). Each network assigned a species identity to each call. We then assigned a final identity to each call by taking the mode of species assignments from the 20 neural networks. Similarly, we determined the identification of a call sequence by taking the mode of the species assignments for

[2] Formerly eastern pipistrelle (*Pipistrellus subflavus*)

calls within that sequence. Sequences were deemed unidentified if the mode of identified calls either was less than four calls or less than 60 percent of all calls in the sequence.

To quantify bat activity levels for each species, we counted the number of minutes within which a species of bat was recorded and identified during a night of sampling. We categorized the amount of activity during a night into three categories: no activity, low-level activity, and high-level activity. Failure to detect a call during a night resulted in categorization as no activity for that location x species combination. If the number of minutes was in the upper quartile of all levels of activity for a species, then that location was considered to have a high level of activity during a night. If the number of minutes was in the lower three quartiles for a species, the location was considered to have a low level of activity during the night in question. The level of activity at a site was summarized into a detection history for each location across all sample nights. The activity level for each species of bat was considered independent.

Statistical Models

Due to differences in the years sampled, we analyzed bat activity data collected during 2007 at existing forest openings separately from data collected during both 2007 and 2008 at interior forest locations prior to silviculture treatments. Diversity and activity levels recorded during 2007 for the three types of existing forest openings and interior forest locations were compared using an analysis of variance (ANOVA) and Tukey's method of post-hoc comparisons (R Development Core Team 2011, Yandell 1997). The sample unit was considered to be the sample site with activity pooled across both sample nights. Minutes of activity were log transformed to improve assumptions of normality. Diversity was calculated using the exponential of the Shannon-Wiener index (Jost 2006) and was considered only for sample locations with >10 minutes of recorded activity. We also used these methods to compare overall bat activity and diversity between areas designated for silvicultural treatments.

For these data the forest stand was considered the sample unit and minutes of activity were pooled across the three sample sites within each stand, and across all sample nights for both years of the study (2007 and 2008).

We relied on occupancy models to make species-specific comparisons of bat activity that would account for potential differences in the probability of detection among species and between habitats. Because bats can easily travel among our sampling locations, our occupancy models provide a relative probability of use rather than "occupancy" for our sample locations (MacKenzie 2006). We use the term "activity" to describe this use to remain consistent with existing bat literature. For data collected at existing forest openings during 2007, we used a single-state occupancy model (MacKenzie et al. 2006) to estimate two parameters: the probability of detecting high activity levels (p) and the probability of high activity levels occurring at a site for a species of bat (Ψ). The models resulting from this process were based on two detection nights at each sample site during a single year.

For data collected during 2007 and 2008 at interior forest sites, we used a multi-state occupancy model to estimate the probability of either low activity levels or high activity levels for a species of bat at a sample site (Nichols et al. 2007). The models resulting from this process were based on 4 detection nights at each sample site: 2 nights of sampling each year. The multistate occupancy models estimated five types of parameters. Two probabilities of detection were estimated: the probability of detecting low activity levels (p_1) and the probability of detecting high activity levels (p_2). Additionally, two probabilities of occurrence were estimated: the probability of low activity for a species occurring at a site (ψ_1) and the probability of high activity for a species occurring at a site (ψ_2). The fifth estimated parameter was for the probability that a site with high activity levels would appear to have low activity levels for a species during a sample night (δ).

The highest level of activity detected for a species at a site was considered to characterize activity at that site. Inconsistencies in the detections of the highest level activity were attributed to imperfect detection of activity levels at a site. Statistical models were run in program MARK (White 2011) using an extension package RMark (Laake and Rexstad 2011) within program R (R Development Core Team 2011).

With both types of models, we used a similar approach to building models. When modeling the parameters detection probability (p) and probability of activity levels occurring at a site (ψ), we tested a null model estimating a single value for each parameter against models where the values of parameters would be estimated separately for particular factors. The types of factors fell into two general categories: location differences and species differences. For the models examining forest openings, location factors were based on the type of forest opening sampled: forest edge, pre-existing harvest openings, and open corridors.

For the models examining interior forest plots over 2 years, the location factors were based on the planned experimental silvicultural treatments: even-aged, uneven-aged, and non-harvest treatments. Species factors were the individual species included in the models. Models allowing for factor effects were built in a forward step-wise fashion, beginning with single factor effects and then testing additional additive effects. We first built models describing detection probability (p), followed by models describing probability of activity level occurring at a site (ψ). Model comparisons were based on AICc values (Burnham and Anderson 2002).

RESULTS

Summary of Recorded Echolocation Calls

All seven species known to be summer residents in the area were detected via echolocation call recordings (Table 1). We recorded bat echolocation calls during

Table 1.—Summary of (a) number of minutes with a call recorded from each bat species for each treatment plot during 2007-08 and (b) additional sampled habitats in 2007.

(a)	2007			2008			
	Uneven	Even	Control	Uneven	Even	Control	Total
Big brown bat	11	6	2	2	3	5	29
Eastern red bat	8	5	1	11	7	6	38
Hoary bat	0	0	0	0	0	0	0
Little brown myotis	0	7	1	21	7	6	42
Northern myotis	89	35	31	136	89	58	438
Indiana myotis	62	24	7	122	34	47	296
Tri-colored bat	32	25	28	69	35	72	261
Total	202	102	70	361	175	194	1,104

(b)	Corridors	Edges	Gaps	Total
Big brown bat	106	268	203	577
Eastern red bat	158	176	451	785
Hoary bat	6	8	0	14
Little brown myotis	19	66	17	102
Northern myotis	181	22	23	226
Indiana myotis	144	34	20	198
Tri-colored bat	239	113	401	753
Total	853	687	1,115	2,655

3,759 minutes on 516 nights across 157 locations. During 2007, detected activity levels differed among habitat types ($F_{3,153}$ = 48.8 and P < 0.001). Detected activity was lowest at interior forest locations (mean = 2.38±5.01 SD) compared to pre-existing harvest openings (mean = 45.71±30.56 SD; P < 0.001), forest edge (mean = 31.88±42.37SD; P < 0.001), and corridors (mean = 19.13±24.96 SD; P < 0.001). Pre-existing harvest openings also had greater activity than corridors (P < 0.001), but were not statistically different from edge locations (P = 0.127; Table 1). The species diversity of bat activity at locations with more than 10 minutes of activity was similar across all four forest habitats ($F_{3,43}$ = 0.609; P = 0.613): forest edges (mean = 3.4±0.9 SD), forest corridors (mean = 3.1±0.9 SD), interior (mean = 3.1±1.2 SD), and pre-existing harvest openings (mean = 2.8±1.0 SD).

Among interior forest locations in both 2007 and 2008, minutes of bat activity among stands designated for the three silvicultural treatment types were similar ($F_{2,33}$ = 1.629 and P = 0.211): stands designated for uneven-aged harvest (mean = 46.9±61.19 SD), stands designated for no harvest (mean = 22.0±30.1 SD), and stands designated for even-aged harvest (mean = 23.1±27.8 SD) (Table 1). Diversity at treatment areas with greater than 10 minutes of activity appeared to be higher in stands designated for even-aged management (mean = 3.8±0.9 SD) relative to stands designated for no harvest (mean = 2.7±0.2 SD) ($F_{2,33}$ = 3.357; P = 0.058). Diversity of bats in stands designated for uneven-aged harvest (mean = 3.1±1.0 SD) was similar to the other two treatments.

Bat Activity Levels in Treatment Plots

We used a multi-state occupancy model to describe the low and high activity levels of bats at the interior forest plots prior to silvicultural treatments. Only species with activity recorded on more than 20 nights were included in statistical models: eastern red bats (22 nights), northern myotis (95 nights), Indiana myotis (56 nights), and tri-colored bats (127 nights). The best of 73 models considered for describing detection probabilities for both low activity (p_1) and high activity (p_2) included terms for a lower probability of detecting activity for eastern red bats for both measures, a higher probability of detecting low activity levels (p_1) of tri-colored bats, and higher probability of detecting bat activity during the second year of sampling (2008) for both measures (Tables 2, 3, and 4).

Table 2.—Rankings for occupancy models describing high activity levels by bats at interior forest locations of treatment stands before implementation of silvicultural treatments. All models estimated detection probabilities as described in Table 3. Twenty-one models were tested, and those within 2 AICc of the highest ranking model were considered competing models. All models within 5 AICc of the highest ranking model along with those that were nested within competing models are shown.

Models for high activity levels ψ_2	Parameters	AICc	Δ AICc	Weight
Northern myotis + uneven treatment stands	12	1679.993	0.000	0.735
Northern myotis + harvest treatment stands	12	1682.897	2.904	0.172
Northern myotis	11	1684.632	4.639	0.072
Uneven treatment stands	11	1687.549	7.556	0.017
Constant	10	1690.823	10.830	0.003

Table 3.—Regression coefficients and uncertainty estimates for highest ranking occupancy model describing high activity levels by bats at interior forest locations of treatment stands before implementation of silvicultural treatments.

Model parameters	Regression model coefficients	Standard error	Lower 95% CI	Upper 95% CI
ψ_1 = *Pr* (low activity)				
Intercept	2.801	1.536	-0.209	5.813
ψ_2 = *Pr* (high activity)				
Intercept	-1.660	0.460	-2.563	-0.758
Northern myotis	1.143	0.386	0.387	1.900
Uneven treatment stands	0.934	0.375	0.198	1.671
δ = *Pr* (detecting low activity at high-activity site)				
Intercept	-0.967	0.239	-1.436	-0.498
p_1 = *Pr* (detection of low activity)				
Intercept	-1.979	0.280	-2.527	-1.430
Tri-colored bat	1.627	0.321	0.998	2.255
Eastern red bat	-1.368	0.659	-2.661	-0.076
Year 2008	-1.401	0.405	-2.195	-0.608
p_2 = *Pr* (detection of high activity)				
Intercept	-0.800	0.216	-1.225	-0.376
Eastern red bat	-1.273	0.425	-2.106	-0.440
Year 2008	0.814	0.269	0.286	1.342

Table 4.—Occupancy model parameter and uncertainty estimates based on highest-ranking model describing high-level use by bats at interior forest locations of treatment stands before implementation of silvicultural treatments.

Model parameters	Parameter estimates	Standard error	Lower 95% CI	Upper 95% CI
ψ_1 = *Pr* (low activity)				
Constant	0.943	0.083	0.802	1.000
ψ_2 = *Pr* (high activity)				
Even and control experimental plots	0.400	0.047	0.313	0.494
Uneven experimental plots	0.870	0.077	0.639	0.962
Even and control experimental plots by northern myotis	0.761	0.104	0.509	0.907
Uneven experimental plots by northern myotis	0.774	0.120	0.472	0.929
δ = *Pr* (detecting low activity at high-activity site)				
Constant	0.362	0.064	0.248	0.494
p_1 = *Pr* (detection of low activity)				
Tri-colored bat in 2007	0.413	0.070	0.306	0.567
Eastern red bats in 2007	0.034	0.021	0.115	0.178
Other bats in 2007	0.121	0.030	0.069	0.189
Tri-colored bats in 2008	0.148	0.039	0.075	0.229
Eastern red bats in 2008	0.009	0.007	0.003	0.032
Other bats in 2008	0.033	0.015	0.011	0.070
p_2 = *Pr* (detection of high activity)				
Eastern red bats in 2007	0.111	0.042	0.040	0.224
Other bats in 2007	0.311	0.046	0.229	0.411
Eastern red bats in 2008	0.221	0.074	0.085	0.400
Other bats in 2008	0.503	0.063	0.386	0.628

The probability that low activity levels would be detected at a location with high activity levels (δ) was left constant. We then tested a series of 21 models that estimated the probability of high and low bat activity levels at sample locations. The best model estimated a single probability of low-level activity (ψ_1) across our sites, and the probability of high activity levels (ψ_2) occurring at a sample location was greater for northern myotis and for all bats in areas designated for uneven-aged silvicultural treatments (Tables 2, 3, and 4).

Bat Use of Existing Forest Openings

We used a simpler single-state occupancy model to describe the probability of high bat activity levels within existing forest openings during the first year of sampling (2007). We included only species with at least 10 nights of high activity: big brown bats (24 nights), eastern red bats (34 nights), northern myotis (12 nights), Indiana myotis (10 nights), and tri-colored bats (25 nights). The best of 15 models considered for describing probability of detecting high activity levels included terms for a greater probability of detection (p) for eastern red bats (Tables 5, 6, and 7). We then tested a series of 34 models that estimated the probability of high activity levels (ψ) occurring in open habitats. Our model comparisons resulted in two competing models that described a lower probability of high activity levels within existing forest openings by Indiana myotis and northern myotis relative to other species (Tables 5, 6, and 7). One of the competing models also described an even lower probability of high activity levels for northern myotis along forest edges (Tables 5, 6, and 7).

Table 5.—Rankings for models describing high activity levels by bats within existing forest openings. All models estimated detection probabilities as described in Table 6. Fifteen models were tested and those within 2 AICc of the highest ranking model were considered competing models. All models within 5 AICc of the highest ranking model along with those that were nested within competing models are shown.

Models for high activity levels ψ	Parameters	AICc	Δ AICc	Weight
Indiana myotis + northern myotis along forest edges	5	450.568	0.000	0.494
Indiana myotis + northern myotis	5	452.263	1.695	0.212
Indiana myotis + big brown bats in forest corridors	5	454.538	3.970	0.068
Indiana myotis + big brown bats along forest edges	5	454.584	4.015	0.070
Indiana myotis + Eastern red bats in forest corridors	5	455.027	4.458	0.053
Indiana myotis + tri-colored bats	5	455.047	4.479	0.053
Indiana myotis	4	455.950	5.381	0.034
Northern myotis along forest edges	4	457.881	7.312	0.013
Northern myotis	4	461.048	10.479	0.003
Constant	3	461.451	10.883	0.002

Table 6.—Model-averaged regression coefficients and uncertainty estimates for competing models describing high activity levels by bats within existing forest openings.

Model parameters	Regression model coefficients	Standard error	Lower 95% CI	Upper 95% CI
ψ = *Pr* (high activity)				
Intercept	1.447	0.796	-0.113	3.007
Indiana myotis	-2.141	0.817	-3.742	-0.540
Northern myotis along forest edges	-2.216	1.076	-4.325	-0.107
Northern myotis	-0.537	0.479	-1.476	0.402
p = *Pr* (detection of high activity)				
Intercept	-0.690	0.257	-1.194	-0.186
Eastern red bats	0.696	0.339	0.032	1.360

Table 7.—Model-averaged parameter and uncertainty estimates based on competing models describing high activity levels by bats within existing forest openings in stands before implementation of silvicultural treatments.

Model parameters	Parameter estimates	Standard error	Lower 95% CI	Upper 95% CI
ψ = *Pr* (high activity)				
Indiana myotis	0.333	0.117	0.151	0.585
Northern myotis along forest edges	0.236	0.197	0.035	0.725
Northern myotis in other habitats	0.695	0.202	0.260	0.937
Other bat species	0.809	0.122	0.475	0.952
p = *Pr* (detection of high activity)				
Eastern red bats	0.501	0.096	0.321	0.682
Other bat species	0.334	0.057	0.233	0.453

DISCUSSION

After accounting for the probability of detection, we found that forest stands proposed for uneven-aged silvicultural treatments had a greater probability of high bat activity when compared to forest stands designated for other treatments. When examining the existing data for these stands, we could find no obvious reason to account for this difference. Also, interior forest locations prior to silvicultural treatment were more likely to experience high activity levels by northern myotis relative to other species. In contrast, the probability of high activity levels within existing forest openings was lower for both northern myotis and Indiana myotis relative to other species. Among pre-existing forest openings, northern myotis activity appeared to be especially low along forest edges.

The actual estimates of probability of use in general had wide confidence intervals. This variation likely reflects the minimal sampling of two occasions when estimating detection probability during a year. Given this caveat, the trends in our current modeling effort can help develop expectations of the effect of planned silvicultural treatments on the activity of foraging bats (Yates and Muzika 2006). Although use by Indiana myotis and northern myotis appeared to differ relative to other species, the actual estimates for the probability of high activity levels were similar at both forest interior locations and within existing forest openings. Other bat species had higher activity levels in existing forest openings when compared to interior forest locations. Based on these results, we expect that creating forest openings via silvicultural treatments in our experimental stands will increase the frequency of high activity levels by other species of bats, and maintain a similar level of activity by both Indiana myotis and northern myotis.

The probability of an interior forest location's experiencing low activity levels was estimated to be high with upper confidence intervals nearing 1. High activity levels were described as greater than 4 minutes of recorded activity during a night, which was the upper quartile of all recorded activity across all species. Low activity levels in interior forest sites were common, but inconsistent on a nightly basis. These low activity levels could reflect the lower probability of detecting bat activity at interior forest sites relative to existing forest openings. It also could be representative of the behavioral pattern of bats foraging in a cluttered environment.

The greater activity by bats in existing forest openings is likely due to a reduction in structural density, or "clutter." Our results were consistent with previous studies of bats using forested habitats in the eastern United States (Menzel et al. 2002, Owen et al. 2004, Titchenell et al. 2011). The greatest amount of nightly

activity was recorded in pre-existing harvest openings and along the forest edge. Bats that cannot tolerate clutter spend much of their time foraging in these types of openings within forest stands (Hogburg et al. 2002, Owen et al. 2004, Patriquin and Barclay 2003, Titchenell et al. 2011). Additionally previous studies have found that forest edges may experience lower levels of activity by some species in the genus *Myotis*, such as the northern myotis, that roost and forage in the forest (Menzel et al. 2001, 2005a; Patriquin and Barclay 2003). Forest interior sites, which are characterized by greater clutter, appeared to be avoided by most bat species (Loeb and O'Keefe 2006) and thus were occupied least.

Activity by Bat Species

As expected, higher levels of activity by big brown and eastern red bats were recorded in pre-existing harvest openings and edges compared to other habitats. This difference is consistent with wing morphology that is specialized for habitats with minimal clutter (Elmore et al. 2004, Norberg and Rayner 1987). Big brown bats forage in open uncluttered environments (Duchamp et al. 2004, Loeb and O'Keefe 2006). Although this species typically exploits human structures for roosts, it is known to commute several kilometers between roosting and foraging locations (Duchamp et al. 2004). This species may commute into and out of Morgan-Monroe and Yellowwood State Forests from human structures outside of the study area. In contrast, eastern red bats roost in foliage (Mager and Nelson 2001, O'Keefe et al. 2009) within the forest and characteristically exploit gaps within the forest. Yates and Muzika (2006) found that eastern red bat habitat use was more likely in stands with more open understory.

The northern myotis, Indiana myotis, and the tri-colored bat used forest interior habitat at relatively high levels. However, the northern myotis and the Indiana myotis used forest openings less relative to other species. The northern myotis, a clutter-adapted species, is considered to prefer continuous forests and older forest stands (Loeb and O'Keefe 2006, Owen et al. 2003), and could be negatively affected by openings created during silvicultural treatments. However, Yates and Muzika (2006) found no evidence that northern myotis were negatively affected by fragmentation. With silvicultural treatments that create larger openings that are less cluttered, northern myotis may prefer forest structured like Morgan-Monroe and Yellowwood State Forests as of 2007: 30-80 years in age and a timber harvest regime based primarily on single-tree and small-group selection cuts. Indiana myotis can tolerate some clutter, but are more often detected along forest edges, forest openings, and corridors (Sparks et al. 2005). The tri-colored bat is considered a clutter-adapted species (Menzel et al. 2005b). In our study, however, it appeared to exploit both forest interior and forest openings on a regular basis and should perhaps be considered a habitat generalist (Loeb and O'Keefe 2006).

High activity levels were rarely recorded for little brown myotis and hoary bats during our study, and neither was included in our occupancy models. The echolocation call (Arita and Fenton 1997) and medium wing loading (Broders et al. 2004) of little brown myotis suggest this species uses a moderately cluttered habitat. They are known to forage along edges (Hogburg et al. 2002) and over still water (Barclay and Brigham 1991), which was rare in the upland study area. A lack of still water for foraging and few anthropogenic structures for day-roosts may explain the relative rarity of this species in Morgan-Monroe and Yellowwood State Forests. Little brown myotis, like big brown bats, may use corridors to commute from roosts outside the forest. The echolocation calls and high wing loading of hoary bats suggest the use of a clutter-free environment (Barclay et al. 1999, Norberg and Rayner 1987).

Although all seven species of bats captured during concurrent mist-net surveys (Sheets, Whitaker, et al.,

this publication) also were detected acoustically, two additional species, silver-haired bat (*Lasionycteris noctivagans*) and evening bat (*Nycticeius humeralis*), occur in this region of Indiana (Whitaker et al. 2007). Evening bats were not captured during 3 years of mist-net surveys (Sheets, Whitaker, et al. this publication) and were not included as potential species to be identified during acoustic surveys. Silver-haired bats are not thought to be summer residents in the area (Whitaker et al. 2007). The evening bat does occur in the region during the summer months, but the distribution is spotty and it is most often found in lowland areas rather than upland forests such as those sampled for this study (Whitaker et al. 2007).

Like every sampling method, bat detectors have biases. In forests, clutter from vegetation, vertical vegetation layering, call intensity, weather conditions, theft, and vandalism, among others, may affect the accuracy, quantity, and quality of calls detected (Duchamp et al. 2006, Hayes 2000, Weller and Zabel 2002). We estimated variation in detection probability between species and habitat types so that it did not bias our comparisons of activity levels. We also maintained a uniform sampling space directly in front of our microphones by directing bat detectors away from roads and heavily cluttered stands of vegetation. Neither vandalism nor theft occurred. We avoided sampling on nights with prolonged periods of adverse weather conditions. On occasion, temporary adverse weather conditions may have occurred, but they should not have affected overall activity levels during an entire night.

CONCLUSIONS

When testing for differences in the use of forest stands selected for silvicultural treatments, we found that stands selected for uneven-aged management had higher levels of bat activity than our other stands. Although we are not sure of the reasons, we will need to be aware of this pre-disposition as we monitor the response to silvicultural treatments as part of the Hardwood Ecosystem Experiment. We also found differences in our ability to detect bats acoustically among bat species and between forest habitats. These differences emphasize the importance of continuing to estimate detection probability as the Hardwood Ecosystem Experiment proceeds.

Finally, our results also provide some insight into the potential responses of bats to silvicultural treatments. When interior forest locations were compared to a variety of forest openings, there was a consistent response of increased use of openings by bat species that are adapted to foraging in open spaces. Bats adapted to foraging in an environment with higher densities of vegetation used these forest openings along with the interior forest locations during foraging flights.

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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.

PRE-TREATMENT ASSEMBLAGES OF WOOD-BORING BEETLES (COLEOPTERA: BUPRESTIDAE, CERAMBYCIDAE) OF THE HARDWOOD ECOSYSTEM EXPERIMENT

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Abstract.—Longhorned beetles are a diverse and important group of insects in forest ecosystems; several species attack weakened or stressed trees, relatively few attack healthy trees, and most species use only dead and decomposing wood. We surveyed longhorned beetles and metallic wood-boring beetles using four different types of traps at 36 Hardwood Ecosystem Experiment (Indiana) locations that spanned all silvicultural treatments from 2006 through 2008. During two of these pre-treatment years we conducted surveys on sites destined to be within treatment areas, and in one year we sampled within the landscape outside the treated areas. We found no differences in the species richness or diversity between sites destined for future treatments. We identified a difference between future treatment and non-treatment sites that appeared to be caused by sampling them in different years. We discuss the importance of taking generation times of the species into account in future surveys.

INTRODUCTION

The hardwood forests of Indiana are one of the state's greatest natural resources. Indiana's forests support nearly 130,000 workers and provide an estimated $17 billion annually in state revenue (Indiana Department of Natural Resources-Division of Forestry 2010). Additionally, the hardwood industry in Indiana consistently rates as one of the most important elements of the state's economy (e.g., Piva and Gallion 2007). Understanding how to best manage the wood resource is an important issue facing Indiana foresters, especially when the introduction of invasive species already threatens the balance and stability of the state's forests. The Asian longhorned beetle (*Anoplophora glabripennis* Motschulsky) alone has cost close to $400 million to control in its first decade in North America (Haack et al. 2010), and the emerald ash borer (*Agrilus planipennis* Fairmaire) has killed millions of ash (*Fraxinus*) trees in both natural and urban environments throughout northeastern North America (Poland and McCullough 2006). The need to improve management strategies concerning invasive species is clear, but the basic knowledge of how biogeography, biodiversity, landscape structure, and land management impact the establishment and spread of exotic species—and endemic species—is incomplete (Turner 2005). Knowledge of how both pest and non-pest xylophagous beetles respond to different forest management systems will be crucial for safeguarding the hardwood industry in Indiana.

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Longhorned beetles (Coleoptera: Cerambycidae) can have a large impact on this industry by killing valuable hardwood trees and degrading lumber (Linsley 1959). For example, larvae of the red oak borer (*Enaphalodes rufulus* Haldeman) thrive in stressed oak (*Quercus*) trees, and recent evidence shows that it may also successfully complete development in healthy oaks (Fierke et al. 2005). This beetle was a major factor in the widespread mortality of oaks in the Ozark National Forest of Arkansas (Haavik et al. 2010, Heitzman 2003). Painted hickory borer larvae (*Megacyllene caryae* Gahan) infest recently cut or killed hickory (*Carya*) (Craighead 1950) and can reach high densities within the wood (Holland 2009a), greatly degrading the value of lumber because of the extensive galleries they excavate throughout both sapwood and heartwood (Blatchley 1910, Craighead 1950). Similar damage from galleries of *E. rufulus* greatly reduces lumber value in oak (Donley 1974). Approximately one-quarter of the species of longhorned beetles in Indiana are considered pest species that cause ecological and economic losses.

Metallic wood-boring beetles (Coleoptera: Buprestidae) have a life history similar to longhorned beetles, but a larger proportion of species that are pests on living trees. Most species of Buprestidae attack weakened, dying, or recently dead trees. Females generally lay eggs singly or in small clusters in crevices on the bark of a host tree. Eggs hatch and larvae bore through the bark to the interface of the sapwood and cambial tissue, where a majority of larvae remain and feed (Drooz 1985). Some genera, such as *Dicerca* and *Texania*, usually mine through both the sapwood and heartwood of their respective hosts. Once ready to pupate, larvae of some species construct a pupal chamber just under the bark and overwinter there, while others create a chamber in the sapwood near the bark. The next spring, after pupation occurs, adults emerge by chewing through the bark, usually leaving a D-shaped exit hole (Solomon 1995). Not all genera or species in a genus are restricted to tree trunks, and they do not necessarily need a stressed host. *Brachys* and *Taphrocerus* larvae, for example, are leaf miners and feed between the upper and lower dermal layers of leaves (Bright 1987).

The ecological and economic impacts of some wood-boring beetles that attack living and recently cut trees and lumber are important, but most species are not pests (Evans et al. 2004). Rather, larvae of many species develop within decaying wood, often logs on forest floors or dead limbs and twigs on standing trees. These insects using dead wood make up an important and diverse component of forest biodiversity. By decomposing dead wood in forests, these species may contribute to nutrient cycling and to reducing the severity of forest fires by hastening decomposition of coarse woody debris (Gutowski 1987). It has even been suggested that wood-feeding insects can aid in forest productivity by cycling nutrients from weakened trees to the soil and thence to vigorous trees (Berryman 1986). Larvae of many species provide food for both invertebrate and vertebrate animals and create habitat for many species within dead wood (e.g., Holland 2009a). When adult beetles disperse and mate, many species act as pollinators while seeking floral resources (Linsley 1961) and are commonly collected on flowers (e.g., Bond and Philips 1999, Gosling 1984, Ødegaard and Frame 2007).

Different forestry management regimes influence the composition and abundance of wood-boring beetles in forests. Wood-boring beetles respond readily to disturbances such as fire and windthrow (Allison et al. 2004, Hopkin et al. 2001). Beneficial wood-boring beetles that provide ecosystem services are more abundant in managed forest areas where harvesting has left slash and dead wood. A study on the saproxylic beetles (those dependent on dead wood) in Australia showed that many species found within the old-growth assemblage did not persist in regrowth forests that had been logged (Grove 2002a). Some species of wood-boring beetles may increase in abundance

following management operations that either leave abundant freshly killed wood or alter environmental conditions and place stress on surrounding uncut trees. Other studies have shown that many non-pest species, whose major role is decomposition of dead wood and pollination, may be negatively influenced by forest management. Nonetheless, many species appear to prefer habitat with some canopy openings (Barbalat 1998).

It would be beneficial to understand how different forest management strategies affect both pests and non-pest species that play roles in ecosystem services (Gutowski 1987). There is much recent interest in northern Europe in conserving remaining saproxylic insects. Much of the region has a long history of intensive management that has greatly reduced coarse woody debris and thereby threatened many species (McGeoch et al. 2007). Ten longhorned beetle species are listed on the International Union for Conservation of Nature (IUCN) Red List of endangered species (IUCN 2010).

Adults of some wood-boring beetles, such as longhorned beetles in the genus *Saperda*, prefer forest edges (Hammond 1997). Within forests, species richness and abundance of both longhorned beetles and metallic wood-boring beetles are higher near clearings (Barbalat 1996). Forestry operations that create edge habitat and openings may lead to an increase in the abundance of several species. Forest harvest operations can also influence the abundance of longhorned beetles by altering the amount of dead wood available for larval development. Harvesting creates a large sudden input of dead wood in the harvest area that slowly decreases over time as it decays. Ulyshen et al. (2004) found that the abundance and diversity of several saproxylic beetle families including cerambycids initially increased in small openings of 0.13-0.5 ha, above the levels seen in surrounding forests. However, 6-year-old clearings had a fauna that was reduced below that of the surrounding forest.

Openings from forest operations can also increase the habitat quality for many species of saproxylic beetles (Barbalat 1996) by increasing light penetration (Barbalat 1998). Finally, harvest openings have the effect of lowering the forest mantle within the harvest area. Wermelinger et al. (2007) define the forest mantle as the outer surface layer of the three-dimensional forest volume, and found that most cerambycids tended to be caught in the lower mantle and open areas. The variety of responses within the wood-boring beetles to different habitat types, and climatic conditions caused by gaps and edges, may be partially responsible for the variation in the response to forest disturbance seen within this group (e.g., Saint-Germain and Drapeau 2011).

The main factor influencing the occurrence of species of xylophagous beetles is host plant presence (Barbalat 1998). Different tree species vary in the richness of their associated wood-boring fauna (Barbalat 1998). Forest management operations that selectively promote or remove some tree species over others may influence the species richness and compositions of the native saproxylic communities. Selective operations will have a larger influence on the wood-boring species that attack and develop within living trees than on those species that develop within decayed wood, as the former are more host-tree specific (Hanks 1999, Linsley 1959).

The presence of suitable larval substrate is one necessary condition for a wood-boring species to be present; another is that adults must have the ability to reach the area containing the substrate (Grove 2002b). Forest fragmentation has extirpated at least one species of longhorned beetle from a large part of the host plant range, which includes much of Indiana (Holland 2009b,c). At smaller scales, saproxylic beetles respond to the connectedness of the network of dead wood on the forest floor within 150 m of surveyed points (Schiegg 2000). The movement abilities of longhorned beetles vary greatly, from flightless species that may move 100 m in 10 days (Baur et al. 2002) to some that

can track host plant volatiles over a distance of >0.8 km in less than an hour (Beeson 1930).

Several terrestrial insect groups including wood-boring beetles are considered to be important bioindicators of forest health and the effects of ecosystem management. Indications of forest ecosystem health and the effects of management can be assessed by studying changes in the density and species composition of these insect populations (Werner 1996). Longhorned beetles and metallic wood-boring beetles are considered to be among the most important indicators of biodiversity in forest ecosystems because of their diversity, specific habitats, and tree host species (Speight 1989). Because of this indicator value, along with the diversity of roles wood-boring beetles play both as pests and as beneficial insects, these beetles are an ideal group for studying the impact of timber harvesting practices on forest biota.

Our objective in this study was to determine the species of longhorned beetles and metallic wood-boring beetles present at the study sites of the Hardwood Ecosystem Experiment (HEE) before harvests were initiated. We tested the hypothesis that there was no difference in these beetle assemblages between the different (future) harvest types before treatment began, and the hypothesis that there was no difference between the beetle assemblages of future harvest and non-harvest areas. Differences in the guilds of wood-borers (e.g., species that develop in live trees versus rotting wood) will be the focus of future publications. A second objective was to determine how species with generation times greater than 1 year could explain any differences found.

STUDY AREA

All nine research cores making up the HEE were surveyed in the wood-boring beetle study. These replicated landscapes are each 78-110 ha of forest within Yellowwood State Forest or Morgan-Monroe State Forest in Indiana (Kalb and Mycroft, this publication). Three of these research cores were designated for treatment with one of the three types of silvicultural regeneration systems, no-harvest controls, uneven-aged management, and even-aged management, so that each system was replicated three times. To eliminate neighborhood effects each research core, regardless of the treatment, was surrounded with an approximately 300-ha buffer wherein single-tree and small-group selection harvests occurred. Within the even-aged research cores there were four intensive sampling units (ISUs) of approximately 4 ha each that would later be subjected to either clearcutting or shelterwood harvest (Fig. 1). These ISUs are so named because efforts to study the effects of harvest treatments are focused within them, although not exclusively.

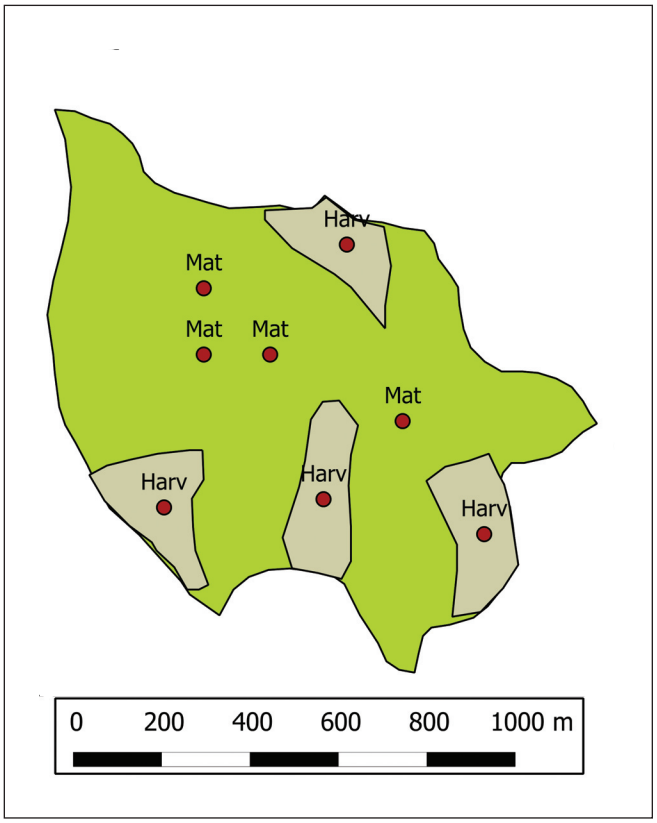


Figure 1.—Example research core showing location of intensive sampling units (ISUs) and beetle trap array locations. This even-aged research core (green) contains four 4-ha clearcuts (gray). Arrays within harvested ISUs (sampled in 2006 and 2007) are indicated by a circle labeled Harv. Arrays within the matrix (sampled in 2008) outside ISUs are indicated by a circle labeled Mat. Surrounding buffer area is not shown.

One ISU of each treatment was located on north- to east-facing slopes and one of each treatment was located on south- to west-facing slopes. The uneven-aged research cores contained eight ISUs of 0.5-2.0 ha that were later subjected to harvest. These smaller openings would rarely span ridge top to ridge bottom, and were therefore stratified across physiographic locations. The uneven-aged research cores also were subjected to single-tree harvest throughout the unit. The control units had four ISUs where no timber harvesting was to be carried out, and these also were stratified evenly among north- to east- and south- to west-facing slopes. We placed one array of traps within each of the four ISUs within all even-aged and control research cores (Fig. 1), and within four of the eight ISUs within each uneven-aged research core. Additional details on the HEE study design can be found in Kalb and Mycroft (this publication).

MATERIALS AND METHODS

Trap arrays in 2006 and 2007 within each ISU were approximately centered on the bird survey tree closest to the center of the unit (Kalb and Mycroft, this publication). Traps were randomly designated to a cardinal direction and placed approximately 20 m from the tree in this direction. They were hung so that the bottom of the trap was 1.5-2 m above the ground. The four traps making up each array were: one Lindgren multiple-funnel trap (12 funnel model; Pherotech, Delta, BC), one Panel Trap for Bark Beetles (Alpha Scents, Portland, OR), one intersecting pane window trap of our own design, and one linear purple sticky trap modeled after Oliver et al. (2005). For details on traps see Holland (2010).

We used 0.61m x 0.61m rain covers on the first three trap types, as we have found that covers greatly reduce dilution of trap fluid by rain. On each trap except the purple trap, we hung a 125-ml Nalgene bottle containing 60 ml of 99 percent ethanol that evaporated out of four 2-mm holes in the top. The beetles are assumed to be responding to the ethanol lure (Montgomery and Wargo 1983) and the dark vertical silhouette of the trap (Lindgren 1983), but it is possible that some species are not able to use primary attraction to assess individual trees for suitability (Saint-Germain et al. 2006, 2007a). The collection jar used for all but the purple traps had many centimeters of ethylene glycol to kill and preserve the beetles. A few drops of liquid soap were added to the killing fluid in each trap to limit its surface tension. Purple traps were covered with clear plastic sheeting before being coated with Tangle Trap (Tanglefoot Co., Grand Rapids, MI).

In 2008, we followed the same sampling methods but located trapping arrays outside the ISUs where the initial harvest treatments were to take place. These traps are considered to be located within the matrix of each research core. These arrays were used to examine the changes in beetle assemblages in the larger landscape outside of harvests. They were located at bird survey points that were at least 200 m from any treatment area, 50 m from any road or trail, and 100 m from any previously surveyed beetle point. Within each research core, we randomly selected four bird survey points from those that met these criteria.

Every 3 weeks we removed all insects from the traps. The plastic sheet on the purple sticky traps was removed and replaced with a new sheet. All longhorned beetles and metallic wood-boring beetles were removed from the catch in the laboratory, pinned, and identified using guides or keys by Lingafelter (2007), Linsley (1962a,b; 1963; 1964), Linsley and Chemsak (1972, 1976), and Yanega (1996). Buprestidae were identified using keys by Downie and Arnett (1996), Fisher (1928, 1942), and Wellso and Manley (2007). All specimens currently reside in the insect collection of the Biodiversity and Landscape Ecology Laboratory in the Department of Entomology at Purdue University.

We used adonis in the vegan package (Oksanen et al. 2011) for R (R Development Core Team 2010) to compare the beetle assemblages between research

cores with different planned harvest techniques. Adonis is a multivariate analysis of variance (ANOVA) that uses distance matrices and tests for differences using permutation (Oksanen et al. 2011). We used 1,000 permutations of the data to generate the null distribution and constrained the permutation according to year. In addition, we included in the adonis model a term to compare the beetle assemblages of sites within the sites destined for future harvests (all ISU regardless of planned harvest technique) and sites outside the future harvests. We label these latter sites matrix sites. The model included a year term as well.

For each site we summed the abundances of each species within each trapping array and within each year. We therefore did these analyses with 108 array-years: 3 yr x 9 units x 4 arrays/unit. Cerambycidae and Buprestidae species that were found at fewer than 5 percent ($n<6$) of the locations across the 3 pre-treatment years were omitted from the multivariate adonis tests for statistical reasons. Because we did not dissect logs to ensure the species caught were actually using the host wood material in the immediate area (Saint-Germain et al. 2007b), it is possible that these less commonly captured species were incidental. We then calculated the Bray-Curtis similarity on square root abundance for the remaining species. We did not standardize, so more abundant species have a larger influence on the similarity measure.

To visualize the differences in groups, we performed a non-metric multi-dimensional scaling (NMDS) on the square root abundances of all species caught using the package MASS (Venables and Ripley 2002) in R. To determine the species that were driving differences between the sampled assemblages, we used a custom R function to perform a similarities percentage (SIMPER) analysis (Clarke and Warwick 2001). This procedure uses the contribution of each species to the Bray-Curtis dissimilarity measure to determine species that are largely responsible for separating groups. If there were pair-wise differences between the assemblages at sites destined for different treatments, we used an iterative procedure to determine the subset of species causing the differences. We first removed the species that most contributed to the dissimilarity. This process was repeated until the treatments were not significantly different and the species thus removed were considered the major drivers of the differences. We then examined the generation time of the species in this subset.

Shannon diversity was calculated for all sites using the vegan package in R. We included all species in this diversity calculation and so may include some incidental captures. We used ANOVA to test for differences in diversity and species richness between the future treatment groups. We used the vegan package in R to plot a species accumulation curve, to determine how well our sampling captured the majority of species present.

RESULTS

We caught 2,058 individuals from 6 subfamiles and 85 species of longhorned beetles and 400 individuals from 17 species of metallic wood-boring beetles. We did not detect any species that are recent exotic invasive species. Some species are exotics that have been in North America for a considerable time (e.g., *Xylotrechus colonus*) and are likely now considered well established and a permanent component of the fauna. The shorter period of sampling in 2006 led to lower numbers in that year.

The species accumulation curve showed that the species richness of wood-boring beetles was leveling off (Fig. 2), suggesting that the sampling effort was adequate for determining most of the species present. There were no differences among trapping arrays of future harvest treatments (control, even-aged management, uneven-aged management) within the research cores ($p>0.05$, Fig. 3A). However, adonis did reveal differences between the assemblages of future harvest sites (ISU) and matrix sites ($p < 0.05$, Fig. 3B).

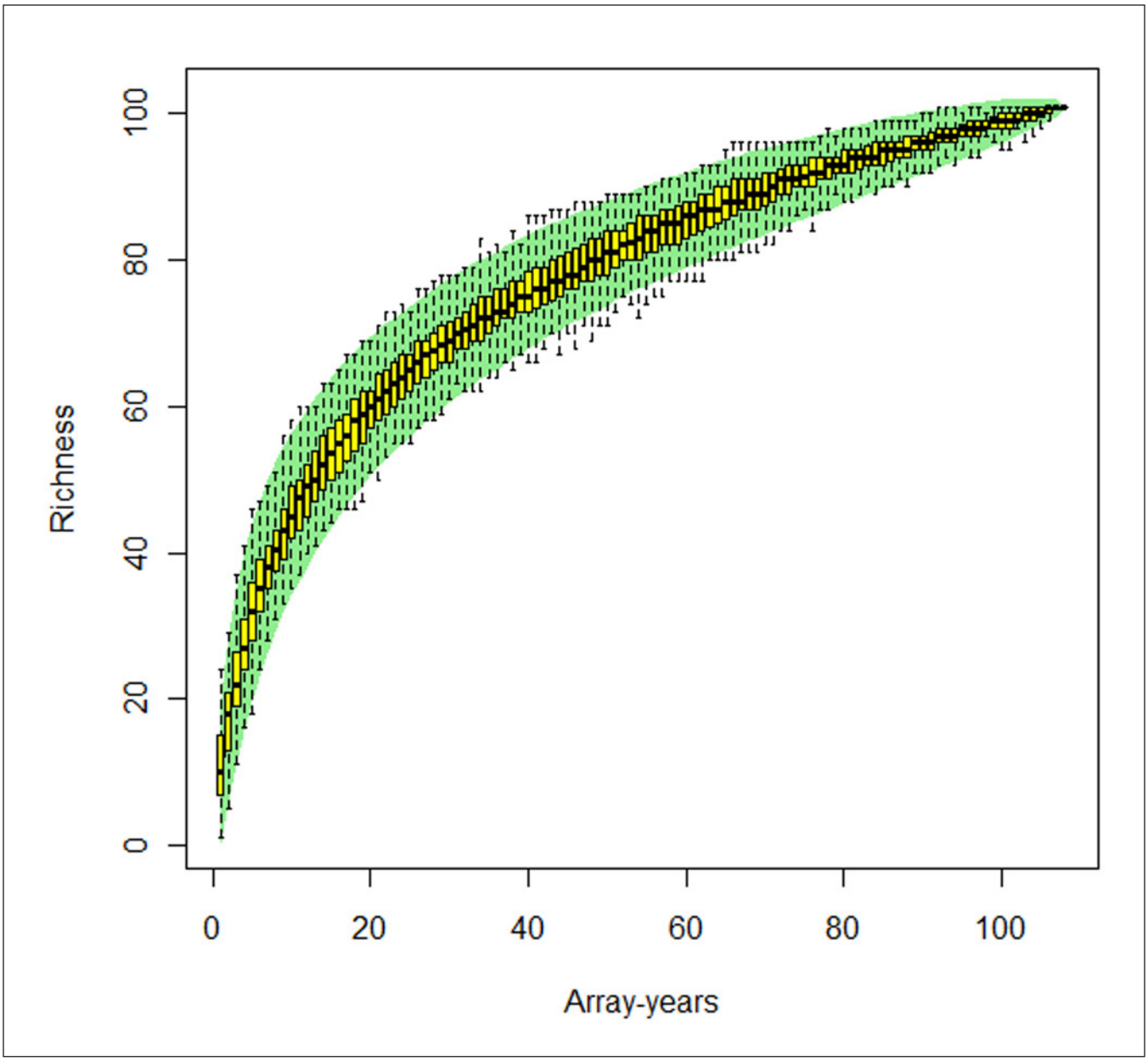


Figure 2.—Species accumulation curve for the 108 site-years sampled within the pre-treatment study. Sampling effort is given as total array-years. Boxes are first and third quartiles and whiskers are range, all derived from randomizing the order of including site-year data. A total of 102 species were identified.

These differences appear to be a result of logistical constraints forcing us to sample harvest and matrix sites in different years.

A year term in the adonis model revealed that year may be a factor behind changes in the community assemblage ($p < 0.05$, Fig. 3C). The betadisper function in the vegan package revealed overdispersion in the 2006 data compared to the other years ($df = 2,105$, $F = 17.3$, Tukey post-hoc comparisons: 2006-07: $p < 0.0001$; 2006-08: $p = 0.0005$; 2007-08: $p = 0.15$). This result was not altered by using Jaccard's dissimilarity on presence-absence data. We attempted to remove the heterogeneity of dispersion by including only species caught during the 2006 season, but doing so did not alter the

results or cure the dispersion heterogeneity. We were able to compare future harvest and matrix sites by comparing only the 2007 and 2008 data. This adonis test revealed that assemblages differed according to future treatment (df = 2, F = 1.6, p = 0.016).

The arrays differed in species assemblage when grouped by year sampled, and therefore also when grouped into ISU or matrix sites. The species that contributed to these differences are listed in Table 1. At least half of these 14 species have, or may have,

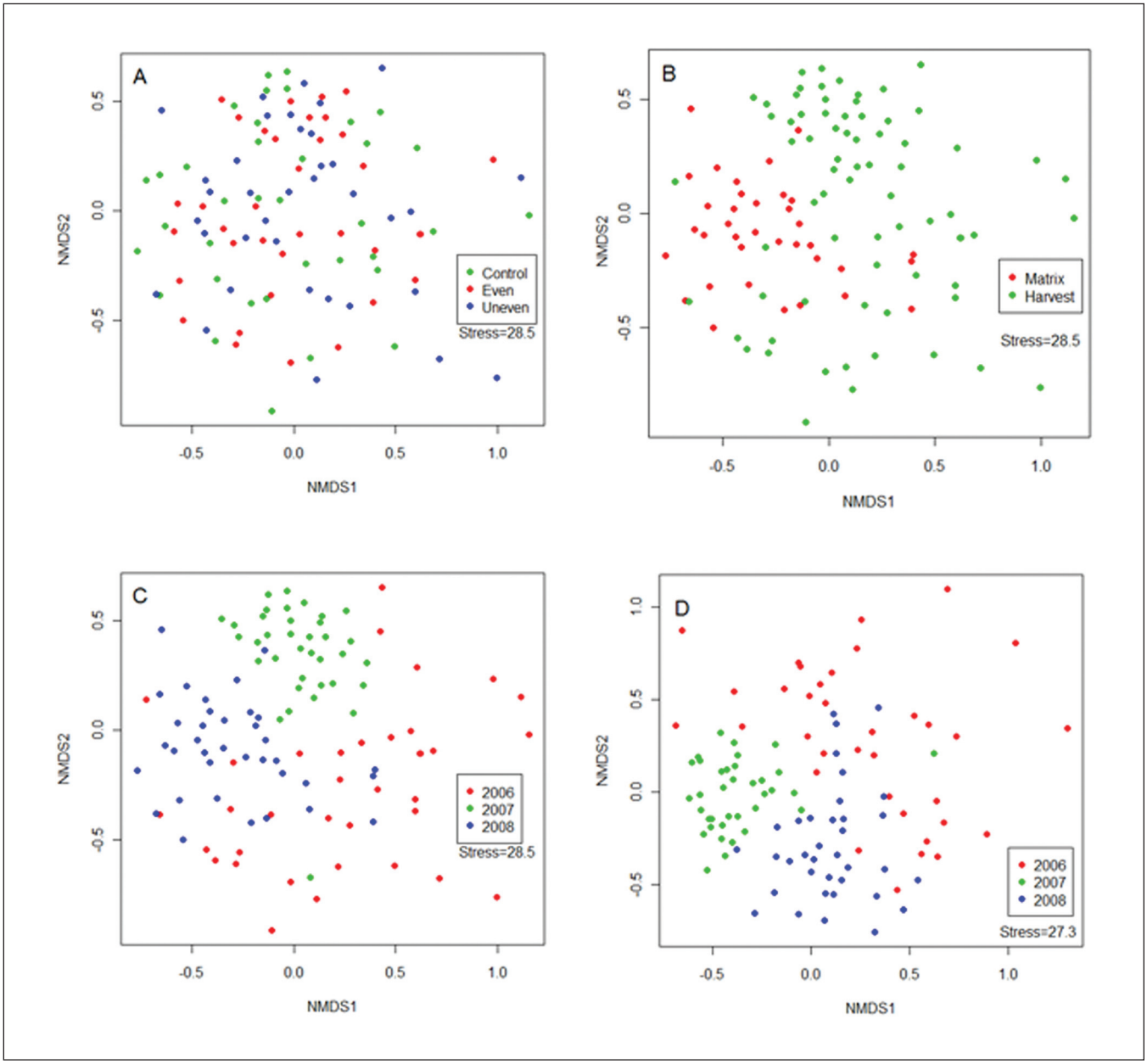


Figure 3.—Non-metric multidimensional scaling plots of 108 site-year samples where wood-boring beetles were sampled, using data from those species found at greater than 5 sites. Plots (A) through (C) represent the relative inter-sample dissimilarities of each site-year in 44-dimensional species-space. Plot (A) has samples coded according to the three future treatments: control, even-aged management, and uneven-aged management. Plot (B) has sites coded according to whether they are located in future matrix (not harvested) sites or future harvest sites. Plot (C) shows samples according to the year in which they were obtained. Plot (D) is an ordination showing the relative location of all 108 sites using only the subset of species (n=14) found by the SIMPER analysis to be driving the differences between years.

Table 1.—The 14 species of wood-boring beetles that account for the differences among species assemblages between years. At least half of these species are likely to spend more than 1 year in the larval stage. For four species, we did not find voltinism data in the literature.

Family	Scientific name	Voltinism	Source
Buprestidae	*Agrilus bilineatus* Weber	1	Bright 1987
Buprestidae	*Agrilus cephalicus* LeConte		
Buprestidae	*Dicerca divaricata* Say	2-3	Bright 1987
Cerambycidae	*Analeptura lineola* Say		
Cerambycidae	*Anelaphus villosus* Fabricius	2	Solomon 1995
Cerambycidae	*Clytus ruricola* Olivier		
Cerambycidae	*Cyrtophorus verrucosus* Olivier	1	Linsley 1964
Cerambycidae	*Elaphidion mucronatum* Say	2	Linsley 1964
Cerambycidae	*Molorchus b. bimaculatus* Say	1-2	Blackman and Stage 1924
Cerambycidae	*Neoclytus a. acuminatus* Fabricius	1-2	Blackman and Stage 1924
Cerambycidae	*Orthosoma brunneum* Forster	2-3	Linsley 1962a
Cerambycidae	*Typocerus v. velutinus* Olivier	2	Linsley 1976
Cerambycidae	*Urgleptes querci* Fitch		
Cerambycidae	*Xylotrechus colonus* Fabricius	1	Blackman and Stage 1924

a generation time greater than 1 year. Much of the dissimilarity between assemblages sampled in different years is due to these 14 species (Fig. 3D). This analysis should be viewed as a post-hoc test, and the voltinism hypothesis remains to be tested in future work.

No differences were found in the Shannon-Wiener diversity estimates at sites scheduled for future treatments ($df = 1,106$, $F = 0.238$, $p = 0.627$, Fig. 4). The Simpson diversity index showed similar results ($df = 1,106$, $F = 0.075$, $p = 0.785$) and were not plotted. Species richness was also similar among arrays in different future treatments ($df = 1,106$, $F = 1.2608$, $p = 0.2640$, Fig. 4). Adjusting the degrees of freedom to account for the arrays being nested within research cores (4 arrays per unit) would not change the outcome that there are no differences in these metrics.

DISCUSSION

The number of individual wood-boring beetles captured by this sampling effort is similar to other studies carried out in Indiana using similar techniques (e.g., Holland 2006). The species richness is slightly higher than another survey carried out by Holland in Indiana, likely because several species that are sensitive to forest fragmentation at large spatial scales remain extant in the heavily forested landscapes of south-central Indiana where the present study was conducted. Several uncommon species (Appendices 1 and 2) were found that had not previously been found in a survey across Indiana (Holland 2006), and we will publish several new state and county records as a result. Most species of Buprestidae commonly collected in this study are known to use oak as a host. The only species that we captured in large numbers that does not have oak recorded as a host were *Dicerca lurida* Fabricius and *Agrilus cephalicus* LeConte. *Dicerca lurida* is known from *Alnus*, *Carpinus*, *Carya*, *Prunus*, and *Salix*, while *A. cephalicus* is known from *Cornus* (Nelson et al. 2008).

As expected, there did not appear to be differences among the trap arrays in ISU destined for different treatments in the future (Fig. 3A), but we were unable to test this hypothesis statistically because data were overdispersed in 2006. The comparison with 2007 and 2008 data did show differences among future treatments. The differences between the ISU array locations sampled in 2006 and 2007 were different

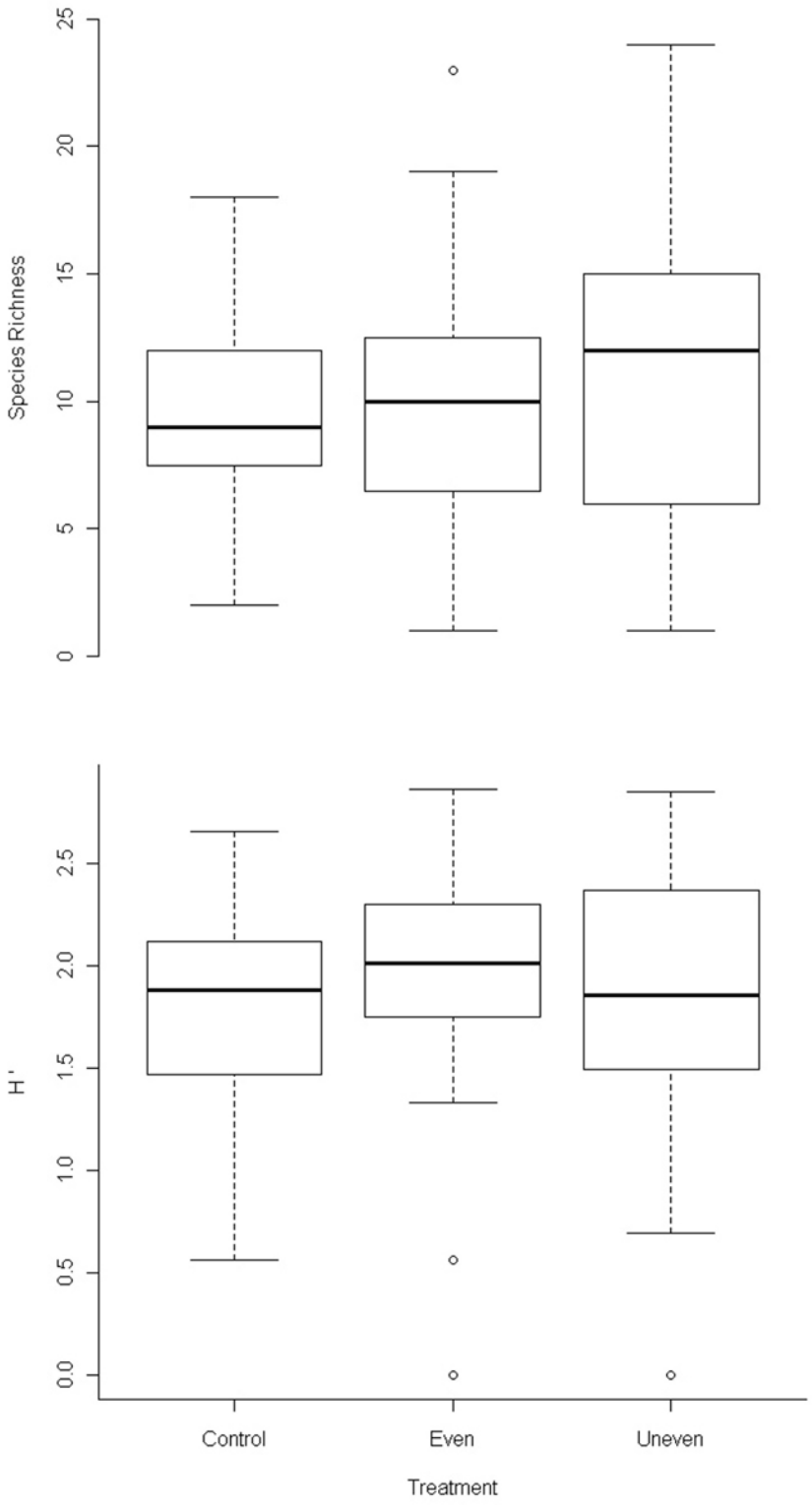


Figure 4.—No differences were found in species richness or Shannon diversity index (H') at future control, even-aged management, and uneven-aged management sites. Thirty-six sites were sampled across the future treatment groups.

from the landscape matrix locations sampled in 2008 (Fig. 3B). This finding would seem at first to suggest that later comparisons between harvested areas and the landscape around them would not be possible as differences were already evident before harvesting. However, there are several possible reasons for the differences between these years. Taking these differences into account may help with future comparisons.

In 2006, logistical constraints led to a much shorter collecting period, which influenced year-to-year comparisons. The summer of 2007 in south-central Indiana was slightly warmer and drier than average, with mean temperature for June-August (24.3 °C, 75.8 °F) 2 percent above average and rainfall (206 mm, 8.12 inches) only 66 percent of normal summer conditions. By comparison, summer 2008 mean temperature (23.4 °C, 74.2 °F) was approximately average, and summer rainfall (271 mm, 10.7 inches) was 87 percent of the long-term average (Indiana State Climate Office 2011). The warmer and drier conditions of 2007 may have altered the species assemblage or flight activity of the beetles, making annual comparisons difficult. Ideally, samples in different groups (harvest or matrix) should be taken at the same time. Unfortunately, logistical constraints prevented us from sampling matrix and harvest locations in the same years.

Another possible cause for the different assemblages in different years is that many species of wood-borers remain several years in the larval stage. Some species of wood-borers with generation times longer than 1 year are known to synchronize adult emergence, leading to year-to-year fluctuations that can influence the number and species of beetles captured (Barbalat and Borcard 1997). We tested this relationship in a post-hoc way, and found that at least half of the species that were responsible for distinguishing the year-to-year differences, and thus the future harvest site and landscape matrix site differences, have, or may have, generation times longer than 1 year (Table 1). If these species undergo variation in their annual abundance caused by synchronization of adult stages, such variation would have contributed to the differences we report.

The species listed in Table 1 represent less than 20 percent of the species captured. Consequently, one way to minimize the influence of longer-lived species on comparisons would be to exclude these species, but this step may bias results toward species that use living wood. The latter species tend to have shorter generation times, likely because of the higher-quality larval food. A better approach would be to have a priori information on the species that are unlikely to be in the adult stage for a particular year and either adjust the timing of sampling to obtain the most species possible, or remove those species previously determined to have most individuals in the larval stage. Combining data from several years of surveying may also allow us to minimize the influence of annual variation.

The oak forests in the HEE support a diverse assemblage of wood-boring beetles, with most species using decaying wood as larval habitat. The great diversity is not surprising, as oak supports more beetle species than other trees, such as beech (*Fagus*) (Barbalat 1998). Tree species diversity will also increase species richness of wood-using insects, especially among those groups with specialized diets or habitat requirements. A diversity of dead wood is also important in supporting diverse saproxylic assemblages (McGeoch et al. 2007), in terms of wood species, as is a range of decay classes and microclimates.

Saproxylic insects and the ecological roles they play are recognized as important, but populations of these insects can decrease in response to disturbance. There may also be thresholds in the response to disturbance so that once the intensity or extent of disturbance crosses some level, the populations and the ecosystem function they play, decrease rapidly. These threshold

responses are unpredictable and poorly understood (Niwa et al. 2001). With the many roles that saproxylic insects play, such as pollination and nutrient cycling, attention must focus on the coarse woody debris that serves as habitat for this group. Sustainable forest management should strive to retain this habitat (Ulyshen et al. 2004). Many forest species, including saproxylic insects, have evolved to use coarse woody debris long before humans became interested in it (Hagan and Grove 1999). However, post-harvest management techniques may reduce coarse woody debris habitat greatly, some more than even fire (Tinker and Knight 2001).

Conservation of the diversity of agents that perform ecological roles and maintain forest health and function, including wood-boring beetles, is known to be important in ensuring sustainable use and enhancing resilience (Fischer et al. 2006). Fischer et al. (2006) have published 10 general guiding principles for such conservation in forestry and agricultural landscapes. The details of how the different taxa respond and thereby further alter forested systems remain to be filled in by long-term, large-scale experiments such as the HEE.

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APPENDIX 1.

Species of Cerambycidae caught 2006-08. Recent changes in taxonomy are indicated in parentheses and follow Bousquet (2007). Abundance and prevalence are the total number of individuals and the number of site-years (out of 108 possible) where the species was detected, respectively.

Species	Abundance	Prevalence
Aegomorphus modestus Gyllenhal	14	11
Analeptura lineola Say	85	29
Anelaphus parallelus Newman	2	2
Anelaphus pumilus Newman	1	1
Anelaphus villosus Fabricius	415	60
Astyleiopus variegatus Haldeman (=Sternidius)	1	1
Astylidius parvus LeConte	5	5
Astylopsis macula Say	23	17
Astylopsis sexguttata Say	1	1
Bellamira scalaris Say	7	6
Brachyleptura champlaini Casey	2	1
Brachyleptura rubrica Say	4	4
Clytus ruricola Olivier	77	41
Cyrtophorus verrucosus Olivier	69	39
Dorcaschema cinereum Olivier	3	3
Ecyrus d. dasycerus Say	4	4
Elaphidion mucronatum Say	49	30
Elytrimitatrix undata Fabricius	1	1
Enaphalodes atomarius Drury	1	1
Enaphalodes rufulus Haldeman	1	1
Encyclops caerulea Say	1	1
Euderces picipes Fabricius	8	8
Eupogonius pauper LeConte	12	11
Gaurotes cyanipennis Say	30	24
Glycobius speciosus Say	2	2
Goes pulverulentus Haldeman	1	1
Grammoptera haematites Newman	1	1
Graphisurus despectus LeConte (=Urographis)	24	18
Graphisurus fasciatus DeGeer (=Urographis)	45	25
Hesperophanes pubescens Haldeman	1	1
Heterachthes quadrimaculatus Haldeman	19	16
Hyperplatys aspersa Say	6	4
Hyperplatys maculata Haldeman	3	3
Leptostylus transversus Gyllenhal	20	17
Lepturges confluens Haldeman	13	12
Lepturges symmetricus Haldeman	1	1
Megacyllene caryae Gahan	4	4
Metacmaeops vittata Swederus	18	16
Micranoplium unicolor Haldeman	2	2
Microclytus gazellula Haldeman	1	1
Microgoes oculatus LeConte	18	15
Molorchus b. bimaculatus Say	68	20

(Appendix 1 continued on next page)

APPENDIX 1 (continued).

Species of Cerambycidae caught 2006-08. Recent changes in taxonomy are indicated in parentheses and follow Bousquet (2007). Abundance and prevalence are the total number of individuals and the number of site-years (out of 108 possible) where the species was detected, respectively.

Species	Abundance	Prevalence
Neandra brunnea Fabricius	25	22
Necydalis mellita Say	4	4
Neoclytus a. acuminatus Fabricius	91	52
Neoclytus m. mucronatus Fabricius	16	12
Neoclytus scutellaris Olivier	23	13
Oberea deficiens Casey	1	1
Oberea praelonga Casey	4	4
Oberea ruficollis Fabricius	1	1
Obrium maculatum Olivier	5	4
Obrium rufulum Gahan	1	1
Orthosoma brunneum Forster	68	46
Parelaphidion aspersum Haldeman	1	1
Parelaphidion incertum Newman	11	9
Phymatodes amoenus Say	22	14
Prionus laticollis Drury	16	14
Psenocerus supernotatus Say	14	9
Rhopalophora longipes Say	2	2
Saperda discoidea Fabricius	2	2
Saperda imitans Felt and Joutel	4	3
Saperda lateralis Fabricius	10	9
Saperda vestita Say	1	1
Sarosesthes fulminans Fabricius	4	4
Sphenostethus taslei Buquet	5	5
Stenelytrana emarginata Fabricius	5	5
Stenocorus cinnamopterus Randall	0	0
Stenocorus schaumii LeConte	1	1
Stenosphenus notatus Olivier	1	1
Sternidius alpha Say (=Liopinus)	3	3
Sternidius punctatus Haldeman (=Liopinus)	1	1
Strangalepta abbreviata Germar	30	13
Strangalia bicolor Swederus	2	2
Strangalia luteicornis Fabricius	49	31
Strophiona nitens Forster	4	4
Tessaropa tenuipes Haldeman	11	8
Trachysida mutabilis Newman	6	6
Trigonarthris proxima Say	1	1
Typocerus deceptus LeConte	12	8
Typocerus v. velutinus Olivier	238	59
Urgleptes facetus Say	1	1
Urgleptes querci Fitch	43	30
Urgleptes signatus LeConte	10	10
Xylotrechus aceris Fisher	1	1
Xylotrechus colonus Fabricius	245	83

APPENDIX 2.

Species of Buprestidae caught 2006-2008. Abundance and prevalence are the total number of individuals and the number of site-years (out of 108 possible) where the species was detected, respectively.

Species	Abundance	Prevalence
Actenodes acornis Say	1	1
Agrilus amelanchieri Knull	2	2
Agrilus bilineatus Weber	89	25
Agrilus bilineatus carpini Knull	1	1
Agrilus cephalicus LeConte	215	41
Agrilus lecontei Saunders	3	2
Agrilus masculinus Horn	2	2
Agrilus obsoletogutatus Gory	10	7
Agrilus olentangyi Champlain & Knull	1	1
Agrilus putillus Say	8	5
Anthaxia viridifrons Gory	2	2
Chrysobothris sexsignata Say	8	7
Dicerca asperata Laporte & Gory	1	1
Dicerca divaricata Say	32	23
Dicerca lepida LeConte	2	2
Dicerca lurida Fabricius	20	17
Ptosima gibbicollis Say	3	2

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.

THE LEPIDOPTERA AS PREDICTABLE COMMUNITIES OF HERBIVORES: A TEST OF NICHE ASSEMBLY USING THE MOTH COMMUNITIES OF MORGAN-MONROE STATE FOREST

Keith S. Summerville, Michael R. Saunders, and Jamie L. Lane[1]

Abstract.—The response of forest insect communities to disturbances such as timber harvest likely will depend on the underlying ecological assembly rules that affect community structure. Two competing hypotheses are niche assembly, which seeks to demonstrate significant species-environment correlations, and dispersal-assembly, which seeks to demonstrate spatial autocorrelation in the absence of species-environment correlations. Unfortunately, many studies of forest management never explicitly test what factors are responsible for maintaining community structure prior to harvest. The goal of this study is to examine variation in the community structure of forest Lepidoptera using the pre-timber harvest data on Lepidoptera from 18 forest sites within 3 management units of the Hardwood Ecosystem Experiment at Morgan-Monroe State Forest (MMSF) in Indiana.

We sampled 14,270 individuals representing 277 species of macrolepidoptera from MMSF in 2007. Canonical correspondence analysis suggested significant correlations between the moth assemblage found within a stand and levels of three environmental variables: importance of oaks, log-tree density, and percent basal area of shrubs and saplings. In contrast, Mantel tests suggested that forest moth communities at MMSF were, at most, weakly autocorrelated. The results here suggest some support of niche-assembly processes within forest macrolepidoptera. Still, a significant portion of the variation in species assemblages among forest stands remained unexplained, suggesting that stochastic factors and sampling bias may be important to consider when discussing patterns of lepidopteran diversity in space and time.

INTRODUCTION

The Lepidoptera is one of the most important insect orders in forest ecosystems, partly because of its constituent species' critical role in ecosystem functions such as trophic interactions and pollination and partly due to the irruptive potential of certain defoliators. Studies of bird communities in eastern deciduous forest systems regularly report the importance of lepidopteran larvae in the diet of birds (Holmes and Schultz 1988, McNulty et al. 2008, Robinson and Holmes 1982) and mammals (Burford et al. 1999). Species such as *Choristoneura rosaceana* (Har.), *Choristoneura fumifernana* (Clem.), *Malacosoma americanum* (F.), and *Lymantria dispar* (L.) can contribute to significant reductions in canopy cover, reduced tree growth increment, and in rare cases, direct mortality of tree species (Hennigar et al. 2007, Man et al. 2008, Tobin and Whitmire 2005). Therefore, understanding the key ecological processes that

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structure lepidopteran communities will be useful from the perspective of both forest conservation biology and forest pest management.

A small but growing body of literature suggests that both niche-based and dispersal-based assembly processes play a significant role in structuring forest lepidopteran assemblages. Niche-based assembly theory suggests that species diversity patterns are maintained by competitive resource partitioning, species sorting along environmental gradients, or some combination of both processes (Chesson 2000, Franklin et al. 2003, Lepš et al. 1998, Murakami et al. 2007). Species sorting is often a function of species traits, and can be a relatively predictable phenomenon (Summerville 2008).

In contrast, dispersal assembly is a neutral model that suggests that species membership within a community is more closely regulated by barriers to movement and founder effects associated with low frequency of colonization events (Leibold et al. 2004). Dispersal events are often modeled as stochastic processes related to interpatch distances rather than species traits per se. Under dispersal assembly, communities gain species from mass effects, where species that have established populations in favored habitat may colonize unsuitable habitat through dispersal dynamics (Leibold et al. 2004).

For Lepidoptera, tree species composition, herbaceous diversity, and stand volume all have been described as predictors of assemblage composition (reviewed in Summerville and Crist 2008). Dispersal limitation has proven difficult to disentangle from effects of habitat loss (which increases patch isolation but also diminishes habitat area) and biogeographic effects (Doak 2000, Kuussaari et al. 2007, Summerville et al. 2004). Donor pools for colonizing Lepidoptera are also poorly known (Holl 1996). Community studies have often focused on characterizing local assemblage structure and inferred landscape composition as an extension of local stand dynamics (Summerville and Crist 2008, but see Franklin et al. 2003).

Species sorting and dispersal assembly are processes that represent contrasting, but not mutually exclusive, answers to the basic question – why is this set of species found in this particular habitat? Testing for the relative importance of each process is critical for understanding how forest lepidopteran assemblages recover from disturbances. Studies of timber harvest have established that forest moth communities appear slow to recover from clearcutting, even when managed patches are small relative to the total stand area (Forkner et al. 2006, Holl 1996, Summerville and Crist 2004). Questions that remain to be answered include: (i) Did moth communities fail to recover an original lepidopteran species composition because of shifts in floristic composition? or (ii) Did moth communities fail to recover an original lepidopteran species composition because timber management created new barriers to recolonization? Before either of these questions can be answered, however, a test of the suitability of niche-based and dispersal-based assembly theory should be made pre-harvest. That is, prior to disturbance, what appears to be the proper hypothetical model explaining moth community structure? Does one process tend to explain the bulk of the variance in community structure or are both equally compelling?

The goal of this paper is to ascertain how moth community structure is maintained in a relatively undisturbed forest landscape. Specifically, we use data from macrolepidoptera sampled from 20 forest stands within Morgan-Monroe State Forest (MMSF) in Indiana to determine the degree to which niche-based or dispersal-based assembly mechanisms were significant in predicting moth species diversity and composition. Because many Lepidoptera are obligately connected to a narrow suite of host plants for larval development, we predicted that niche-based assembly processes would be of much greater importance than dispersal-based assembly processes. Further, we hypothesized that two critical floristic variables would determine moth species composition within forest stands: 1) the importance of oaks (*Quercus* spp.) because a large number of species are specialized on

this host plant (as opposed to maples [*Acer* spp.], for example), and 2) the density of the stand (because high tree density within a stand indicates prior disturbance and, perhaps, lower foliage quality among trees (see Fry et al. 2009, Summerville and Crist 2002). Moth diversity was expected to be higher in stands with a dominant oak component and a low stand density. If dispersal-assembly is more important in predicting moth species composition, then we expected to see stronger evidence of spatial autocorrelation in moth assemblages than observed in the stand tree assemblage (see Nekola and White 1999). Thus, moth species would be structured at a finer grain across the forest landscape than the food resources upon which their larvae are known to depend.

MATERIALS AND METHODS

Study Site

MMSF occurs within the north-central interior dry-mesic oak forest and woodland ecological system (39°31′28″N, 86°44′13″W). This system is widespread across the glaciated regions of the Midwest, from North Dakota to Ohio and Minnesota to Missouri. Soils are typically well-drained mollisols or alfisols from loam to sandy loam (Braun 2001). Located in the Central U.S. Mixed Hardwood Forest ecoregions, MMSF has dry slopes dominated by oak-hickory (*Carya*) forest. In unfragmented landscapes, canopy cover tends to be dense, although historic fire regimes likely maintained open canopy (Homoya et al. 1985). Chestnut oak (*Q. prinus*) is a dominant species with white oak (*Q. alba*), red oak (*Q. rubra*), and black oak (*Q. velutina*) present depending on soil moisture regimes. Bitternut hickory (*C. cordiformis*) and shagbark hickory (*C. ovata*) are also common. Additionally, this ecoregion is among North America's richest for herbaceous plants and shrubs, with >2,000 species described (Homoya et al. 1985). Most of the sites that were used for sampling have remained unlogged for at least 50 years. Salvage logging has occurred on a localized basis after major windstorm events, but these areas were avoided for the purpose of this study. Several forest stands, however, had not been cut after the mid-1920s.

Lepidoptera Sampling Methodology

In early 2007, we identified three management units within the Hardwood Ecosystem Experiment (HEE) at MMSF for lepidopteran sampling. Management units were ~500 ha in size, were 1-3 km apart, and were scheduled subsequent to our sampling to have forest harvest treatments applied to an inner core area that ranged from 78 to 110 ha in size (Kalb and Mycroft, this publication). Tracts were assigned to a specific timber management regime a priori, although the harvest schedule was crafted such that timber would be removed only after establishing a baseline biodiversity assessment. Management unit 1 was assigned an uneven-aged management regime using a combination of patch cuts and single-tree selection, unit 2 was dedicated as an unharvested reserve to serve as an experimental control, and unit 3 was assigned an even-aged management regime using three-stage shelterwoods and clearcut harvesting.

To develop a baseline assessment for forest moth biodiversity, we identified eight stands within each of the two managed units and four stands in the control unit (n = 20 total sites). Stands were separated by >150 m (range 150-750 m) and were grouped such that forest stands within each management unit were clustered into stand pairs. Stand pairs were generally 150 m apart and were chosen so that one occurred within the center of an area to be harvested and the other within the unharvested matrix adjacent to a harvest boundary (see Kalb and Mycroft, this publication).

Moths were collected from forest stands using Universal 12-watt blacklight traps (BioQuip Products, Inc., Rancho Dominguez, CA) powered by 12-volt, 26 ampere hr^{-1} batteries. On nights of operation, a single trap was placed at each site on a platform 2 m above the ground and remained lit from 2000 to 0700 CDT. We sampled Lepidoptera approximately every

14 days from 30 May to 30 August 2007, producing 100 total samples from the 20 sites (five per site). This level of sampling correlated with a sampling efficiency of 85 percent, which is equal to most other studies of Lepidoptera in temperate deciduous forests (Summerville and Crist 2008).

Weather and moon intensity are known to affect sampling efficiency of blacklight traps, so trapping was restricted to nights that had a minimum temperature of 16 °C, no precipitation, and low levels of ambient moonlight (half to new moon phases) (cf. Yela and Holyoak 1997). Lepidopteran nomenclature and authorities follow Hodges et al. (1983) with revisions as noted in Ferguson (2008) and LaFontaine and Schmidt (2010). Voucher specimens were deposited at Drake University, Des Moines, IA.

Forest Vegetation Sampling Methodology

From September 2007 to August 2008, and before all harvesting, forest vegetation was sampled across each management unit using a systematic grid on a 75m × 150m spacing. All trees and standing snags ≥11.5 cm diameter at breast height (d.b.h.; = 1.37 m above ground) were inventoried within circular 0.10-ha plots on each grid point; saplings, shrubs, and standing snags 1.5-11.4 cm d.b.h. were measured within a nested 0.025-ha subplot.

Across the three units, 163 plots were established. For each tree, sapling, or shrub, species, d.b.h., condition, light field (sensu Bechtold 2003), stratum, and spatial location were recorded. Height and lowest live crown were recorded for a subsample of trees and saplings. For snags, species (if known), d.b.h., height, decay class (sensu Cline et al. 1980), and spatial location were recorded. More information regarding this inventory can be found in Saunders and Arseneault (this publication). This study included only those vegetation plots within 100 m of the moth sampling sites. Therefore, between two and five plots per moth sampling site were used for analysis. Data for 2 of the timber stands were not obtained (both from management unit 3), so 18 stands among the 3 management units were used in subsequent analyses.

Data Analyses

We tested whether communities of forest macrolepidoptera were structured by niche-based assembly rules using canonical correspondence analysis (CCA). This technique is a multivariate ordination approach that determines whether environmental variation is important in creating structure in a species-abundance matrix (ter Braak 1986). A detailed statistical treatment of CCA can be found in McCune and Grace (2006). To summarize, the algorithm for CCA defines linear combinations of orthogonal environmental variables that maximize the separation distance among species in ordination space. Site scores are derived as weighted averages of species scores and are plotted in 2- or 3-dimensional space. Thus, species with scores that are close to a given site point on a CCA ordination figure are likely to attain a high abundance at that particular site. The scatter of site scores in ordination space depicts how overall regional species composition varies across the measured environmental gradients. The influence of environmental variables on species abundances can be qualitatively modeled on the ordination figure through the use of joint bi-plots, which are graphic renderings of species sorting through niche dynamics (Jongman et al. 1995).

We performed CCA using PC-ORD for Windows (version 5.0, MjM Software Design 2006). We pooled species-abundance data from the 5 sampling nights for each stand, producing one, season-long characterization of lepidopteran community structure, as recommended by Summerville and Crist (2002). Microlepidopterans in families such as Gelechiidae, Tortricidae, and Gracillariidae were excluded due to the difficulty in validating species-level determinations.

Moth community data were then filtered in two ways to test for niche assembly. First, we ran a CCA using the entire species pool from each site minus species sampled as either singletons or doubletons (a total of 1 or 2 individuals sampled across all sites). Other authors have argued convincingly that exceptionally rare species can mask niche assembly dynamics because low abundance may be a consequence of sampling bias, species phenology, or weak photo-attraction (Hamback et al. 2007, Summerville and Crist 2008). Second, because the forest stands within MMSF were historically oak-dominated, we tested for niche assembly using only moth species known to feed on oaks as larvae (see Robinson et al. 2002). These two CCAs allowed us to differentiate between niche assembly at the level of the entire community and niche assembly within what should be the dominant guild of moths across MMSF.

We used the same secondary matrix of environmental variables for both CCAs. Initially, we included the following variables in the environmental matrix: total stand basal area (m^2 ha^{-1}, calculated using only those overstory sampling plots within 100 m of each moth sample point [n=2-5]), average tree density (# ha^{-1}), importance of oak, importance of maple, importance of sassafras (*Sassafras albidum*), importance of ash (*Fraxinus* spp.), elevation (m), aspect (°), and average percentage basal area of shrubs and saplings. Importance values were calculated as the arithmetic mean of basal area, density, and frequency values across all overstory plots adjacent to a moth sampling point. For analysis, shrubs and saplings were defined as individuals with stem d.b.h. ≤10.0 cm. Sassafras importance was included as an environmental variable because its occurrence is an indicator of past disturbance (Heitzman et al. 2007, Welch et al. 2000). Ash importance was included because its long-term viability within Indiana forests is compromised by range expansion of the emerald ash borer (*Agrilus planipennis*). Basal area and density measurements were log-transformed prior to CCA to normalize distributions (Jongman et al. 1995).

Because CCA requires that all environmental variables in the final model be orthogonal, we removed correlated variables through a process of pre-screening with Spearman Rank-Correlation Analysis (McCune and Grace 2006). Five environmental variables were selected for subsequent analyses: oak importance, sassafras importance, average percentage shrub/sapling basal area, total stand basal area, and average tree density. To assess the significance of CCA ordination axes, we used the Monte Carlo simulation option within PC-ORD. This step allowed us to test the specific null hypothesis that there was no correlation between the moth species-abundance matrix and the environmental variable-site matrix (McCune and Grace 2006). We selected a conservative r^2 value (0.20) for the minimum level of species-environment correlation for bi-plot display on the CCA ordination.

Finally, we performed Mantel tests to assess the degree of spatial autocorrelation among moth communities within and among management units. As with CCA, two separate Mantel tests were generated, using all species minus singletons and doubletons and using only oak-feeding species. The overall species-abundance matrix was used to calculate community similarity among forest sites. The distance matrix consisted of Euclidean distances among each pair of sites as measured in kilometers. Finally, a third Mantel test was performed to assess whether the overstory tree community displayed significant spatial structure. As with CCA, Mantel tests were generated using PC-ORD.

RESULTS

A total of 14,270 macrolepidopteran individuals representing 277 species were sampled from MMSF in 2007 (Table 1, Appendix 1). Of these, 213 species contained total abundances >2 (i.e., were not singletons or doubletons) and 94 species were determined to be capable of feeding on oak. Most of these oak-feeding species would fit the description of host plant "generalist," however, as host plant breadth

Table 1.—Number of macrolepidopteran species and individuals of forest Lepidoptera sampled from Morgan-Monroe State Forest, 2007.

Family	Number of species	Number of individuals
Apatelodidae	2	56
Cossidae	1	6
Drepanidae	1	37
Erebidae	67	5,138
Geometridae	72	2,008
Lasiocampidae	3	172
Limacodidae	9	72
Megalopygidae	2	114
Mimalonidae	1	1
Noctuidae	75	3,117
Nolidae	4	29
Notodontidae	28	2,724
Saturniidae	12	685
Sphingidae	6	103
Yponmeutidae	1	8
Total	277	14,270

for all but 39 oak-feeding moths exceeded two plant families. Regardless, nearly 34 percent of all of the species we sampled could be described as oak feeding, making this the largest guild of species sampled in 2007. In addition, a few species that tend not to be sampled in high abundance in oak-hickory forests of Indiana and Ohio were found, including *Zanclognatha jaccusalis* (Erebidae), *Lacosoma chiridota* (Mimalonidae), and *Anistoa virginiensis* (Saturniidae). Overall, however, most of the species sampled from MMSF are reasonably well represented in collections or have been previously reported in the literature from across Indiana, Ohio, and Kentucky.

Both of our CCAs revealed significant evidence of niche assembly in the Lepidoptera of MMSF. First, the analysis of species-environment correlations for all moth species (excluding singletons and doubletons) suggested two significant ordination axes and explained approximately 25 percent of the variance in community structure (Table 2). Sites appeared to weakly separate from one another according to management unit (Fig. 1), but variation in moth species composition within a given management unit was roughly equal to variation among management units. Three environmental variables had an r^2 >0.20: importance of oak, percentage basal area of shrubs and saplings, and log-tree density (Fig. 1).

The importance of oak loaded most significantly onto ordination axis 1 and separated a few stands in management unit 1 (which has oak importance >50 percent) from the remainder of the forest areas in MMSF (Table 3). Species associated with high values of oak importance included *Catocala micronympha* (Erebidae), *Dasylophia thyatiroides* (Notodontidae), *Crambidia pallida* (Erebidae), *Cisthene plumbea* (Erebidae), *Cosmia calami* (Noctuidae), *Besma quercivoraria* (Geometridae), and *Catocala retecta* (Erebidae). Several sites in management unit 3 had very high overstory density and thus contained moth communities different from those in units 1 or 2

Table 2.—Summary of canonical correspondence analyses (CCAs) relating moth communities to stand-level environmental variation. Two analyses were performed: (i) using all moth species sampled except singletons and doubletons and (ii) using only oak-feeding species.

Moth community	Ordination axis	Eigenvalue	Explained variance (percent)	Cumulative variance (percent)	P-value
All species (total abundance >2)	Axis 1	0.085	10.4	10.4	0.025
	Axis 2	0.074	8.8	19.2	0.044
	Axis 3	0.048	5.7	24.9	0.270
Oak-feeding species	Axis 1	0.097	10.8	10.8	0.030
	Axis 2	0.052	6.0	16.8	0.050
	Axis 3	0.027	5.0	21.8	0.340

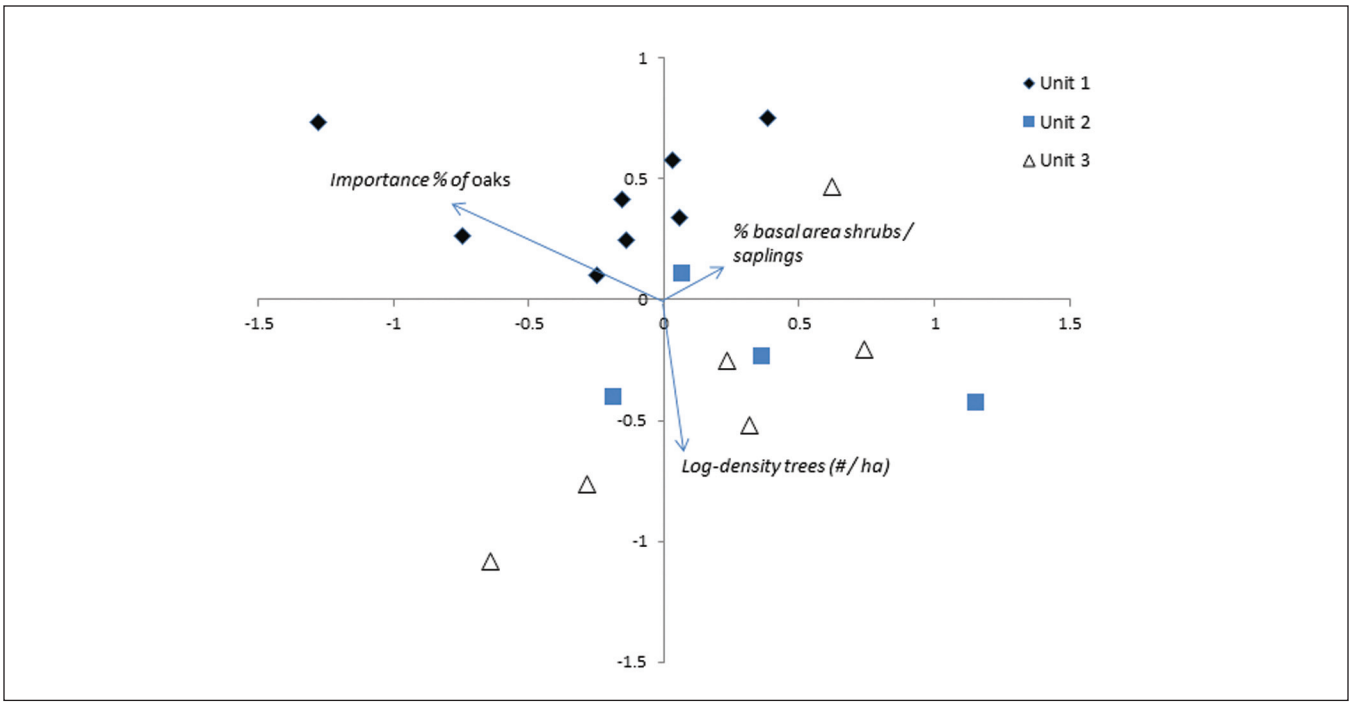


Figure 1.—Canonical correspondence analysis of 18 forest stands in MMSF and 213 moth species with total sampled abundances >2. Three environmental variables had significant r^2 values (≥0.20) with the site scores from ordination axes 1 and 2. These are depicted as joint bi-plots in ordination space.

Table 3.—Canonical coefficients derived from CCAs of all moth species with total abundance >2 sampled from Morgan-Monroe State Forest, 2007. Only the first two ordination axes were significant.

Environmental variable	Axis 1 score	Axis 2 score	Axis 3 score
% importance of oak	-0.513	0.189	0.133
% importance of sassafras	-0.002	0.151	-0.374
% basal area of shrubs and saplings	0.012	0.259	0.270
Log-tree density (# ha^{-1})	-0.258	-0.431	-0.001
Log-stand basal area (m^2 ha^{-1})	-0.270	0.040	-0.299

(Fig. 1). Species with high site scores for areas with large values of log-transformed tree density included potential management concerns such as *Malacosoma americanum* (Lasiocampidae), *Hyphantria cunea* (Erebidae), and *Datana integerrima* (Notodontidae). Sites with a high importance of oaks also tended to have a low percentage basal area of shrubs, although the partial correlation coefficient was marginal at 0.39 (Fig. 1). Several species of moths were highly correlated with sites that possessed a large value for percentage basal area in shrubs/saplings: *Digrammia ocellinata* (Geometridae), *Chlorochlamys chloroleucaria* (Geometridae), and *Hypercompe scribonia* (Erebidae). Importance of *Sassafras* and *Fraxinus* were not significant predictors of moth species composition (Table 3).

When only oak-feeding moth species were included in the analysis, our CCA explained slightly less of the total variance in species composition (21.8 percent; see Table 2). Both ordination axes 1 and 2 were significant, and the same three environmental variables were significant predictors of moth species composition (Table 4). The same species from the prior analysis attained large abundance in sites with high importance of oak, with the notable addition of most species in genus *Acronitca* (Noctuidae). The relative importance of log-transformed tree density and percentage basal area of shrubs and saplings was reversed, however (Fig. 2). Shrub and sapling basal area seemed to differentiate a couple of stands in unit 3 and one stand in unit 1 from the others, while tree density was associated with similar moth assemblages in units 2 and 3. Oak-feeding species associated with sites possessing high trunk density included *Orgyia definita* (Erebidae), *Hyphantria cunea* (Erebidae), *Natada nasoni* (Limacodidae), and *Charadra deridens* (Noctuidae). Moths preferring stands with large values of shrub and sapling basal area included host plant generalists such as species in the genera *Baileya* (Nolidae), *Hyperstrotia* (Erebidae), and *Eupithecia* (Geometridae).

Table 4.—Canonical coefficients derived from CCAs of oak-feeding moth species sampled from Morgan-Monroe State Forest, 2007. Only the first two ordination axes were significant.

Environmental variable	Axis 1 score	Axis 2 score	Axis 3 score
% importance of oak	-0.494	-0.210	0.153
% importance of sassafras	-0.034	0.059	0.006
% basal area of shrubs and saplings	0.047	-0.361	0.294
Log-tree density (# ha^{-1})	-0.243	0.273	0.315
Log-stand basal area (m^2 ha^{-1})	-0.192	0.090	0.073

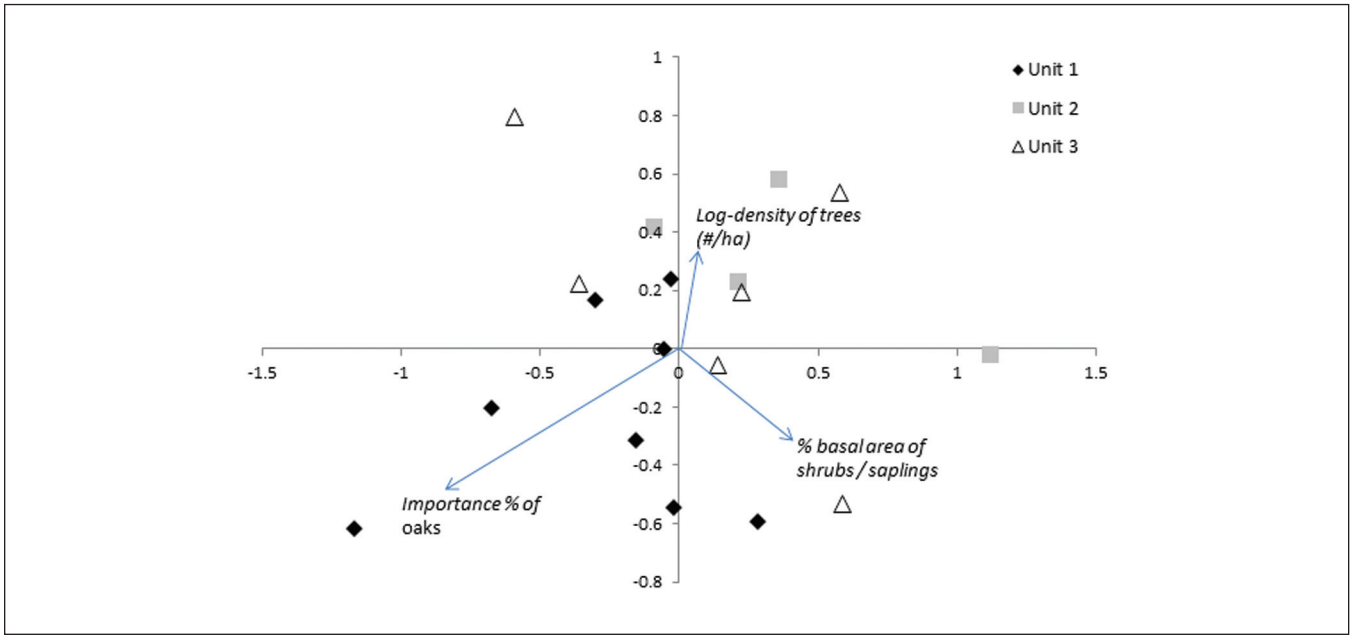


Figure 2.—Canonical correspondence analysis of 18 forest stands in MMSF and 94 oak-feeding moth species. Three environmental variables had significant r^2 values (≥0.20) with the site scores from ordination axes 1 and 2. These are depicted as joint bi-plots in ordination space.

The Mantel tests failed to detect any significant relationships between similarity in lepidopteran community composition and the linear distances between forest stands. For all moths, the relationship suggested weak spatial autocorrelation ($Z = 360618$, $r = 0.133$, $P = 0.076$). When the data were restricted to oak-feeders, no spatial autocorrelation was detected ($Z = 318712$, $r = 0.098$, $P = 0.15$). Similarly, there was no evidence of spatial dependence in the tree data from plots adjacent to moth sampling points ($Z = 327654$, $r = 0.087$, $P = 0.21$). Thus, spatial dependence in moth assemblages appeared less important to community structure than species sorting effects from environmental variables.

DISCUSSION

We found clear evidence that niche assembly processes are important determinants of community structure in forest macrolepidoptera. In contrast to expectation, however, the measured environmental variables explained only approximately 25 percent of the variation in species composition among the 18 forest stands. This result is very close to the variance explained using CCA to tease apart similarity in moth assemblages in the Appalachian foothills (Summerville and Crist 2002) and the Missouri oak woodlands (Forkner et al. 2006). Studies at broader spatial scales appear to have even less explanatory power (e.g., Hammond and Miller 1998). This observation prompts the question: Are niche assembly processes really minor, have studies (including this one) measured the wrong environmental variables, or are moth communities less deterministic than we assume?

Although not conducted in eastern deciduous forests of the United States, a two-decade study by Lepš et al. (1998) offers some insight into why direct effects of environmental variables on moth communities appear small. Lepš et al. demonstrated that moth communities change predictably along a seral gradient, but that turnover in species membership within a community is much more deterministic for dietary specialists. Generalist moths (i.e., those whose caterpillars feed on ≥ two plant families) have much more stochastic population fluctuations and are less predictable members of an assemblage (see Butler and Straznac 2000, Spitzer et al. 1984). Specialist moths, especially those that feed on oak, tend to form predictable assemblages in space and time (Fukumoto and Kajimara 2011, Turčáni et al. 2010). Our data from MMSF suggest that >75 percent of the species we sampled were generalists. Even within the species known to feed on oaks, 60 percent were considered generalists. Because generalist moths have larger inter-annual population fluctuations, their occurrence within a community is a product more of demographic and environmental stochasticity than of occurrence of suitable host plants (Murphy and Lill 2010, Summerville and Crist 2008).

Movement dynamics may also play a significant role in structuring moth assemblages in space, especially for microlepidoptera. In some species, dispersal assembly is a product of adult movements and mass effects (Summerville and Crist 2008), but in others it is a consequence of larval ballooning and neighborhood effects (Turčáni et al. 2010). We found little evidence of either type of dispersal assembly process for these species of macrolepidoptera.

Disturbance and environmental variables associated with disturbance are also critical determinants of moth community structure (e.g., Broome et al. 2011), and we attempted to incorporate indirect assessments of past disturbance by building Sassafras importance and tree density into our CCA. Disturbance also leaves a lasting impact on the forest understory, and some studies have suggested that forest understory vegetation is as important as overstory stand composition in generating patterns of macrolepidopteran biodiversity (Summerville and Crist 2008). Interestingly, we failed to sample many species that are restricted to woodland forbs or graminoids across MMSF, so it is possible that if these species used to be part of the forest community, they have already been lost (Holl 1996).

Placed within the context of current metacommunity theory, moth assemblages do appear to be far less a consequence of niche assembly processes than a function of a large number of independent, and as yet unexplained, effects. Of course, this conclusion should be placed within the context of an important caveat: the predictability of a forest moth assemblage will generally be low, except when disturbances significantly alter overstory composition. Whether as part of natural gap dynamics or due to timber extraction, forest disturbance alters forest microclimate, facilitates shifts in stand composition, and allows moth species the opportunity to colonize new habitats (Summerville et al. 2009).

If a previously oak-dominated forest is logged and recovers as mixed beech (*Fagus*)-maple, then a new assemblage lacking oak specialists will likely replace it. Because between 50 and 60 percent of oak-feeding macrolepidoptera have been documented to feed upon maple or beech, however, the mass extinction of lepidopterans may not occur. Of course, documenting that moth larvae will feed on foliage is not equivalent to demonstrating that the foliage is ultimately suitable for sustaining viable populations (see Matsuki et al. 2011). Landscape-level loss of oak from forest stands can homogenize forest moth assemblages even when species composition within individual tree crowns is the result of random ovipositions by generalist feeding species (Broome et al. 2011, Holl 1996, Summerville and Crist 2008).

Forest managers can take two main points away from our analyses here: (1) addition or subtraction of tree species from a stand will tend to add and subtract specialist macromoth species and (2) substantial losses in moth species from the regional metacommunity can largely be avoided by focusing on maintaining stand heterogeneity at the landscape level. The former point should empower the forest manager to predict how loss of tree species will impact moth species with narrow niches, and the latter suggests that stand management through timber harvesting can proceed as long as care is given to the broader forest landscape.

Ongoing recruitment challenges of oak and the continued replacement of oak and hickory with maple and beech suggest that the Central Hardwood Forest faces an extinction debt of oak-feeding lepidopterans (sensu Sang et al. 2010). The looming feedback loop between climate change on a global scale and increased irruptions of defoliating herbivores suggests that broad-scale stand homogeneity is creating a template for extreme outbreak events of some pest Lepidoptera (Clark et al. 2010, Dymond et al. 2010, Hennigar and MacLean 2010). Should crown defoliation exacerbate shifts in tree species composition away from the current pattern, loss of some lepidopteran species from all eastern deciduous forests might become troublingly easy to predict.

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APPENDIX 1.

Lepidoptera species sampled from MMSF in 2007. Species are arranged by taxonomic family. Nomenclature follows Hodges et al. (1983) with revisions to the Geometroidea after Ferguson (2008) and to the Noctuoidea following LaFontaine and Schmidt (2010).

Species	Author	Family	Species	Author	Family
Apatelodes torrefacta	J.E. Smith	Apatelodidae	*Panopoda carneicosta*	Gn.	Erebidae
Ocleclostera angelica	Grt.	Apatelodidae	*Panopoda rufimargo*	Hbn.	Erebidae
Prionoxystus robiniae	Peck	Cossidae	*Parallelia bistriaris*	Hbn.	Erebidae
Oreta rosea	Wlk.	Drepanidae	*Phalaenostola larentioides*	Grt.	Erebidae
Allotria elonympha	Hbn.	Erebidae	*Plusiodonta compressipalps*	Gn.	Erebidae
Apantesis vittata	F.	Erebidae	*Pyrrharctia isabella*	J.E. Smith	Erebidae
Bleptina caradrinalis	Gn.	Erebidae	*Renia discoloralis*	Gn.	Erebidae
Caenurgina erechtea	Cram.	Erebidae	*Renia flavipunctalis*	Gey.	Erebidae
Catocala amica	Hbn.	Erebidae	*Renia sobrialis*	Wlk.	Erebidae
Catocala ilia	Cram.	Erebidae	*Scolecocapma liburna*	Gey.	Erebidae
Catocala innubens	Gn.	Erebidae	*Spilosoma latipennis*	Stretch	Erebidae
Catocala micronympha	Gn.	Erebidae	*Spilosoma virginica*	F.	Erebidae
Catocala nebulosa	Edw.	Erebidae	*Virbia opella*	Grt.	Erebidae
Catocala obscura	Stkr.	Erebidae	*Zale lunata*	Drury	Erebidae
Catocala palaeogama	Gn.	Erebidae	*Zale minerea*	Gn.	Erebidae
Catocala residua	Grt.	Erebidae	*Zanclognatha cruralis*	Gn.	Erebidae
Catocala retecta	Grt.	Erebidae	*Zanclognatha jacchusalis*	Wlk.	Erebidae
Catocala ultronia	Hbn.	Erebidae	*Zanclognatha laevigata*	Grt.	Erebidae
Catocala vidua	J.E. Smith	Erebidae	*Zanglognatha obscuripennis*	Grt.	Erebidae
Cisseps fulvicollis	Hbn.	Erebidae	*Anacamptodes defectaria*	Gn.	Geometridae
Cisthene packardii	Grt.	Erebidae	*Anacamptodes ephyraria*	Wlk.	Geometridae
Cisthene plumbea	Stretch	Erebidae	*Anacamptodes humaria*	Gn.	Geometridae
Clemensia albata	Pack	Erebidae	*Anavitrinelia pampinaria*	Gn.	Geometridae
Crambidia lithosioides	Dyar	Erebidae	*Besma quercivoraria*	Gn.	Geometridae
Crambidia pallida	Pack	Erebidae	*Biston betularia*	L.	Geometridae
Cycnia inopinatus	H. Edw.	Erebidae	*Callizzia amorata*	Pack.	Geometridae
Cycnia tenera	Hbn.	Erebidae	*Campaea perlata*	Gn.	Geometridae
Dasychira basiflava	Pack.	Erebidae	*Cepphis armataria*	H. & S.	Geometridae
Dasychira vagans	B. & McD.	Erebidae	*Cyclophora pendulinaria*	Gn.	Geometridae
Euchaetes egle	Drury	Erebidae	*Digrammia ocellinata*	Gn.	Geometridae
Grammia figurata	Drury	Erebidae	*Ectropis crepuscularia*	D. & S.	Geometridae
Grammia virgo	L.	Erebidae	*Ennomos subsignaria*	Hbn.	Geometridae
Halysidota tessellaris	J.E. Smith	Erebidae	*Epimecis hortaria*	F.	Geometridae
Haploa clymene	Brown	Erebidae	*Erastria coloraria*	F.	Geometridae
Haploa lecontei	Guér.-Méneville	Erebidae	*Eubaphe mendica*	Wlk.	Geometridae
Haploa reversa	Stretch	Erebidae	*Euchlaena amoenaria*	Gn.	Geometridae
Hypercompe scribonia	Stoll	Erebidae	*Euchlaena irraria*	B. & McD.	Geometridae
Hyperstrotia pervertens	B. & McD.	Erebidae	*Euchlaena obtusaria*	Hbn.	Geometridae
Hyphantria cunea	Drury	Erebidae	*Euchlaena pectinaria*	D. & S.	Geometridae
Hypoprepia fucosa	Hbn.	Erebidae	*Euchlaena tigrinaria*	Gn.	Geometridae
Idia aemula	Hbn.	Erebidae	*Eulithis diversilineata*	Hbn.	Geometridae
Idia americalis	Gn.	Erebidae	*Eupithecia miserulata*	Grt.	Geometridae
Idia lubricalis	Gey.	Erebidae	*Eutrapela clemataria*	J.E. Smith	Geometridae
Idia rotundalis	Wlk.	Erebidae	*Haematopis grataria*	F.	Geometridae
Lophocampa caryae	Hodges	Erebidae	*Heterophleps triguttaria*	H.-S	Geometridae
Lymantria dispar	L.	Erebidae	*Hydrelia inornata*	Hulst	Geometridae
Macrochilo absorptalis	Wlk.	Erebidae	*Hypagyrtis unipunctata*	Haw.	Geometridae
Metalectra richardsi	Hbn.	Erebidae	*Iridopsis larvaria*	Gn.	Geometridae
Orgyia definita	Pack.	Erebidae	*Itame evagaria*	Hulst	Geometridae
Orgyia leucostigma	J.E. Smith	Erebidae	*Itame pustularia*	Hulst	Geometridae
Palthis asopialis	Gn.	Erebidae	*Itame subcessaria*	Hulst	Geometridae

(Appendix 1 continued on next page)

APPENDIX 1 (continued).

Lepidoptera species sampled from MMSF in 2007. Species are arranged by taxonomic family. Nomenclature follows Hodges et al. (1983) with revisions to the Geometroidea after Ferguson (2008) and to the Noctuoidea following LaFontaine and Schmidt (2010).

Species	Author	Family	Species	Author	Family
Lambdina pellucidaria	G. & R.	Geometridae	*Acronicta americana*	Harr.	Noctuidae
Lomographa vestaliata	Gn.	Geometridae	*Acronicta exilis*	Grt.	Noctuidae
Lytrosis unitaria	H.-S.	Geometridae	*Acronicta fragilis*	Gn.	Noctuidae
Macaria aemulataria	Wlk.	Geometridae	*Acronicta grisea*	Wlk.	Noctuidae
Macaria bisignata	Wlk.	Geometridae	*Acronicta haesitata*	Grt.	Noctuidae
Macaria continuata	Wlk.	Geometridae	*Acronicta hamemelis*	Grt.	Noctuidae
Macaria promiscuata	Fgn.	Geometridae	*Acronicta hasta*	Gn.	Noctuidae
Macariamultilineata	Pack.	Geometridae	*Acronicta impleta*	Wlk.	Noctuidae
Melanolophia canadaria	Gn.	Geometridae	*Acronicta inclara*	Sm.	Noctuidae
Mellilla xanthometata	Wlk.	Geometridae	*Acronicta interrupta*	Gn.	Noctuidae
Metarranthis hypocharia	H.-S.	Geometridae	*Acronicta laetifica*	Sm.	Noctuidae
Nacophora quernaria	J.E. Smith	Geometridae	*Acronicta lithospila*	Grt.	Noctuidae
Nemoria bistriaria	Hbn.	Geometridae	*Acronicta modica*	Wlk.	Noctuidae
Orthonama centrostrigaria	Woll.	Geometridae	*Acronicta morula*	G. & R.	Noctuidae
Orthonama obstipata	F.	Geometridae	*Acronicta oblinita*	J. E. Smith	Noctuidae
Pero honestaria	Wlk.	Geometridae	*Acronicta ovata*	Grt.	Noctuidae
Plagodis alcoolaria	Gn.	Geometridae	*Acronicta retardata*	Wlk.	Noctuidae
Plagodis fervidaria	H.-S.	Geometridae	*Acronicta spinigera*	Gn.	Noctuidae
Plagodis phlogosaria	Gn.	Geometridae	*Acronicta tristis*	Sm.	Noctuidae
Plagodis serinaria	H.-S.	Geometridae	*Agrotis ipsilon*	Hufn.	Noctuidae
Probole amicaria	H.-S.	Geometridae	*Agrotis venerabilis*	Wlk.	Noctuidae
Probole nyssaria	Gn.	Geometridae	*Anagrapha falcifera*	Kby.	Noctuidae
Prochoerodes transversata	Drury	Geometridae	*Athetis tarda*	Gn.	Noctuidae
Protoboarmia porcelaria	Gn.	Geometridae	*Autographa precationis*	Gn.	Noctuidae
Scopula inductata	Gn.	Geometridae	*Balsa malana*	Fitch	Noctuidae
Scopula limboundata	Haw.	Geometridae	*Callopistria mollissima*	Gn.	Noctuidae
Sicya macularia	Harr.	Geometridae	*Cerma cerintha*	Tr.	Noctuidae
Synchlora aerate	F.	Geometridae	*Charadra deridens*	Gn.	Noctuidae
Tetracis crocallata	Gn.	Geometridae	*Chytonix palliatricula*	Gn.	Noctuidae
Trigrammia quadrinotaria	H.-S.	Geometridae	*Colocasia flavicornis*	Sm.	Noctuidae
Triphosa haesitata	Gn.	Geometridae	*Condica sutor*	Gn.	Noctuidae
Xanthorhoe ferrugata	Cl.	Geometridae	*Condica vecors*	Gn.	Noctuidae
Xanthorhoe lacustrata	Gn.	Geometridae	*Elaphria grata*	Hbn.	Noctuidae
Heteropacha rileyana	Harv.	Lasiocampidae	*Elaphria versicolor*	Grt.	Noctuidae
Malacosoma americanum	F.	Lasiocampidae	*Eudryas grata*	F.	Noctuidae
Malacosoma disstria	Hbn.	Lasiocampidae	*Euplexia benesimilis*	McD.	Noctuidae
Isa textula	H.-S.	Limacodidae	*Hypena scabra*	F.	Noctuidae
Lithacodes fasciola	H.-S.	Limacodidae	*Lacinipolia renigera*	Steph.	Noctuidae
Natada nasoni	Grt.	Limacodidae	*Marimatha nigrofimbria*	Gn.	Noctuidae
Packardia geminata	Pack.	Limacodidae	*Melanchra adjuncta*	Gn.	Noctuidae
Parasa chloris	H.-S.	Limacodidae	*Morrisonia latex*	Gn.	Noctuidae
Phyllodesma americana	Harris	Limacodidae	*Mythimna unipuncta*	Haw.	Noctuidae
Prolimacodes badia	Hbn.	Limacodidae	*Noctua pronuba*	L.	
Tortricidia flexusoa	Grt.	Limacodidae	*Noctuidae*		
Tortricidia testacea	Pack.	Limacodidae	*Ochropleura implecta*	LaFontaine	Noctuidae
Lagoa crispate	Pack.	Megalopygidae	*Ogdoconta cinereola*	Gn.	Noctuidae
Norape ovina	Sepp	Megalopygidae	*Orthodes majuscula*	Butler	Noctuidae
Lacosoma chiridota	Grt.	Mimallonidae	*Peridroma saucia*	Hbn.	Noctuidae
“Orthodes” detracta	Wlk.	Noctuidae	*Phosphila miselioides*	Hbn.	Noctuidae
Abagrotis alternata	Grote	Noctuidae	*Polygrammate hebraeicum*	Hbn.	Noctuidae
Acronicta afflicta	Grt.	Noctuidae	*Protodeltote muscosula*	Gn.	Noctuidae

(Appendix 1 continued on next page)

APPENDIX 1 (continued).

Lepidoptera species sampled from MMSF in 2007. Species are arranged by taxonomic family. Nomenclature follows Hodges et al. (1983) with revisions to the Geometroidea after Ferguson (2008) and to the Noctuoidea following LaFontaine and Schmidt (2010).

Species	Author	Family	Species	Author	Family
Protolampra brunneicollis	Grt.	Noctuidae	*Oligocentria lignicolor*	Wlk.	Notodontidae
Raphia frater	Grt.	Noctuidae	*Oligocentria semirufescens*	Wlk.	Notodontidae
Spodoptera frugiperda	J.E. Smith	Noctuidae	*Peridea angulosa*	J.E. Smith	Notodontidae
Spodoptera ornithogalli	Gn.	Noctuidae	*Peridea basitriens*	Wlk.	Notodontidae
Sympistis infixa	Grote	Noctuidae	*Pheosia rimosa*	Pack.	Notodontidae
Baileya australis	Grt.	Nolidae	*Schizura concinna*	J.E. Smith	Notodontidae
Baileya dormitans	Gn.	Nolidae	*Schizura ipomoeae*	Doubleday	Notodontidae
Baileya ophthalmica	Gn.	Nolidae	*Schizura unicornis*	J.E. Smith	Notodontidae
Meganola minuscula	Zell.	Nolidae	*Symmerista albifrons*	J.E. Smith	Notodontidae
Clostera albosigma	Fitch	Notodontidae	*Actias luna*	L.	Saturniidae
Clostera inclusa	Hbn.	Notodontidae	*Anisota stigma*	F.	Saturniidae
Dasylophia anguina	J.E. Smith	Notodontidae	*Anisota virginiensis*	Drury	Saturniidae
Dasylophia thyatiroides	Wlk.	Notodontidae	*Antheraea polyphemus*	Cram.	Saturniidae
Datana angusii	G. & R.	Notodontidae	*Automeris io*	F.	Saturniidae
Datana integerrima	G. & R.	Notodontidae	*Callosamia angulifera*	Wlk.	Saturniidae
Datana ministra	Drury	Notodontidae	*Callosamia promethea*	Drury	Saturniidae
Datana perspicua	G. & R.	Notodontidae	*Citheronia regalis*	F.	Saturniidae
Ellida caniplaga	Wlk.	Notodontidae	*Dryocampa rubicunda*	F.	Saturniidae
Gluphisia septentrionis	Wlk.	Notodontidae	*Eacles imperialis*	Drury	Saturniidae
Heterocampa guttivitta	Wlk.	Notodontidae	*Hyalophora cercropia*	L.	Saturniidae
Heterocampa obliqua	Pack.	Notodontidae	*Sphingicampa bicolor*	Harr.	Saturniidae
Heterocampa subrotata	Harv.	Notodontidae	*Ceratomia undulosa*	Wlk.	Sphingidae
Heterocampa umbrata	Wlk.	Notodontidae	*Darapsa Myron*	Cram.	Sphingidae
Hyperaeschra georgica	H.-S.	Notodontidae	*Laothoe juglandis*	F.	Sphingidae
Lochmaeus bilineata	Pack.	Notodontidae	*Pachysphinx modesta*	Harr.	Sphingidae
Macrurocampa marthesia	Cram.	Notodontidae	*Paonias excaecatus*	J.E. Smith	Sphingidae
Misogada unicolor	Pack.	Notodontidae	*Paonias myops*	J.E. Smith	Sphingidae
Nadata gibbosa	J.E. Smith	Notodontidae	*Atteva punctella*	Cram.	Yponmeutidae

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.

SOCIO-ECONOMIC IMPLICATIONS OF FOREST MANAGEMENT

PUBLIC ACCEPTABILITY OF FOREST MANAGEMENT PRACTICES AT MORGAN-MONROE STATE FOREST

Shannon C. Rogers, William L. Hoover, and Shorna B. Allred[1]

Abstract.—Forest management practices on public forests are controversial with many organizational and individual stakeholders. Forest managers' understanding of the attitudes of stakeholders is necessary to honor statutory requirements and the social contract under which they operate. The human dimension component of the Hardwood Ecosystem Experiment (HEE) in Indiana included a study of recreationists' and neighboring landowners' forest management attitudes by examining the acceptability of alternative management practices on Morgan-Monroe State Forest (MMSF) before and after providing brief explanations of alternative management practices. An on-site survey of recreationists and a mail survey of landowners neighboring MMSF were used. Both surveys also included an investigation of the influence of information about timber management practices on respondents' attitudes.

As expected, as forest stand density increased, so did the acceptability of management practices, desirability of forest scenes, and likelihood of visiting managed forests for both recreationists and neighboring landowners. Results indicate that informational interventions had a statistically significant influence on the acceptability of forest management practices, but the practical change in attitudes was small. Ordinal regression models indicated that landowner attitudes about the benefits derived by harvesting timber had a small influence on the acceptability of some treatments. It is arguable whether the increase in the acceptability of forest practices resulting from informational intervention is large enough to change stakeholders' positions regarding how MMSF should be managed.

INTRODUCTION

To increase understanding of residents in the Midwest relative to forest management on public land, the present study used survey methodologies to examine stakeholder acceptability of alternative management practices on Morgan-Monroe State Forest (MMSF) by examining the influence of information, values, and visual characteristics on acceptability of specific forest management practices (Rogers 2008).

This research is part of a larger long-range study, the Hardwood Ecosystem Experiment (HEE), being conducted at MMSF (Kalb and Mycroft, this publication). A variety of forest management techniques that involve timber harvesting are being studied. This study provided an opportunity to analyze the acceptability of these techniques. Furthermore, because the HEE was designed to provide detailed cause-and-effect information about forest management practices, interested parties will be able to make scientifically informed judgments about these practices.

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Managers of public forest land are charged with implementing forest management techniques that maintain healthy ecosystems and provide for the benefit of forest stakeholders. The mission statement of the Indiana Department of Natural Resources (IDNR)-Division of Forestry (DoF) charges its staff to "serve as stewards of natural, cultural, historic, and recreational resources for the benefit of present and future generations" (IDNR-DoF n.d.). Although Indiana does not statutorily require the DoF to use a formal public participation process, a broad communications program is conducted to make the general public and forest users aware of management plans and specific activities. DoF foresters recognize that they manage forest land as a public trust for current and future generations. Using stakeholder attitudes to inform forest management allows for a social and ecological balance in decisions. "Social acceptability ... can be used to define a target for managers to strive for, or a threshold of tolerance they dare not fall below" (Brunson 1993: 119).

Even-aged management, commonly referred to as clearcutting, remains especially controversial as a silvicultural treatment. Studies in several states confirm objections to clearcutting. Private forest landowners and the general public in Pennsylvania favored banning clearcutting altogether (Bourke and Luloff 1994). A study in the Tennessee Valley region found that only 14 percent of private forest owners and the general public felt that clearcutting on public forest land is appropriate (Bliss et al. 1994). Oregonians and stakeholders nationwide agreed that clearcutting should be banned on federal land (Shindler et al. 1993). However, professionally trained foresters generally agree that even-aged management is required to maintain the range of successional stages that would exist under natural conditions (Egan 2001, Egan et al. 1997).

Opposition to even-aged management by "casual viewers of forestry activities" is aesthetically driven (Bliss 2000). Members of a variety of organization types in western Washington and Oregon were surveyed by Ribe and Matteson (2002) to find out their views of forest management practices. They categorized respondents into three interest groups: utilitarians desiring forest products from public lands, forest preservationists, and those who support a balance between these two extremes. They also found that even when clearcutting is presented as the safest and most economical harvesting method, most respondents supported minimizing environmental impacts by imposing regulations. Scenic beauty in relation to management acceptability on federal land was the primary determinant. Hikers in the White Mountain National Forest in New Hampshire viewed vistas with no clearcuts as having greater scenic value than those with clearcut areas (Palmer et al. 1995). The greater the cutting intensity in study areas, the less acceptable scenic hikers found it to be. A study of members of natural resource-related organizations indicated that "old forestry," which includes clearcutting and plantations, is unpopular and that simply hiding such managed areas from view is not favored (Ribe and Matteson 2002).

The National Environmental Policy Act of 1969 and the National Forest Management Act of 1976 encourage federal planners to consider scenic beauty (Ribe 2002). Several studies found that managers assumed that if the public finds the scenery acceptable, then the plan and its environmental impacts may be considered acceptable (Greider and Garkovich 1994, Ribe 2002). However, a problem occurs if the social acceptability of a management practice cannot be inferred from scenic acceptability (Ribe 2002). For example, someone may think that a clearcut forest looks atrocious yet finds clearcutting an acceptable practice based on other rationales. Brunson (1993) suggested that forest management practices are judged cognitively based on personal values, not solely on aesthetics. For example, some who believe nature has intrinsic value may find a muddy area more acceptable if it is used as a mud wallow by elk as opposed to a landing for harvested trees (Brunson 1993).

Fire is used to a limited extent as a management tool on MMSF. Its use is likely to increase, however, making its acceptability relevant. The National Fire Management System requires consideration of impacts on recreation use. Englin et al. (2001) found that visitors to national forests in the Intermountain West desired trail access to burned areas for educational purposes. Kneeshaw et al. (2004) found that visitors to national forests in the West generally accepted letting naturally started fires burn out except when damage to private property was likely. However, Englin et al. (2001) noted that the acceptability of fire as a management tool may differ among regions (Bowker et al. 2008).

Those living near public forest land constitute critical stakeholders. They self-identify as stakeholders and use local ordinances to moderate harvesting activity. As the U.S. population continues to increase and spread more widely over landscapes, conflicts arising from the proximity of forest harvesting to forest neighbors will continue to grow (Shelby et al. 2004). Residents living near Oregon State University's research forest complained to forest managers, community leaders, and the State after harvesting was conducted near residents' property (Shelby et al. 2004). Neighbors were concerned that the aesthetic value of the landscape had declined and their property values and recreational opportunities were diminished. Because the potential for conflicts increases when stakeholders are not informed about forest harvesting activities, communications and improved relationships between forest managers and the public was recognized as an important need (Shelby et al. 2004).

A widely held assumption is that if management goals and public perceptions are competing factors, educating the public will draw the two together (Kearney 2001). The link between information and changes in attitudes, perceptions, knowledge, and behavior is complicated. People with formed opinions or strongly held attitudes might not be receptive to new information, making it difficult to change their perceptions (Kearney 2001). Hill and Daniel (2008) had college students rate the acceptability and scenic beauty of landscape scenes from public parks and forests in Missouri. Before rating the scenes, participants were provided a message on the ecological benefits of forest openings and of maintaining tree density. The authors concluded that there was not enough evidence to suggest that the information had an effect on the acceptability or scenic beauty ratings.

However, studies by Anderson (1981), Brunson and Reiter (1996), Hodgson and Thayer (1980), Kearney (2001), and Ribe (1999) propose that a person's perception of a forest scene can be influenced by providing information about the scene. A study in California of county park visitors, college students, and residents sampled in a census tract showed that judgments of landscape beauty can be influenced by the meaning assigned to a landscape (Hodgson and Thayer 1980). Scenes with labels implying human influence, such as a "tree farm," were ranked lower than scenes displaying labels with natural meanings, such as forest growth. Anderson (1981) had introductory psychology students at the University of Arizona evaluate landscape scenes, mostly of pine forests, for scenic quality. The appearance of landscape scenes as well as assigned land-use labels affected judgments of scenic quality among the students. "Wilderness area," "national forest," and "national park" labels yielded higher scenic-quality ratings than "recreation area," "leased grazing area," and "commercial timber stand" labels. Ribe (1999) suggested that managers should focus on more than the appearance of a landscape to gain social acceptability because people use cognitive and visual information when perceiving harvests.

Vining and Tyler (1999) suggested that understanding who the interested public is and how to address its conflicting desires can be beneficial in the development of management plans. They noted that understanding the interested public's values, concerns, and desires is "a moral imperative in a democratic

society" (Vining and Tyler 1999: 21). Gobster (1999) suggested that landscape designers have portrayed 17th- and 18th-century paintings in their designs, giving Americans a "naturalistic" interpretation of the landscape. Wood (1988) implied that this portrayal of art furthers the idea that landscapes are static and neat in appearance. This unrealistic ideal may work in parks, but perceptions need to be broadened where ecological values are the primary concern (Tuan 1989).

In the past 40 years, forest management has broadened by supporting nonhuman benefits, such as providing wildlife habitat, and non-market values, as well as supporting human benefits and market values (Tarrant et al. 2003). Support for nonhuman and non-economic benefits can be seen in a study of Vermont residents regarding national forest management by Manning et al. (1999). Most residents viewed aesthetic and ecological values as most important while market values ranked least important. Tarrant et al. (2003) supported this finding by noting that people who have moved to urban areas or who were already residents of urban areas tend to support non-extractive values because they do not have a physical connection to the forest. They suggested that because public values have been shown to change, land managers need relevant and timely information regarding public views about forest management on public lands.

Managers of public land use public hearings to seek public opinions. However, Vining (2004) suggested that a representative public opinion is not always reflected by traditional public hearings. Some citizens are not comfortable voicing their comments in this type of setting. Vining (1992) contended that the strongly voiced opinions of interest groups may be overrepresented, while the views of the broader public may be unknown to land managers (Vining 2004).

Citizens want to be involved with land management decisions that might affect them. In a study of Oregonians and people from across the United States, Shindler et al. (1993) found that most citizens in both groups support citizen participation in forest policy planning and believe government officials should be most responsive to local affected communities. This finding led the authors to suggest that professionals should not let the dominating views of interest groups overshadow professional assessment of public opinion.

Although there are methods besides surveys, survey research is one technique to obtain the views of the general public (Shindler et al. 1993, Vining 2004). Using surveys is a useful technique because it provides the opportunity to gain a representative public opinion (Shindler et al. 1993, Vining 2004). Because stakeholders of public forests can be affected by management decisions and want to be involved, public land managers should consider those who have a stake in the forest.

Some studies suggest that information may influence acceptability of a management practice (Kearney 2001, Ribe 1999). However, Hill and Daniel (2008) commented that the Kearney (2001) study was inconclusive. Kearney found that information affected forest scene preferences; however, forest treatments after the informational intervention were labeled when the scenes were being judged (Hill and Daniel 2008). The authors suggested that the question of whether information can affect a scenic beauty judgment or a more complex judgment is still an important one for research (Hill and Daniel 2008). Nonetheless, literature on contingent valuation or stated preference has resolved that additional information can have an effect; then the research question is to what extent and in what direction (Cummings and Taylor 1999). The present study attempts to substantiate and build on past findings and provide needed information about acceptability to stakeholders to reduce conflict between them and forest managers in the management of MMSF.

MATERIALS AND METHODS

Two categories of stakeholders were sampled. The proximity of MMSF to the Indianapolis metropolitan area and Indiana University's Bloomington campus provides a large pool of MMSF recreational users, referred to as "recreationists." The large number of private tracts, many touching only one boundary of a MMSF tract or none at all, provides a large pool of neighboring landowners, referred to as "landowners." Essentially the same survey was used for both categories and the same hypotheses tested; therefore, the same analytical procedures were used for both groups. Data were used to assess the impact of information on acceptability of forest treatments and how visual characteristics influence the acceptability, desirability, and likelihood of visiting MMSF. The study also examined whether knowledge of the economic and ecological benefits of timber harvests affected acceptability. Data were analyzed using SPSS 15.0 for Windows (SPSS Inc., Chicago, IL).

Survey Instrument

The survey instrument was a 24-page booklet with color pictures of representative stands post-treatment (see Appendix 1 for survey questions). It contained questions regarding: attitudes about forest harvesting when outputs are identified, acceptability of forest management conditions, desirability of forest scenes, likelihood of visiting a forest area, and demographics. Responses were in the form of a five-option Likert scale: strongly agree, agree, neutral, disagree, and strongly disagree. Participants were asked to respond while thinking of actively managed federal and state-owned forest land in Indiana. Timber management treatments included in the survey were no harvest, single-tree selection, shelterwood, and clearcut. Although not involving harvests, burned forest scenes were also included. The four harvesting treatments were assigned a relative stand density based on trees per hectare. Post-treatment stand densities from lowest to highest were level 1 - clearcut, level 2 - shelterwood, level 3 - group selection, and level 4 - no harvesting. The fire photographs were not ranked because they do not represent a level of tree removal. The fire scenes were of burned fully stocked stands. Demographic information collected were gender, year of birth, place of residence, frequency of public forest use, and recreational activities pursued.

Informational Intervention

The informational intervention consisted of two to three paragraphs (approximately 180 words) and an adjacent photograph representing the corresponding forest management practice. The informational paragraphs were reviewed by foresters for accuracy and bias. The Flesch-Kincaid grade level for the descriptions ranged from 9.4 to 11.1. Respondents first answered the questions without having read the information paragraphs, "pre-information." They then responded to the same questions after reading the informational paragraphs, "post-information."

Pre- and Post-Intervention Questions

Questions in the pre- and post-information sections referred to color photographs of the five management practices. Participants were asked: "Imagine you were having an enjoyable hike in a forest on public forestland (federal and state owned forestland) in Indiana. At different places along the trail, you stop to look at the landscape on either side. A picture has been taken of each of these scenes you might experience while hiking on the trail. Please answer the following questions about each of these forest scenes as though they were something you saw during your hiking experience." Questions concerning acceptability of forest management conditions, desirability of a forest scene, and likelihood of visiting a forest area accompanied each photograph.

Sampling Methods

Recreational Users

It was impractical to identify MMSF recreationists by sampling the general population in the region. Recreationists self-identified by their presence at parking lots within MMSF. The attitudes of this category were assessed using an on-site survey with mail follow-up.

Surveys were conducted on-site from August to October 2007 by enumerators making personal contact with visitors in parking lots. Although MMSF is open year-round, the fall season was selected because the MMSF manager reported that this is the season of maximum use. Sampling was conducted on Tuesday, Thursday, Friday, and Saturday. Friday was included because many recreationists start their weekend activities on Friday. Weekdays were considered Monday through Thursday, and weekend days were considered Friday and Saturday. Surveys were conducted in variable 5-hour blocks randomly selected in the range of 9 a.m. to 9 p.m. Two hundred seventy-four people were approached at the forest to participate in the survey.

Recreationists could fill out the survey while at the forest or return it later by mail. In the latter case, names and addresses were collected and a reminder postcard sent about a week later. A replacement survey was mailed to non-respondents a week after the postcard was mailed. If the survey was not sent back within that week, a third and final survey was mailed a week after the second survey was sent. Of the 239 recreationists that accepted a survey, 165 respondents returned it (58 during the forest visit, 107 later by mail), yielding a 69-percent response rate.

Neighboring Landowners

The 2,515 owners of land in all Public Land Survey sections, 259 ha (640 acres), bordering all tracts of MMSF were sent a mail survey using the Tailored-Design Method (Dillman 2000). Contact information was obtained from records at the Morgan County and Monroe County assessors' and recorders' offices. Out of the 2,515 survey instruments mailed, 90 were undeliverable and 1,239 completed surveys were returned, a 51-percent response rate. Surveys that were at least 80 percent complete were included in the analysis. Eleven surveys did not meet this criterion. No duplicate cases were found using SPSS software.

Statistical Procedures

Paired-samples t-tests were used to analyze the hypothesis that information had an impact on acceptability of a forest management practice, defined by the level of stand density. Repeated measures one-way analysis of variance (ANOVA) tests with post-hoc Bonferroni adjustment for the number of comparisons were conducted for the four levels of stand density. The density levels were treated as discrete variables.

Logistic regression models were used to test the hypothesis that respondents' level of agreement with the importance of outputs and their demographic profiles were correlated with their acceptance of the specified forest management practices. The basic model was: Agree(1-5) = Logistic (output importance, demographics).

Recreational Activity Groupings

Logistic modeling of the acceptability of treatments was facilitated by categorizing recreational activities into two mutually exclusive groups: consumptive-motorized, and non-consumptive-non-motorized. Although the recreationist and adjacent landowner groups were analyzed separately, the recreational activity categorizations were the same. The consumptive-motorized category consisted of: fishing, boating, off-roading (snowmobile or all-terrain vehicle), hunting, and mushroom gathering. The non-consumptive-non-motorized category comprised camping, wildlife viewing, canoeing/kayaking, hiking, biking, picnicking, photography, and horseback riding. Two binary variables resulted, one for the type of recreational activities respondents engaged in, and one for the activities they were engaged in on the day they were invited on-site to participate in the survey. The second variable was not applicable for landowners because they were contacted by mail.

Harvesting Benefits

It was hypothesized that information about the benefits of harvesting timber influenced respondents' attitudes about harvesting. Two factors were identified using the principal components analysis extraction method with Varimax rotation: acceptability of forest outputs for human benefits, and acceptability of forest outputs for ecological benefits. Cronbach's alpha was equal to 0.877 for acceptability of forest outputs for human benefits and 0.854 for acceptability of outputs of forests for ecological benefits. These factors explained 56.9 percent of the total variance.

Ordinal logistic regression was used to test the influence of forest outputs on acceptability and desirability of each forest management practice. Strongly agree and agree were combined to form one variable, as were disagree and strongly disagree. Acceptability (or desirability) of each practice was the dependent variable with three ordinal levels: (1) strongly agree and agree, (2) neutral, and (3) strongly disagree and disagree. The independent variables were the acceptability of forest outputs for human and ecological benefits, and acceptability of managing for old-growth forests.

Visual Acceptability

We hypothesized that the visual characteristics of a forest scene were correlated with its acceptability by both recreationists and landowners. The four scenes presented in the questionnaire (see Appendix 1) represented an associated management practice defined visually by its stand density. Before and after acceptability for the fire scene was not tested because stand density was not representative of this treatment. Respondents were asked to respond to the same questions both before and after the informational intervention included in the questionnaire. Repeated-measures one-way ANOVA with post-hoc Bonferroni adjustment and paired t-tests were used to analyze whether there were differences in acceptability between the before and after levels.

RESULTS

Response and Descriptive Statistics

Recreationists

One hundred sixty-five people returned the survey instrument. Surveys at least 80 percent complete were included in the analysis (N=163). Respondents reported whether they used public forests weekly, monthly, several times per year, or rarely. The typical respondent was a male born in 1962 who had come to MMSF to hike on the day invited to participate in the survey (Table 1).

Table 1.—Demographic profile of recreationists and landowners and participation in outdoor recreation activities.

Recreationists		Landowners	
Characteristic/activity	Percent of respondents	Characteristic/activity	Percent of respondents
Live in a suburb	48	Live in rural area	83
Gender		Gender	
Female	40	Female	38
Male	60	Male	62
Mean year of birth: 1962		Mean year of birth: 1951	
Activity		Activity	
Camping	82.2	Camping	52.8
Fishing	52.1	Mushroom gathering	54.3
Wildlife viewing	57.7	Wildlife viewing	64.5
Hiking	89.6	Hiking	74.6
Picnicking	60.7	Picnicking	50.2

Landowners

Surveys were returned by 1,239 landowners, of whom 1,228 had completed at least 80 percent of the survey. Landowners reported how often they use public forests for recreation: weekly, monthly, several times per year (occasional visitors), or rarely. The most frequent response was several times per year (39.7 percent, n=1,202). The typical respondent was a male born in 1951 who lived in a rural area and hiked (Table 1). A majority indicated that they did not feel informed about forest management activities occurring on MMSF (78.1 percent, n=1,200). Fifty-two percent reported that the property they owned does not share a border with any parcel of MMSF. This situation results from the high degree of fragmentation of tracts constituting MMSF (Jenkins, this publication).

Stand Density and Acceptability

Repeated measures one-way ANOVA tests for both categories, recreationists and neighboring landowners, revealed that there were significant differences in acceptability of forest management practices between the four levels of stand density, both pre- and post-information at the $p < 0.001$ level (Table 2). Bonferroni comparisons indicated that the means of all four stand density levels were significantly different from each other pre-information. This was also the case except for the recreation category, post-information. For level 2 the means of levels 1 and 3, 1 and 4, 2 and 3, and 3 and 4 were not significantly different. For level 3 the means of 1 and 2, 1 and 4, 2 and 3, and 2 and 4 were not significantly different (Table 2).

Harvesting Benefits and Acceptability

Separate ordinal logistic regression models for both categories were constructed for the acceptability of each treatment pre- and post-information based on the benefits of timber harvesting.

Pre-Information

The pre-information models for the recreationist category for group selection, clearcut, and no harvest were not significant (Table 3). The model for the acceptability of shelterwood treatment (pre-information) was significant ($p < 0.01$) with one significant effect in the model (Table 3). Respondents who disagreed or strongly disagreed with outputs of forests for ecological benefits were 13 times more likely to exhibit a one-unit reduction in acceptability score for shelterwood forest management compared to people who agreed or strongly agreed. The model for acceptability of fire (pre-information) also was significant ($p < 0.05$). There were two significant effects in this model (Table 4): gender and frequency of public forest use. Males were 4 times more likely than females to exhibit a one-level increase in acceptability score for fire management. People who reported using public forests weekly for recreation were 15 times more likely to increase by one level the acceptability for fire management compared to those who reported rarely using public forests for recreation. For the remainder of the logistic models only significant variables are reported for significant models (treatments). See Rogers (2008) for the complete models.

Table 2.—Repeated measures ANOVA for stand density effects on forest treatment acceptability. Treatment groups are recreationists pre-information (Rec-pre) and post-information (Rec-post) and neighboring landowners pre-information (Neighbors-pre) and post-information (Neighbors-post). Level 1 = clearcut, level 2 = shelterwood, level 3 = group selection, level 4 = no harvesting.

	Means by level				d.f.				
Treatment	1	2	3	4	Num.	Denom.	MS	F	P
Rec-pre	2.36[a]	3.57[c]	3.22[b]	4.49[d]	2.9	454	129.3	151.1	<0.0001
Rec-post	2.72[a]	3.90[b]	3.76[b]	4.44[c]	2.8	443	91.5	122.2	<0.0001
Neighbors-pre	2.36[a]	3.51[c]	3.06[b]	4.36[d]	2.9	3,462	882.6	991.0	<0.0001
Neighbors-post	2.75[a]	3.77[c]	3.65[b]	4.27[d]	2.4	2,893	593.4	517.8	<0.0001

Means with the same lettered superscript are not significantly different at alpha = 0.05.

Table 3.—Significance of ordinal logistic regression model for effects of forest outputs on forest treatment acceptability to recreationists (pre-information). Asterisk (*) values indicate significance at alpha = 0.05.

Model	Group selection	Clearcut	Shelterwood	No harvest	Fire
Model fitting information					
Chi-square	29.177	25.912	38.311	20.799	34.856
d.f.	19.0	19.0	19.0	19.0	19.0
Significance	0.063	0.133	0.005*	0.348	0.015*
Goodness-of-fit					
Pearson					
Chi-square	271.381	252.877	242.22	80.364	269.725
d.f.	251	251	253	255	255
Significance	0.18	0.455	0.676	1	0.252
Deviance					
Chi-square	252.177	221.725	196.505	37.559	230.056
d.f.	251	251	253	255	255
Significance	0.467	0.908	0.997	1	0.867
Test of parallel lines					
Chi-square	10.176	13.312	14.935	18.865	28.462
d.f.	19	19	19	19	19
Significance	0.948	0.822	0.727	0.466	0.075

Table 4.—Parameter estimates for the effects of forest outputs on fire acceptability to recreationists (pre-information). Asterisk (*) values indicate significance at alpha = 0.05.

Independent variable	Estimate	SE	Sig.	OR[a]
Outputs for ecological benefits				
Disagree	1.519	1.069	0.155	4.57
Neutral or undecided	0.525	0.561	0.349	1.69
Agree	Reference			
Outputs for human benefits				
Disagree	-0.683	0.598	0.253	1.98
Neutral or undecided	-0.349	0.509	0.492	1.42
Agree	Reference			
Old-growth				
Disagree	-0.069	1.047	0.947	1.07
Neutral or undecided	-1.115	0.741	0.133	3.05
Agree	Reference			
Consumptive/motorized				
No participation	0.531	0.419	0.205	1.70
Participation	Reference			
Nonconsumptive/motorized				
No participation	-0.585	1.550	0.706	1.79
Participation	Reference			
Gender				
Male	1.337	0.394	0.001*	3.81
Female	Reference			
Age				
Under 25	1.061	1.012	0.294	2.89
25 to 34	1.159	0.763	0.129	3.19
35 to 44	0.080	0.771	0.918	1.08
45 to 54	0.653	0.708	0.357	1.92
55 to 64	0.604	0.766	0.430	1.83
Place you live				
Rural	0.475	0.588	0.419	1.61
Suburban	-0.214	0.474	0.652	1.24
Urban	Reference			
Public forest use				
Weekly	2.677	0.955	0.005*	14.54
Monthly	0.943	0.782	0.228	2.57
Several times per year	1.228	0.763	0.108	3.41
Rarely	Reference			

[a] OR is an odds ratio calculated as $e^{|ß|}$, where ß is the coefficient estimate.

For the landowner category, ordinal logistic regression showed significant model fits ($p < 0.01$) pre-information for all treatments (Table 5). However, the no-harvest acceptability model violated the test of parallel lines, negating interpretations. An assumption in ordinal regression is that the effect of the independent variables is the same across all levels of the dependent variable. If this assumption is violated, estimates may be biased (Garson 2006). The significant parameters for the other four treatments were outputs for ecological benefits, human benefits, and age (Table 6).

Post-Information

For the recreationists category, the post-information models for the acceptability of no harvest and fire were not significant ($p > 0.05$, Table 7). The models for group selection, clearcut, and shelterwood were significant. However, the group selection and clearcut acceptability models violated the test of parallel lines. Therefore, the coefficients were not interpreted. The model fit for shelterwood acceptability was significant ($p < 0.01$). People who disagreed or strongly disagreed with outputs of forests for ecological benefits were 71 times more likely to exhibit a one-unit reduction in acceptability score for shelterwood forest management compared with people who agreed or strongly agreed (Table 8).

The landowner models for all five treatments (post-information) were significant ($p < 0.01$, Table 9). However, the no-harvest and fire management acceptability models violated the test of parallel lines. Therefore, the coefficients were not interpreted. For the group selection, clearcut, and shelterwood treatments, people who reported disagreeing or strongly disagreeing with outputs for ecological benefits were 12, 16, and 26 times more likely, respectively, than people who reported agreeing or strongly agreeing to decrease one level of acceptability for these treatments (Table 10). People in the 25 to 34 years of age category were 2.2 and 2.0 times more likely to decrease one level of acceptability for group selection and clearcut treatments, respectively. People had the strongest feelings toward the clearcut treatment, indicated by seven significant variables. This was the case especially for people using public forests weekly.

Table 5.—Ordinal logistic regression model for effects of forest outputs on forest treatment acceptability to neighboring landowners (pre-information). Asterisk (*) values indicate significance at alpha = 0.05.

Model	Group selection	Clearcut	Shelterwood	No harvest	Fire
Model fitting information					
Chi-square	90.598	186.038	279.233	45.941	99.005
d.f.	19	19	19	19	19
Significance	0.000*	0.000*	0.000*	0.001*	0.000*
Goodness-of-fit					
Pearson					
Chi-square	1067.728	1064.728	1171.572	894.398	1077.645
d.f.	1075	1073	1073	1073	1071
Significance	0.557	0.565	0.019*	1	0.437
Deviance					
Chi-square	1115.907	903.565	1042.761	425.368	1069.857
d.f.	1075	1073	1073	1073	1071
Significance	0.188	1.000	0.740	1	0.504
Test of parallel lines					
Null hypothesis					
Chi-square	22.236	9.908	28.199	48.855	27.085
d.f.	19	19	19	19	19
Significance	0.273	0.955	0.080	0.000*	0.103

Table 6.—Estimates of only the significant parameters for the effects of forest outputs on group selection, clearcut, shelterwood, and fire acceptability by neighboring landowners (pre-information). Asterisk (*) values indicate significance at alpha = 0.05.

Independent variable	Estimate	SE	Sig.	OR[a]
Group selection treatment				
Outputs for ecological benefits				
Disagree	-1.101	0.242	0.000*	3.01
Neutral or undecided	-0.375	0.166	0.024*	1.45
Agree	Reference			
Outputs for human benefits				
Disagree	-0.662	0.180	0.000*	1.94
Neutral or undecided	-0.450	0.153	0.003*	1.57
Agree	Reference			
Age				
45 to 54	0.477	0.170	0.005*	1.61
55 to 64	0.357	0.165	0.030*	1.43
Public forest use				
Weekly	0.423	0.210	0.044*	1.53
Clearcut				
Outputs for ecological benefits				
Disagree	-1.126	0.331	0.001*	3.08
Neutral or undecided	-0.620	0.208	0.003*	1.86
Agree	Reference			
Outputs for human benefits				
Disagree	-1.536	0.203	0.000*	4.65
Neutral or undecided	-1.009	0.158	0.000*	2.74
Agree	Reference			
Old-growth				
Disagree	0.945	0.268	0.000*	2.57
Neutral or undecided	0.638	0.194	0.001*	1.89
Agree	Reference			
Age				
45 to 54	0.678	0.191	0.000*	1.97
55 to 64	0.390	0.187	0.037*	1.48
Shelterwood				
Outputs for ecological benefits				
Disagree	-2.264	0.258	0.000*	9.62
Neutral or undecided	-0.998	0.174	0.000*	2.71
Agree	Reference			
Outputs for human benefits				
Disagree	-1.255	0.215	0.000*	3.51
Neutral or undecided	-0.411	0.197	0.037*	1.51
Agree	Reference			
Fire				
Nonconsumptive/motorized				
No participation	-0.606	0.269	0.024*	1.83
Participation	Reference			
Gender				
Male	0.427	0.132	0.001*	1.53
Female	Reference			
Age				
25 to 34	0.961	0.327	0.003*	2.61
35 to 44	1.182	0.228	0.000*	3.26
45 to 54	0.825	0.176	0.000*	2.28
55 to 64	0.871	0.170	0.000*	2.39
Public forest use				
Monthly	0.476	0.211	0.024*	1.61
Rarely	Reference			

[a] OR is an odds ratio calculated as $e^{|ß|}$, where ß is the coefficient estimate.

Table 7.—Model information for effects of forest outputs on forest treatment acceptability to recreationists (post-information). Asterisk (*) values indicate significance at alpha = 0.05.

Model	Group selection	Clearcut	Shelterwood	No harvest	Fire
Model fitting information					
Chi-square	47.545	36.164	64.857	24.306	28.222
d.f.	19	19	19	19	19
Significance	0.000*	0.01*	0.000*	0.185	0.079
Goodness-of-fit					
Pearson					
Chi-square	259.426	246.914	253.583	74.101	297.371
d.f.	253	255	255	255	249
Significance	0.377	0.63	0.513	1	0.019*
Deviance					
Chi-square	160.196	257.214	118.942	36.284	187.161
d.f.	253	255	255	255	249
Significance	1	0.449	1	1	0.999
Test of parallel lines					
Null hypothesis					
Chi-square	56.768	45.255	26.417	17.061	49.644
d.f.	19	19	19	19	19
Significance	0.000*	0.001*	0.119	0.586	0.000*

Table 8.—Parameter estimates for the effects of forest outputs on shelterwood acceptability to recreationists (post-information). Asterisk (*) values indicate significance at alpha = 0.05.

Independent variable	Estimate	SE	Sig.	OR[a]
Outputs for ecological benefits				
Disagree	-4.266	1.092	0.000*	71.24
Neutral or undecided	-0.538	0.676	0.426	1.71
Agree	Reference			
Outputs for human benefits				
Disagree	-1.626	1.010	0.107	5.08
Neutral or undecided	-0.654	0.896	0.465	1.92
Agree	Reference			
Old-growth				
Disagree	16.825	7410.535	0.998	>100
Neutral or undecided	0.423	1.094	0.699	1.53
Agree	Reference			
Consumptive/motorized				
No participation	-0.701	0.553	0.205	2.02
Participation	Reference			
Nonconsumptive/motorized				
No participation	-31.273	0.000	0.000	>100
Participation	Reference			
Gender				
Male	-0.755	0.600	0.208	2.13
Female	Reference			
Age				
Under 25	0.196	1.514	0.897	1.22
25 to 34	-0.932	1.081	0.388	2.54
35 to 44	0.798	1.176	0.497	2.22
45 to 54	-0.037	1.089	0.973	1.04
55 to 64	0.053	1.197	0.965	1.05
Place you live				
Rural	-0.720	0.793	0.364	2.05
Suburban	0.015	0.688	0.982	1.02
Urban	Reference			
Public forest use				
Weekly	2.006	1.111	0.071	7.43
Monthly	1.869	0.993	0.060	6.48
Several times per year	2.810	1.042	0.007*	16.61
Rarely	Reference			

[a] OR is an odds ratio calculated as $e^{|ß|}$, where ß is the coefficient estimate.

Table 9.—Model information for effects of forest outputs on forest treatment acceptability to neighboring landowners (post-information). Asterisk (*) values indicate significance at alpha = 0.05.

Model	Group selection	Clearcut	Shelterwood	No harvest	Fire
Model fitting information					
Chi-square	231.648	361.787	392.417	92.889	52.604
d.f.	19	19	19	19	19
Significance	0.000*	0.000*	0.000*	0.000*	0.000*
Goodness-of-fit					
Pearson					
Chi-square	1116.113	1090.123	1122.712	1150.958	1102.955
d.f.	1071	1077	1073	1073	1067
Significance	0.165	0.384	0.142	0.049*	0.216
Deviance					
Chi-square	970.843	987.737	753.959	553.528	1092.644
d.f.	1071	1077	1073	1073	1067
Significance	0.987	0.975	1	1	0.286
Test of parallel lines					
Null hypothesis					
Chi-square	16.179	18.230	16.899	62.263	38.753
d.f.	19	19	19	19	19
Significance	0.645	0.507	0.597	0.000*	0.005*

Table 10.—Parameter estimates for the effects of forest outputs on group selection, clearcut, and shelterwood acceptability to neighboring landowners (post-information). Asterisk (*) values indicate significance at alpha = 0.05.

Independent variable	Estimate	SE	Sig.	OR[a]
Group selection				
Outputs for ecological benefits				
Disagree	-2.445	0.254	0.000*	11.53
Neutral or undecided	-1.069	0.182	0.000*	2.91
Agree	Reference			
Outputs for human benefits				
Disagree	-0.630	0.221	0.004*	1.88
Age				
25 to 34	-0.794	0.329	0.016*	2.21
Clearcut				
Outputs for ecological benefits				
Disagree	-2.762	0.447	0.000*	15.83
Neutral or undecided	-0.987	0.184	0.000*	2.68
Agree	Reference			
Outputs for human benefits				
Disagree	-1.706	0.193	0.000*	5.51
Neutral or undecided	-0.610	0.157	0.000*	1.84
Agree	Reference			
Age				
25 to 34	-0.715	0.325	0.028*	2.04
Public forest use				
Weekly	-0.721	0.230	0.002*	2.06
Monthly	-0.506	0.215	0.018*	1.66
Rarely	Reference			
Shelterwood				
Outputs for ecological benefits				
Disagree	-3.257	0.282	0.000*	25.97
Neutral or undecided	-1.635	0.201	0.000*	5.13
Agree	Reference			
Outputs for human benefits				
Disagree	-1.138	0.294	0.000*	3.12
Agree	Reference			
Consumptive/motorized				
No participation	-0.414	0.209	0.048*	1.51
Participation	Reference			
Age				
45 to 54	-0.632	0.255	0.013*	1.88

[a] OR is an odds ratio calculated as $e^{|ß|}$, where ß is the coefficient estimate.

Stand Density and Visitation

Repeated measures one-way ANOVA tests indicated that there were significant differences in the likelihood of visiting a forest between the four levels of stand density pre- and post-information for both recreationists and neighbors (Table 11). Bonferroni comparisons for the recreationist category revealed that means were significantly different ($p < .05$) from each other between three levels of stand density pre- and post-information. The mean difference in stand density from level 2 to level 3 was not significant. Bonferroni comparisons for the neighboring landowner category revealed that all four means were significantly different from each other. Mean likelihood of visiting a forest significantly increased from stand density level 1 to stand density level 2 and again from stand density level 3 to stand density level 4. Pre- and post-information, mean likelihood of visiting a forest significantly increased as stand density increased (Table 11).

Stand Density and Desirability

Repeated measures one-way ANOVA tests indicated significant differences in desirability of forest management practices among the four levels of stand density pre- and post-information for both recreationists and neighboring landowners (Table 12). Bonferroni comparisons revealed that all four means were significantly different from each other pre- and post-information. For both pre- and post-information, mean desirability of a forest management practice significantly increased from stand density level 1 (lowest) to stand density level 2 and again from stand density level 3 to stand density level 4 (highest). The mean desirability of a forest management practice significantly decreased from stand density level 2 to stand density level 3 both pre- and post-information.

Information

Paired-sample t-tests were used to compare the mean pre- and post-information ranking of the acceptability of the five treatments. The rankings were the average Likert scale with 1 being the least acceptable and 5 being the most acceptable. Information increased the ranking of all treatments for both categories ($p < 0.01$) except no harvest (Table 13).

Table 11.—Repeated measures ANOVA for stand density effects on likelihood of visiting forests with different treatments. Treatment groups are recreationists pre-information (Rec-pre) and post-information (Rec-post) and neighboring landowners pre-information (Neighbors-pre) and post-information (Neighbors-post).

	Means by level				d.f.				
Treatment	1	2	3	4	Num.	Denom.	MS	F	P
Rec-pre	2.67[a]	3.75[b]	3.53[b]	4.57[c]	2.9	457	101.9	144.6	<0.0001
Rec-post	2.77[a]	3.92[b]	3.87[b]	4.58[c]	2.5	398	111.1	148.4	<0.0001
Neighbors-pre	2.63[a]	3.58[c]	3.26[b]	4.36[d]	2.8	3,354	675.1	823.2	<0.0001
Neighbors-post	2.76[a]	3.80[c]	3.69[b]	4.37[d]	2.4	2,818	679.2	702.1	<0.0001

Means with the same lettered superscript are not significantly different at alpha = 0.05.

Table 12.—Repeated measures ANOVA for stand density effects on desirability of treatments. Treatment groups are recreationists pre-information (Rec-pre) and post-information (Rec-post) and neighboring landowners pre-information (Neighbors-pre) and post-information (Neighbors-post).

	Means by level				d.f.				
Treatment	1	2	3	4	Num.	Denom.	MS	F	P
Rec-pre	2.16[a]	3.42[c]	2.91[b]	4.46[d]	2.9	458	156.9	208	<0.0001
Rec-post	2.40[a]	3.70[c]	3.47[b]	4.42[d]	2.8	447	122.9	168	<0.0001
Neighbors-pre	2.25[a]	3.37[c]	2.81[b]	4.30[d]	2.8	3,404	984.6	1199	<0.0001
Neighbors-post	2.44[a]	3.63[c]	3.39[b]	4.29[d]	2.5	2,933	860.2	863	<0.0001

Means with the same lettered superscript are not significantly different at alpha = 0.05.

Table 13.—Results of paired t tests for changes in ranking of acceptability of treatments pre- and post-information. Results are shown for both recreationists and neighboring landowners.

	Acceptability ranking		
Treatment	Pre-information	Post-information	Significance
Recreationists			
Group selection	3.23 (sd = 1.176)	3.76 (sd = 0.915)	t(160) = 3.787, p<0.001
Clearcut	2.35 (sd = 1.147)	2.71 (sd = 1.192)	t(160) = 3.787, p<0.001
Shelterwood	3.58 (sd = 1.004)	3.89 (sd = 0.870)	t(160) = 3.823, p<0.001
No harvest	4.48 (sd = 0.697)	4.44 (sd = 0.763)	t(160) = 0.489, p>0.05
Fire	3.49 (sd = 1.076)	3.71 (sd = 0.936)	t(159) = 2.855, p<0.01
Neighbors			
Group selection	3.05 (sd = 1.164)	3.65 (sd = 1.032)	t(1210) = 17.657, p<0.001
Clearcut	2.35 (sd = 1.164)	2.75 (sd = 1.311)	t(1213) = 11.260, p<0.001
Shelterwood	3.50 (sd = 1.118)	3.78 (sd = 1.031)	t(1215) = 8.948, p<0.001
No harvest	4.36 (sd = 0.812)	4.27 (sd = 0.879)	t(1207) = 2.964, p<0.01
Fire	3.47 (sd = 1.175)	3.58 (sd = 1.013)	t(1186) = 3.197, p<0.01

DISCUSSION

For recreationists and landowners alike, stand density influenced the acceptability and desirability of forest management practices and likelihood of visiting. The pattern of acceptability was similar for pre- and post-information ratings. As the forest stand became denser, acceptability increased. Recreationists' acceptability of intermediate density levels was less clear, even reversed in some cases. The relationship for landowners was reversed for intermediate treatments. This idiosyncrasy might have resulted from respondents' difficulty in assessing stand density for intermediate levels. The group selection photographs depicted tree removal that created an opening in the forest. The shelterwood photographs showed dispersed tree removal. Respondents may have felt that the stand density in the shelterwood was as dense as or more dense than that of the group selection opening because of the differing patterns of tree removal. Brunson and Shelby (1992) found similar results for scenic acceptability. Respondents in their study found old-growth most scenically acceptable and clearcut stands least acceptable, whereas stands with intermediate cutting were in the middle.

Pretesting the instrument with a subsample of recreationists and landowners might have detected misinterpretation of the shelterwood photograph. The instrument was pretested in a focus group of graduate students, and taken by undergraduate and graduate students to determine completion time.

Leaving some trees after a harvest may make the forest more desirable and people more likely to visit, even if stand density is less than that of another area. Tönnes et al. (2004) found that clearcut scenes were increasingly preferred as more trees were shown remaining in the area after harvest, and that respondents preferred retention trees in the form of single mature trees, such as the shelterwood condition in the present study, rather than in groups.

Overall, this study suggests that information about management practices can impact acceptability ratings by recreational users. However, mean ratings did not clearly increase to the next level of acceptability for any forest treatment. The results showed a small but significant increase in acceptability from scenes ranked before information about forest management practices was provided to scenes ranked after the information was provided. Information did not change the order of acceptability. In contrast, information did not change acceptability for the no-harvest scene. This result may reflect the acceptability of this treatment pre-information. The no-harvest treatment was in effect the control treatment. These results contrast with Hollenhorst et al. (1993). They reported that information had little effect on aesthetic preferences, suggesting that some informed participants were more critical of tree damage by gypsy moth (*Lymantria dispar*) while others were more tolerant. In contrast, however, the no-harvest treatment for MMSF was not associated with any forest health issues.

Attitudes about forest outputs influenced the acceptability of some forest treatments. Neighboring landowners who disagreed with harvesting and managing the forest for human benefits were more likely to find group selection, clearcut, and shelterwood management practices less acceptable compared to those who agreed with these uses. This finding was true both pre- and post-information. The information about management practices included benefits other than human benefits. However, neighbors who find these forest management practices less acceptable may not believe that the other benefits are truly provided by harvesting, or that they outweigh their negative opinions about harvesting for human benefits.

Before forest information was provided, people who were neutral or undecided about harvesting and managing the forest for human benefits were more likely to find group selection, clearcut, and shelterwood management practices less acceptable compared to those who agreed with harvesting and managing for human benefits. However, after information was provided, this finding was true only for clearcutting. The information provided about forest management was apparently sufficient to dissuade those undecided respondents from considering group selection and shelterwood management practices less acceptable than do those who agree with managing the forest for human benefits.

Neighboring forest landowners who disagreed with harvesting or managing the forest for ecological benefits were more likely to find group selection, clearcut, and shelterwood management practices less acceptable compared to those who agreed with harvesting and managing for ecological benefits. This result was observed in both the pre- and post-information cases. This finding suggests that neighboring landowners who disagreed with harvesting for ecological benefits did not believe ecological benefits were provided by harvesting. Thus, they were less accepting of harvesting compared to those who agreed with harvesting for ecological benefits.

People who are neutral or undecided about harvesting or managing the forest for ecological benefits were more likely to find group selection, clearcut, and shelterwood management practices less acceptable compared to those who agree with harvesting or managing for ecological benefits. This was the case pre- and post-information.

As with recreationists, landowners considered no harvesting as the most acceptable forest management practice (Table 2). Group selection, shelterwood, and fire management practices ranged between neutral and acceptable but approached acceptability after information was provided. For both the recreationists and neighboring landowners, information seemed to elicit the greatest increase in acceptability of group selection (Table 2). Clearcutting was unacceptable but approached neutrality post-information. Information about forest management practices had a significant impact on landowner acceptability ratings. Reading information about the forest management practices included in this study led to increased acceptability scores for fire and harvesting practices and slightly lower scores for no harvesting (Table 2), all of which are consistent with a greater appreciation for the value of forest management. After reading the information about all of the forest treatments, respondents may have felt the benefits of harvesting outweighed the benefits of not harvesting.

Visual characteristics of a forest stand were examined in relation to forest treatment acceptability and desirability and recreational behavior. Stand density, the visual characteristic examined in each case, influenced recreationists' acceptability, desirability, and likelihood of visiting a forest. Moreover, the same patterns were evident for all three dependent variables. Acceptability, desirability, and likelihood increased in each case from lowest stand density to highest stand density, but the opposite trend occurred for intermediate stand densities. Respondents most likely were confused about the perceived residual stocking in forest stands with intermediate densities. Because this overall pattern was seen before and after the informational intervention, it is likely that information does not affect the stand density preferred by neighboring forest landowners. Stands with more trees are identified as more acceptable and desirable, and recreationists are more likely to visit them whether or not information is provided. Other studies found the same results: the denser a forest stand, from clearcuts to old-growth forests, the more scenically acceptable a forest was rated (Brunson and Reiter 1996, Brunson and Shelby 1992). Respondents also viewed intermediate cuttings that have more residual stocking as more acceptable, as Brunson and Shelby (1992) found.

Landowners who disagreed with managing forests for human benefits or ecological benefits were more likely to find group selection, shelterwood, and clearcutting less acceptable than those who agreed with managing for human benefits or ecological benefits, respectively. Respondents were asked to answer questions only about public lands, and those who disagreed with managing for human benefits may have found harvesting less acceptable on public land than on private land. These findings are similar to Bliss et al. (1994) for forest landowners and non-owners in the Tennessee Valley Region. Only a small percentage of respondents agreed that clearcutting should be allowed on public land, compared to a significantly higher proportion who agreed with allowing clearcutting on land owned by paper and lumber companies or individuals. Thus, individuals may believe that clearcutting or other practices should occur elsewhere. In addition, respondents may prefer that the forest be managed for more ecological reasons. Even though information was provided highlighting ecological benefits, these respondents may not believe harvesting is truly necessary to accomplish certain ecological goals. This belief may explain why those who disagree with harvesting for ecological benefits are less accepting of harvesting.

CONCLUSIONS AND RECOMMENDATIONS

The analyses found many similarities between the recreationist and neighboring landowner groups for acceptability of forest management practices, desirability of forest scenes, and likelihood of visiting. The same forest management practices were found

acceptable and unacceptable by both categories. For both categories acceptability increased from lowest stand density to highest stand density. Of the two partial cutting methods, the shelterwood may have appeared to have more residual stocking, resulting in higher preferences. Similarly, respondents in the study by Tönnes et al. (2004) preferred forest scenes with higher tree density.

The present study supports findings by Brunson and Reiter (1996), Ribe (1999), and Kearney (2001), which found that providing information about a forest management practice can affect preferences for those practices. The study is dissimilar to studies by Hollenhorst et al. (1993) and Hill and Daniel (2008), which found that information had little or no effect on scenic beauty preferences or acceptability of forest management practices.

The results of this study support an increased emphasis on outreach activities to explain the purposes of the HEE. Even a nominal amount of unbiased information would be helpful. We assumed that respondents to our survey thought of the information as coming from the managers of MMSF. Increased acceptability may result from in-depth information. The information provided should help stakeholders better understand the practical and scientific bases of treatments, and tradeoffs among them. Dialog with the managing forester and staff would provide an opportunity for stakeholders to understand the benefits and negative consequences of a forest treatment and to answer general questions about the forest management practices applied in the specific forest. The correlation between quality, quantity, and form of education is not known. Additional research should focus on what would be required to switch mean acceptability of the lower-density treatments from unacceptable to acceptable.

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APPENDIX 1.—SURVEY QUESTIONS

Part A: Recreationist Survey Questions

Your Views on Indiana Public Forests

The purpose of this questionnaire is to obtain information about your forest values and attitudes towards management of public forests. This questionnaire is part of a research project being conducted by the Department of Forestry and Natural Resources at Purdue University.

To help us understand the various opinions of landowners, please take a moment to share your views.

This questionnaire is voluntary and anonymous and will take approximately 15 minutes to complete.

No personal information will be collected, so **please do not write your name on the questionnaire.**

If you have questions or want further information, please contact Shannon Rogers at srogers@purdue.edu or Bill Hoover, Ph.D., at whoover@purdue.edu, (765) 494-3580.

Participants must be 18 years of age or older.



For the purposes of this survey, **public forests will include** Federal and State owned forestland in **Indiana** such as National Forests and State Forests (**not** National and State Parks)

When thinking about public forestlands, how important are the following? (Please place an X in the appropriate box.)

The opportunity to enjoy the beauty of nature.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to protect nature in order to ensure human well-being.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to camp, hike, and participate in other recreational activities in nature.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to learn more about nature.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to respect and protect nature and other living things.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to see and experience historical and cultural sites.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to maintain or regain physical health or mental well-being through contact with nature.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity for scientists to study nature and ecology.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to think creatively and be inspired by nature.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to get closer to God or obtain other spiritual meaning through contact with nature.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

The opportunity to get timber, minerals, and other natural resources from nature.

- ☐ Very unimportant
- ☐ Somewhat unimportant
- ☐ No opinion
- ☐ Somewhat important
- ☐ Very important

When thinking about public forestlands, how much do you agree or disagree with the following? (Please place an X in the appropriate box.)

It is acceptable to harvest trees from forests to lessen the severity or intensity of forest fires.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to use forests to produce products such as paper and lumber that humans can use.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to harvest trees from a forest to provide revenue for the state.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to manage forests for roads, accommodations, and services to help local tourism businesses.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable for local industries and communities to have access to raw materials on forests.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable for forests to be managed for non-timber forest products.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to manage forests to protect old-growth forests.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to harvest trees from a forest if it improves habitat for wildlife.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to harvest trees from a forest if it improves outdoor recreation opportunities.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to harvest trees from a forest if it provides jobs and income.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to harvest trees from a forest to provide income needed to manage forestland in Indiana.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to harvest trees from a forest to promote forest health and vigor.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

It is acceptable to harvest trees from a forest to promote diversity of tree species in the forest.

- ☐ Strongly disagree
- ☐ Disagree
- ☐ Neutral or undecided
- ☐ Agree
- ☐ Strongly agree

Instructions:

For the next set of questions, **imagine you were having an enjoyable hike in a forest on public forestland** (Federal and State owned forestland) in **Indiana**. At different places along the trail, you stop to look at the landscape on either side. A picture has been taken of each of these scenes you might experience while hiking on the trail. **Please answer the following questions about each of these forest scenes as though they were something you saw during your hiking experience.**



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely

Instructions:

For the next set of questions, **please read the information describing how the forest in each picture is managed**.

Then, **imagine you were having an enjoyable hike in a forest on public forestland** (Federal and State owned forestland) in **Indiana**. At different places along the trail, you stop to look at the landscape on either side. A picture has been taken of each of these scenes you might experience while hiking on the trail. **Please answer the following questions about each of these forest scenes as though they were something you saw during your hiking experience.**

Group Selection Opening

Removal of a few trees to create openings that vary in size from ½ to 5 acres is called group selection. This is an uneven-aged management practice since these group openings are created over time to create small stands of trees in the forest that vary by age as well as size.

The trees in these small openings often can be of several different species which compete for light and moisture with nearby trees. These openings in the forest are placed where trees are damaged, growing slower than normal, or where growth of desired trees is not possible without allowing for more sunlight to reach the forest floor. Treetops and branches from the cut trees provide shelter for wildlife and help prevent soil erosion.



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely

Clearcut

Sometimes, foresters remove all merchantable trees within a stand to create openings 10 to 25 acres in size. This even-aged management practice, called clearcutting, encourages the survival of trees that need full sunlight. An opening this size ensures that the trees surrounding the opening will not shade the sun-loving seedlings in the opening. Trees grow rapidly in these openings with some growing much faster than others creating a stand that varies in size after a few years.

Clearcuts are sometimes used in locations where an entire area of the forest has been damaged by wildfire or storms. They may also be used to correct past activities that left many undesirable trees and plants. The use of clearcut harvests favors the growth of forests dominated by oaks and hickories or ash, cherry, and yellow-poplar.



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely

Shelterwood

Another even-aged practice, called a shelterwood, is the removal of trees from 10 to 25 acres in 2 or 3 stages. The first cut leaves a few trees scattered across the site to provide for the establishment of new seedlings and sprouts. The remaining trees are removed in 3 to 10 years after the new stand is established. The shelterwood practice provides for the growth of tree species that require an intermediate amount of sunlight in the early years of growth.

Leaving a few large trees protects the site as new seedlings grow and helps to prevent erosion. The result is a natural return of a mixture of tree species. Increasing amounts of sunlight reaching the forest floor and the extended supply of seeds from the remaining trees will allow oaks and hickories to compete with other trees that need full sunlight and trees that grow well under the shade of other trees.



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely

Unmanaged

Unmanaged forests generally favor trees such as sugar maple that can reproduce and grow in the shade of other trees. Natural disturbances such as wind throw and fire create openings in the forest that allow trees to grow which require openings with full sunlight. The species of trees in an unmanaged forest will also vary with changes in growing conditions such as dry or wet soils.

Although no trees are cut or removed, unmanaged forests are full of continuous activity and change. Trees of different ages, sizes, and species occur in an unmanaged forest. Once old trees die, they donate their nutrients back to the soil and create small openings in the forest that other trees quickly fill.



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely

Fire

In the past, oaks were present in large numbers in Indiana's forests. They were able to reproduce and grow because of natural and man-made disturbances, such as fire. Fires were set by Native Americans and early European settlers. The fires released nutrients and opened forests to full sunlight needed by oaks and other sun loving species.

Under certain circumstances and in proper places, fire can help achieve certain management goals. For example, fire is used to get rid of fuel that could build up and cause future damaging wildfires. At times, low intensity forest fires can start naturally from dry conditions. Sometimes these isolated fires are allowed to burn unchecked to maintain the natural ecology of an area under certain conditions and in some forests.



Please place an X in the appropriate box.

How acceptable is the condition of the landscape represented in this scene as part of a publicly owned and managed National or State forest?

- ☐ Very unacceptable
- ☐ Unacceptable
- ☐ Neutral or undecided
- ☐ Acceptable
- ☐ Very acceptable

How desirable is this scene as a site to encounter along a hiking trail?

- ☐ Very undesirable
- ☐ Undesirable
- ☐ Neutral or undecided
- ☐ Desirable
- ☐ Very desirable

How likely are you to visit a forest again if you encountered this scene during your hike?

- ☐ Very unlikely
- ☐ Unlikely
- ☐ Neutral or undecided
- ☐ Likely
- ☐ Very likely

Please place an X in all boxes that apply.

What type of outdoor recreational activities do you participate in?

- ☐ Camping
- ☐ Hiking
- ☐ Fishing
- ☐ Hunting
- ☐ Wildlife watching
- ☐ Biking
- ☐ Boating
- ☐ Picnicking
- ☐ Canoeing/kayaking
- ☐ Photography
- ☐ Off-roading (snowmobile or ATV)
- ☐ Mushroom collecting
- ☐ Horseback riding
- ☐ Other _____

What brings you to the Morgan-Monroe State Forest today?

- ☐ Camping
- ☐ Hiking
- ☐ Fishing
- ☐ Hunting
- ☐ Wildlife watching
- ☐ Biking
- ☐ Boating
- ☐ Picnicking
- ☐ Canoeing/kayaking
- ☐ Photography
- ☐ Off-roading (snowmobile or ATV)
- ☐ Mushroom collecting
- ☐ Other _____

How often would you say that you use public forests for recreation?

- ☐ Weekly
- ☐ Monthly
- ☐ Several times per year
- ☐ Rarely

What is your gender?

- ☐ Male
- ☐ Female

How would you describe the place where you live?

- ☐ Rural
- ☐ Suburban
- ☐ Urban

In what year were you born?

Thank you for your time!

APPENDIX 1 (continued).

Part B: Landowner Survey Questions

All pages in the landowner survey are exactly the same as the recreationist survey instrument (Appendix 1, part A) except the last page of the survey, which is shown here.

Please place an X in all boxes that apply.

What type of outdoor recreational activities do you participate in?

- ☐ Camping
- ☐ Hiking
- ☐ Fishing
- ☐ Hunting
- ☐ Wildlife watching
- ☐ Biking
- ☐ Boating
- ☐ Picnicking
- ☐ Canoeing/kayaking
- ☐ Photography
- ☐ Off-roading (snowmobile or ATV)
- ☐ Mushroom collecting
- ☐ Horseback riding
- ☐ Other _____
- ☐ None

How often would you say that you use public forests for recreation?

- ☐ Weekly
- ☐ Monthly
- ☐ Several times per year
- ☐ Rarely

Do you feel informed about forest management activities occurring on the Morgan-Monroe State Forest?

- ☐ No
- ☐ Yes

How would you describe the place where you live?

- ☐ Rural
- ☐ Suburban
- ☐ Urban

Is the property where you live adjacent to the Morgan-Monroe State Forest?

- ☐ No
- ☐ Yes

What is your gender?

- ☐ Male
- ☐ Female

In what year were you born?

Thank you for your time!

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.

VALUE OF THE MORGAN-MONROE-YELLOWWOOD STATE FOREST COMPLEX

William L. Hoover[1]

Abstract.—As publicly owned forest land, the Morgan-Monroe and Yellowwood State Forests (Indiana), referred to herein as the Morgan-Monroe Yellowwood Complex (MMYC), have many stakeholders with differing management expectations. The Hardwood Ecosystem Experiment (HEE) within the MMYC will significantly increase the science-based information available for forest management decisions. The primary focus of HEE is the impacts of alternative timber management systems on associated communities of flora and fauna. The results will be used to evaluate tradeoffs associated with mixes of silvicultural treatments. Rogers et al. (this publication) found that the values recreationists and adjacent landowners associate with forest scenes declined with decreasing stand density. The range of values ascribed to forests is discussed in this paper to provide a broader context for interpretation and application of HEE findings. Particular emphasis is given to monetized institutional values using discounted cash flow analyses of silvicultural treatments. The results indicate that land expectation values are lower for silvicultural treatments with lower stand densities because of higher opportunity costs from pushing timber revenues into the future. The implications of these findings were found to differ depending on management objectives and the scale at which decisions are made.

INTRODUCTION

Wood was the foundation of the North American economy from earliest European settlement until the coal age (Perlin 1989). Forests are still synonymous with wood products and value-added activities for many stakeholders. Although wood remains a vital commodity, the return of fully stocked forests has provided opportunities for viewing them in a broader context. In many areas public forests offer the only accessible and relatively large contiguous blocks needed for aesthetic value. Forests give some individuals a mystical feeling, especially when in the presence of large trees. Others sense they are a creature of the forest, their true ancestral home. Others feel lost and out of place in a forest. Some forests are cultivated like crops, *Fructus industriales*; others are *Fructus naturales*, resulting in varying perceptions of human intervention in the dynamics of forests. Policy makers and forest managers are challenged to choose options that account for this wide range of attitudes, especially considering the short-term impacts of some silvicultural techniques. Many forest benefits are complementary; others are mutually exclusive, depending on desired outcomes and the timeframe used for evaluation of alternatives.

Many stakeholders' expectations are based on values not rooted in forest science and the tradeoffs estimated for the full range of benefits. Results from the Hardwood Ecosystem Experiment (HEE) in Indiana will be more useful if they are framed within the broad range of values stakeholders will bring to the interpretation of findings and implications drawn.

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After setting the Morgan-Monroe-Yellowwood Complex (MMYC) in its physical and institutional contexts, I review the types of values and discuss their applications to forests. The fiscal management of the MMYC is not discussed because of its political context. Specifically, the 100-year horizon of the HEE makes it necessary to avoid to the extent possible the vagaries of cyclic changes in political climate. For similar reasons I do not discuss opportunities to generate revenue streams from non-timber forest uses, or the efficiency and efficacy of current administrative and operational management practices.

Given the HEE's focus on alternative silvicultural techniques, it was important to value socio-economic tradeoffs among them. Monetized instrumental value, i.e., market value, of revenue and cost streams was used. Standard discounted cash flow techniques were applied. Land expectation values (LEV) were estimated for a range of assumptions for three silvicultural techniques and for the MMYC as a whole using the current allowable cut. Future analyses should consider the relationship between the mix of silvicultural treatments applied across the entire forest and allowable cut.

Data for monetization of recreation activities are available only for hunting and fishing; these limited data were assessed and estimates made when possible. The results indicate that additional estimates of on-site recreational activities and existence value are necessary for comprehensive management decisions.

MORGAN-MONROE-YELLOWWOOD COMPLEX

The HEE is centered in the MMYC, managed by the Indiana Department of Natural Resources (IDNR)-Division of Forestry (DoF) as a unit with a single supervisor. Understanding the economic, social, and cultural roles of the MMYC requires an understanding of the context in which it is located and managed.

The MMYC includes 19,152 ha of land and 61 ha in four lakes. The largest at 54 ha is Yellowwood Lake, used intensively for fishing and as the setting for picnicking and limited camping. The 132 km of maintained trails range from 0.8 to 68 km in length. There are also 34 km designated as horse trails. This infrastructure accommodates a large, but unquantified, level of recreational use.

The MMYC is the northern boundary of the most heavily forested region of Indiana. Parcels constituting the MMYC lie in Morgan, Brown, and Monroe Counties (Kalb and Mycroft, this publication). Approximately 54 percent of these three counties is forested. If Morgan County is excluded, 66 percent is forested, compared with the statewide average of 15 percent.

The MMYC lies within the Highland Rim Natural Region, bounded on the north by the Martinsville Hills Region, on the east by the Scottsburg Lowland, on the south by Mitchell Plateau, and on the west by Crawford Upland (Homoya and Huffman 1997). The Martinsville Hills Region lies at the southern edge of Indiana's glaciated region. It was formed primarily by the erosive and channeling action of glacial outwash. It is characterized by ridges and hollows that provide scenic variety for recreation activities, but also management challenges to maintain the environmental quality established by reforestation following exploitation of the MMYC lands during the early 20th century (Carman, this publication; Jenkins, this publication).

Spatial Aspects

State Road 37 (SR 37), a four-lane highway with at-grade crossings except in the Bloomington area, provides easy travel between Indianapolis and Bloomington and direct access to the MMYC. Extensive housing development is occurring, driven by SR 37 improvements in Johnson County, directly north of the MMYC. Development may speed up when the

Interstate 69 north-south corridor overlapping the SR 37 corridor is completed. The MMYC is the southern boundary of this development. Recreational use of the forest is expected to increase dramatically with population density. This increased use will change the usage balance between residents in the immediate vicinity of the forest and those driving in from up to 80 km away. Development of a trail system connecting Martinsville to the MMYC is expected over the next 10 to 15 years. This development will result in usage pressures in the northern MMYC similar to those currently experienced from Bloomington and Nashville residents and visitors on the southern edge of Yellowwood State Forest adjacent to Brown County State Park.

It is likely that repeat recreational users of the MMYC have favorite sites and trails although researchers have not explored this topic. The value of some recreation activities should not be extrapolated to the entire MMYC. However, value is added for those using their favorite sites from knowing that these sites are part of a larger forest. Those stakeholders whose primary use is scenic drives focus on management of roadside vegetation and scenic vistas.

Harvest levels, the so-called allowable cut, are based on timber surveys and growth estimates for the entire ownership. Recreational use, however, may limit the area to which the allowable cut can be allocated. Most casual recreational use is concentrated along trails and minimally developed sites. Hunting activities are carried out with a roadway as the base point. Gatherers of mushrooms and other non-timber forest products typically have favorite sites, spread across a broader area than trail hikers (Hembram and Hoover 2008). Information about the location of these activities and the manner in which they are carried out is limited to casual observations by DoF employees.

Access and Policing Aspects

Based on characteristics defined by McKean (2000), the MMYC is a common-pool resource in terms of access by recreationists. Common-pool refers not to the physical characteristics of a resource, but "institutional arrangements for the cooperative (shared, joint, collective) use, management, and sometimes ownership of natural resources" (McKean 2000: 27). She defines this category as "goods that can be kept from potential users only at great cost or with difficulty but that are subtractable in consumption and can thus disappear" (McKean 2000: 28). Timber can be characterized in practical terms as a private resource because state statutes authorize harvesting.

Vehicular access is restricted to paved county highways, reducing the possibility of timber theft. Forest roads are gated, allowing only foot traffic. Neighbors watch for timber theft and other inappropriate activities. Non-timber forest products, primarily herbaceous species, are open access because it is not practical to police gathering, even of controlled species. Stripping of bark from live trees is a minor problem in the MMYC, unlike locations in other parts of the Central Hardwood Region (Hembram and Hoover 2008). Use of recreational sites in high demand, such as Yellowwood Lake and picnic shelters, is monitored by MMYC employees and conservation officers on call to enforce regulations. Self-registration is required for use of campsites, all of which are primitive. The level of compliance is unknown. Hunters have open access except for marked safety zones.

Owners of tracts bordering the MMYC are primary stakeholders. In a mail survey of Indiana forest landowners, Broussard[2] (2006) broke out respondents from what she called the Brown County Hills Area, which includes the MMYC. She found that 58 percent of the responding owners classified themselves as conservative and had owned their land for 24 years

[2] Broussard, S.R. 2006. Understanding your perspectives: a survey of Indiana forestland owners. Unpublished report. 50 p. On file with: Purdue University, Department of Forestry and Natural Resources.

on average. Respondents in the Brown County Hills Area were the least likely of the five areas surveyed by Broussard to have goals for their land. Sixty-seven percent of Brown County Hills respondents included timber production as a goal. Even more, 75 percent, also had some aspect of wildlife as a goal. However, only 22 percent had sought professional help in managing their land. Less than one-half of the owners had ever sold timber. This characterization presents challenges for the development of collaborative management plans across boundary lines.

POLICY FRAMEWORK

Estimating the value of natural resources, especially when defined as complex ecosystems, is more than an academic exercise. All resources are limited and have alternative uses. Decisions about the rate of consumptive and non-consumptive uses are determined by comparing the benefits and costs of alternative uses and management regimes. A major tradeoff is between timber revenue and how managed forests are visually perceived by recreational users (Rogers et al., this publication). The value estimates made here will not allow policy makers to conduct a complete comparison of costs and benefits for alternative management regimes. Because data are available, market-based direct benefits from timber harvesting are discussed in detail. Estimates of the direct costs to grow and market timber from the MMYC are accessible. The management benefits and direct and indirect costs from knowledge of non-market benefits and associated costs are discussed and quantified to the extent possible with the data available.

Protected Areas Classification

Although Indiana's state forests are not generally considered to be protected areas, they could be so considered under the classification system used by the International Union for Conservation of Nature (IUCN) (Dudley 2008). The primary focus of IUCN's protected areas program is developing countries, but the paradigm developed is applicable to the MMYC. There are six IUCN Protected Areas and World Heritage Program land-use categories: I - strict nature reserves/wilderness areas, II - national parks, III - natural monuments, IV - habitat/species management areas, V - protected landscapes/seascapes, and VI - managed resource protection areas. Bringing the MMYC initially under Category VI classification would increase the interest of the research community and forest users in the HEE research program, as well as in how the MMYC is managed. This classification would complement the existing Sustainable Forestry Initiative and Forest Stewardship Council certifications awarded to Indiana's State Forests.

The HEE could provide the scientific basis for reclassifying the MMYC from Category VI, an area managed for the sustainable use of natural ecosystems, to Category IV, an area managed mainly for conservation through management intervention. Category IV is defined as an "area of land and/or sea subject to active intervention for management purposes so as to ensure the maintenance of habitats and/or to meet the requirements of specific species." The management objectives for Category IV lands are (1) to secure and maintain the habitat conditions necessary to protect significant species, groups of species, biotic communities, or physical features of the environment where these require specific human manipulation for optimum management; (2) to facilitate scientific research and environmental monitoring as primary activities associated with sustainable resource management; (3) to develop limited areas for public education and appreciation of the characteristics of the habitats concerned and of the work of wildlife management; (4) to eliminate and thereafter prevent exploitation or occupation inimical to the purposes of designation; and (5) to deliver such benefits to people living within the designated area as are consistent with the other objectives of management (Dudley 2008). Raising the classification of the MMYC to Category IV based on HEE research results and subsequent adjustments of management strategies could be a longer-term management goal.

What Determines Value?

Value exists when an institution or people recognize that "property" in some way provides them with a benefit within the context of the relationship between the subject and object (Sheil and Wunder 2002). It is necessary to clearly specify the context and terms of assessment for estimated values to have meaning in a policy framework. Many disputes over management derive from a lack of such an understanding.

"Property" for purposes of this discussion means all attributes of the MMYC, and everything associated with it directly and indirectly. The term "ecosystem" best captures the intended meaning. An ecosystem has static values because of the mere presence of its components at a given point in time, and dynamic values determined by the resiliency and systematic interactions of its components, and with the larger landscape and airshed in which it exists.

Property also refers to place. People dramatically impact the places where they live and work, but place also defines in part who they are (Hawley 1986, Schama 1995). Quality of life for many is determined in large part by where they carry out their lives. Adequate wealth allows many to choose where they live and work. Natural landscapes close at hand are important to many residents' choice of where to live. The MMYC defines the sense of place for many residents in the vicinity of the MMYC. In addition, the location of privately owned tracts within the general area of the MMYC creates value for the owners of these tracts.

The value of the MMYC derives in part from the wilderness roots of residents and visitors. These roots create in them a so-called pioneer spirit, a drive to overcome obstacles. This spirit is associated with historical heroes, such as Daniel Boone in the East and Kit Carson in the West. We are allowed to hunt on most of the "king's" (public) land and add the fruits of success to our larder. The collection and use of weapons is ingrained in the culture of many, especially rural, residents. Country and western music, at least in its original form, recalls the pioneer spirit for many. When those with a choice prefer to reside in a rural rather than urban landscape, Bunce (1994) ascribes that preference to a vestige of our tie to the wilderness our ancestors overcame, and a desire to live a rural lifestyle, even if done on a riding mower.

The center of Indiana's pioneer spirit is Brown County, home of Nashville, where forests and rural landscapes are celebrated in art, music, and treks in Brown County State Park, or the MMYC for those taking a path less traveled. The value of this area is defined primarily by its forest cover. This value is reflected in highly prized and high-priced second homes in the woods and short-term visitors' expenditures.

Intrinsic and Cultural Values

The intrinsic value of the MMYC is the most difficult for some stakeholders to appreciate because of its basis in moral judgments of right and wrong. The logic used to make moral judgments has the same philosophical basis as the scientific method. Both make observations, apply logic, and draw conclusions. The naturalist philosophy holds that the scientific method should be used for the investigation of all areas of reality, even the "human spirit" (Zalta 2007).

A focus of the HEE is the impact of increased timber harvests to increase early successional habitat, and the best silvicultural systems to use for desired outcomes. Most stakeholders value forests in static terms, i.e., what they see when they return to their favorite trail or campsite. Stakeholder support for managing for early successional habitats should be based on explanations of forest dynamics, i.e., a forest as a constantly changing environment. Acceptance of this need requires that stakeholders accept the need for a mix of habitat types. The discussion is commonly framed in terms of increased population of specified bird species. A stakeholder is asked to accept the value

of target bird species and to accept the following reasoning: These species require early successional forest habitat, this habitat is not adequate on other lands to support the desired population levels of the target species, increasing the habitat will increase the population of the target species, and the value of this increase is greater than the value of the default condition of no harvesting. The no-harvest alternative substantially reduces signs or even knowledge of human interference in the progress of the forest toward biological senescence, usually described as old growth. The value judgment is that a higher population of the target species is more important than allowing the progression toward an old-growth state.

The truths of all the elements of this logical chain are subject to hypothesis tests, but rejecting all the null hypotheses would not eliminate the need for individual stakeholders to make their final value judgment. Therefore, policy makers must somehow weigh the implications of the value judgments made by stakeholders. It would not be valid to base management decisions entirely on the scientific evidence proving that a larger area of early successional forest in the MMYC is more valuable than an old-growth state.

Many people ascribe spiritual values to forests. Such values can best be described in philosophical terms. Because it is difficult to capture this value in setting land-use and forest management policies, conflicts often arise among parties advocating policies focused on the extremes of spiritual and market values. For many stakeholders the MMYC represents a new Arcadia, a state of mind that can exist without being physically experienced. Arcadia may take form as Wordsworth's formal English garden, Thoreau's Walden Pond on the edge of a New England town, Ansel Adam's images of El Capitan, or even second-growth forests in Indiana. Treating the MMYC as a new Arcadia is an approach hard to accept for those who read the land in detail across decades.

Aesthetic Value

Aesthetic value exists at a personal level through a psychological response when the observer perceives beauty in the subject (Rolston 1995). The subject may exist physically or conceptually. Value arises when the viewer is willing to expend time and effort to be afforded the opportunity to partake of the beauty in some form. Time and effort are directly associated with money, a medium of exchange, that can be exchanged for experiential opportunities.

Instrumental Value

Something has instrumental value when it plays a role in making something possible. An individual tree has instrumental value with regard to its conversion to a different form, e.g., a wood product. This value is represented by market value. Market value is an instrumental value, i.e., the value that can be derived from the use of an object. Goods and services that are to be sold at their fair market value are offered for sale in an open market where all potential buyers are aware of the offering and have full knowledge of the characteristics of the good or service, any restrictions on its use, and any other factor that would affect its value to a bidder. An assessment process is used to estimate fair market value, and the assessed value may be set as a minimum acceptable price by the seller. Few markets are perfect in this sense, especially when the seller is an agency of government subject to political influence and rent-seeking activity. Timber sales on the MMYC are made using a sealed-bid process; therefore, rent seeking is not a concern.

A forest also has instrumental value as a system that is more than the sum of its parts. Ecosystem service values are now widely recognized, but monetized to a very limited extent (Salzman 2005).

Market Value

The market value of forest land, as with land generally, is derived primarily on the capitalized stream of benefits that the land is expected to generate over

a defined time period, often infinite. Economists refer to a stream of benefits as rent in classical economics. Rent can be from timber harvests, hunting rights, non-timber forest products, and payments for environmental services, such as carbon sequestration. Some benefits are public, accruing to all without one person's use diminishing the amount available to others. Other benefits accrue to a small group of users other than the outright owner. The benefits accruing to an outright owner are limited to those for which they have secured rights. Land rights refer to the holder's ability to capture this stream of benefits. Benefit streams not captured by the holder result in undercapitalization of the true value of the land and conversion to land uses with higher monetized benefit streams (Wunder 2005).

In the major timber-producing regions of the United States, the market value of forest land may similarly be determined by its capacity to produce periodic crops of timber. However, estimates are complicated by the need to determine optimal periodicity (Samuelson 1976). Aronow et al. (2004) estimate that two-thirds of the total returns from timberland are from appreciation in the value of the land itself, timber volume growth, and timber prices. Although lumber price-reporting services have been available since shortly after World War II (e.g., Random Lengths [2010]), reporting of forest land prices did not start until institutional investors became interested in the advantages of forest land for portfolio diversification (Conroy and Miles 1989). Several major life insurance companies had timberland in their portfolios as early as 1950 (Binkley et al. 1996). The need for vertically integrated forest products manufacturers to capture the value of their timberland holdings for stockholders made large holdings available to institutional investors and later to private equity investment groups that accumulated capital for purchases of timberland contracted for management services from timber investment management organizations.

Although an early application of modern portfolio theory to forest investments considered returns from agriculture and timber in Indiana (Mills and Hoover 1982), forest land sale prices were not available for this analysis. Market values of crop land are reported by Purdue University's Department of Agricultural Economics (Dobbins and Cook annual), but prices paid for forest land are not reported. Detailed delivered log price data have been available since 1957 (Hoover and Preston annual) and are used to calculate a price index for trend monitoring. This index can be used as a proxy for financial returns from holding timber, exclusive of volume growth and increases in land value. Based on the index for a high-quality stand of timber, the real annual rate of return is declining on average (Fig. 1). The year-to-year rate of return from holding timber for another year was 2.75 percent for 1957 to 1958. From 2009 to 2010 the rate was -0.002 percent. Between 1957 and 2010, the standard deviation of returns was 0.876. Variation increased substantially after the recession in the mid-1970s, when price inflation of commodities, including forest products, began to trend upward.

There is no basis for estimating the liquidation value of the MMYC land considering the area held. Liquidation value would provide an estimate of the lost one-time revenue and associated opportunity cost incurred

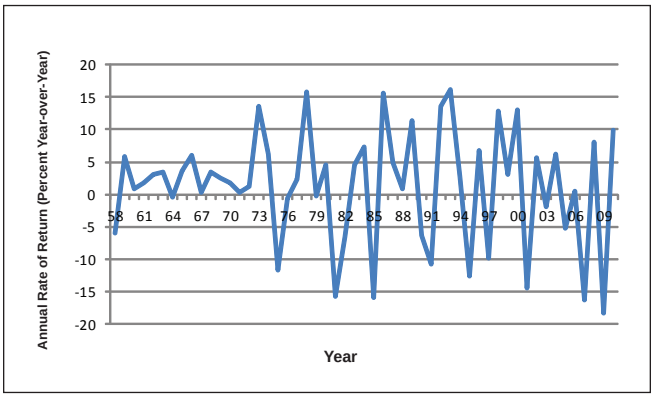


Figure 1.—Annual rate of return from holding timber in Indiana.

by holding the land, but there is no commercial timberland in Indiana to use for comparable sales. Commercial forest land in this context is generally defined as tracts of forest land large enough to attract institutional and private equity investors. A few large hardwood lumber and veneer producers own upwards of 4,047 ha each in scattered tracts of 4 to 81 ha. An additional indication that these lands are not part of a commercial timberland pool is that they are generally held for decades, not turned over periodically to provide cash flow for dividend payments. Large tracts of commercial timberland are open for hunting and other recreational uses in many states. The lack of such lands places additional pressures on public forest land in Indiana.

The U.S. Department of Agriculture, Forest Service (USFS), Forest Inventory Analysis (FIA) conducts an ongoing survey of forest land but no longer attempts to estimate the amount of timberland owned by "forest industry." Timberland is defined for the FIA assessment as forest land capable of producing more than 20 cubic feet per acre (49 cubic feet per ha) per year of industrial wood crops under natural conditions. The last estimate of industry ownership was made for 1998. Of the total 1,757,266 ha of timberland in Indiana, forest industry was estimated to own 7,042 ha and corporations 385,000 ha. In the Knobs FIA unit where the MMYC is located, 3,076 ha were classified as owned by forest industry and 45,365 by corporations. The Knobs Unit includes 694,967 ha, 40 percent of the timberland in Indiana.

Recreation Value

The place where many outdoor recreation (OR) activities occur is as important as the activity itself. The scale of OR venues varies greatly, but many of the most commonly engaged-in activities require large areas. These venues can be linear, such as urban and inter-urban trails, or a large contiguous area such as the MMYC. Recreationists desiring an experience in a more primitive area, such as large blocks of forest land, do not want to have any indicators of development within view. Many also receive pleasure knowing that they are some distance from development and have viewscapes consisting primarily of forests.

The scenic quality of landscapes constitutes a recreational resource (Dearden 1980). Driving through a scenic landscape to reach a venue can create anticipation for what will happen upon arrival, or the drive itself may constitute the activity. A trip to and from the Smoky Mountains, Grand Tetons, or similar areas can be the substance of the activity. For those heading south from Indianapolis on SR 37, the forested ridges immediately south of Martinsville (the northern edge of the MMYC) are the introduction to the most heavily forested area in Indiana. The paved roads through the MMYC are used as scenic byways. The northern edge of the MMYC thus has greater scenic value than those State Forests embedded in the more extensively forested landscapes farther south.

The landscape and interior viewscapes necessary to achieve a desired effect, perhaps best described as a sense of place, are personal. At one extreme are true wilderness areas that allow hikers to be totally out of contact with others for days. Although Indiana has a designated wilderness area, it does not equal the large areas usually envisioned. At the other extreme might be Indiana's State Parks, which are void of almost all manipulation of the terrain and vegetation, but restrict users to designated pathways and intensive-use areas. The MMYC is different because it is actively managed and allows unrestricted movement. Most visitors, however, take advantage of the trail network.

Statewide Outdoor Recreation Plan

The IDNR-Division of Outdoor Recreation conducts periodic surveys of the general population and of individuals engaging in specific OR activities. The results are used as the basis of the Indiana Statewide Comprehensive Outdoor Recreation Plan (SCORP).

The most recent plan covers the period 2006-10 (IDNR-Division of Outdoor Recreation 2007). SCORP ensures that Indiana is eligible for federal funding under the Land and Water Conservation Fund Act of 1965. Equally important, SCORP provides guidance to all public and private OR providers.

Outdoor recreation is an integral part of the lifestyles of a majority of Indiana individuals and families. The OR participation survey of the general public conducted for SCORP indicated that in 2003 OR activities were "essential" to 54 percent of respondents. Eighty-five percent participated in walking/hiking/jogging activities. The second most frequent activity category was fairs/festivals at 68 percent. SCORP states that the large spread between these two leading activity categories in terms of participation indicates a "tremendous need for trails and other linear activity opportunities" (IDNR-Division of Outdoor Recreation 2007: 23). Other frequent activities were in the swimming/SCUBA/snorkeling category at 60.7 percent, nature photography (59.8 percent), camping (53.5 percent), and fishing (52.3 percent). Most respondents reported engaging in activities within more than one category, and more than one activity within a category.

The MMYC is a venue for all of the most important OR activities reported by Indiana residents. The most frequent activity, walking, was broken down in the SCORP report into types of walking, with walking for pleasure undertaken by 78 percent, and hiking by 39 percent of respondents. Most frequent activities in the nature observation/photography category were wildlife viewing by 55.1 percent and fall foliage viewing by 53.2 percent of respondents. The camping category was led by tent camping at 63 percent, greater than RV/trailer camping at 34 percent. In the hunting category, deer hunting dominated at 72.3 percent. Eleven percent indicated that they participated in horseback riding.

The MMYC's proximity to Indianapolis, the state's major population center, and the large active young-adult population in the Bloomington area increase its recreation value. The 2003 OR participation survey did not include questions related to how far respondents were willing to travel to participate in an activity. It did, however, ask a question about acceptable length of travel time to reach a new or improved OR facility. Thirty-three percent were willing to travel for 16 to 60 minutes, 21 percent 60 to 90 minutes, 17 percent 2 hours, 6 percent 3 hours, and 12 percent more than 3 hours. The authors indicated that the disparity in the last two categories may be due to respondents' distinguishing between partial-day activities and all-day activities. The participation survey did not differentiate between day-use facilities and destination facilities. The latter is a facility for which a majority of the users stay over at least 1 night (Van Doren et al. 1985). They also found that "most households repeatedly use a single facility and that there are significant behavioral differences between those who visit a single facility and those who go to a variety of places" (Van Doren et al. 1985: 17).

ESTIMATES OF VALUES OF THE MMYC

Forest Recreation

It is necessary to use secondary sources to estimate the potential monetary value of the MMYC for OR activities. If survey data for the number of visits and segment type were available, these secondary data could provide rough estimates of recreation expenditures. Given the proximity of the Hoosier National Forest (NF), survey data for that forest from the USFS's National Visitor Use Monitoring (NVUM) program should be informative. The first full cycle of the NVUM (Stynes and White 2005) program was conducted from January 2000 to September 2003. The survey uses standardized definitions of "national forest site visits" and "national forest visits." The former is

the entry of one person upon a National Forest site or area to participate in recreation activities for an unspecified period of time. The latter is the entry of one person upon a National Forest to participate in recreation activities not at specific sites and not for a specified period of time. Thus, an NF visit could consist of more than one site visit. A "viewing corridor visit" is the entry of one person onto a travelway, such as a road, trail, railroad, or river that is neither owned nor managed by the USFS, for the purpose of viewing scenery on USFS lands adjacent to the travelway.

The NVUM data are used to develop spending profiles for NF use (Stynes and White 2005). Because of small sample sizes for individual forests, a weighted average profile is developed based on the survey data for high and low categories. The Hoosier NF is in the low category. Expenditure estimates were made for seven visitor trip type segments: (1) Non-local day trips – non-local residents on day trips; (2) Non-local OVN-NF – non-local residents staying overnight on the NF; (3) Non-local OVN – non-local residents staying overnight off the NF; (4) Local day trips – local residents on day trips; (5) Local OVN-NF – local residents staying overnight on the NF; (6) Local-OVN – local residents staying overnight off the NF; and (7) Non-Primary – visits where recreation on the NF is not the primary trip purpose. The average expenditure profiles by segment for the low category are presented in Table 1. Sixty percent of visits to the Hoosier NF were in the local day visit segment (Table 2). The number of visits to the MMYC in this segment category are likely proportionately higher given its location and activities available. The average number of people per vehicle visiting the Hoosier NF was 2.4 (Table 2).

Table 1.—Low spending profiles by segment and spending category, $ per party per trip. Source: Stynes and White (2005: Table 8). OVN-NF – overnight on national forest, OVN – overnight not on national forest.

	Non-local segments			Local segments				
Spending category	Day	OVN-NF	OVN	Day	OVN-NF	OVN	Non-primary	All visits
Lodging	0.00	13.56	41.71	0.00	10.03	12.27	30.81	12.49
Restaurant	11.72	14.91	42.08	5.75	9.18	16.68	27.25	16.25
Groceries	6.90	26.89	20.70	4.31	34.79	16.53	17.88	12.85
Gas & oil	12.63	29.26	29.29	11.84	24.28	22.41	19.71	18.47
Other transportation	0.43	1.12	5.68	0.21	0.00	0.32	5.94	1.73
Activities	3.27	3.04	9.65	1.77	2.46	9.19	1.61	4.03
Admissions/fees	4.30	7.52	7.25	3.54	8.50	8.01	3.15	5.17
Souvenirs/other	2.05	10.42	15.06	2.29	6.96	7.17	12.14	6.58
Total	41.29	106.72	171.42	29.71	96.20	92.59	118.48	77.56
SE Mean of Total	2.62	5.05	8.97	0.98	4.35	6.51	9.54	1.48

Table 2.—Estimates specific to Hoosier National Forest. Source: Stynes and White (2005: Tables A-2 and A-3).

	Non-local segments			Local segments				
Spending category	Day	OVN-NF	OVN	Day	OVN-NF	OVN	Non-primary	Total
Segment shares (%)	13	12	2	60	9	1	3	100
People per vehicle		2.9		2.4	1.7			2.4

Wildlife

The value assigned to wildlife is critical for evaluating tradeoffs among management alternatives impacting timber and wildlife. Values assigned to wildlife affect how available resources are allocated to the habitat for and management of wildlife generally and for target species (Berryman 1987). Managing for abundant deer for hunting, a major recreational use of the MMYC, involves significant tradeoffs with timber regeneration. A variety of forest successional stages increases the number of bird species, but lacking estimates of MMYC recreationists focused on observing a variety of species, it is difficult to evaluate this component.

Benefit-cost analysis, requiring estimates of dollar values, is commonly used to evaluate wildlife management tradeoffs (Stevens et al. 1991). Values most commonly used are use values and existence values. Use value is derived from hunting, fishing, and viewing activities that place the recreationist on site. Existence value accrues to both users and non-users when the value is attributable to those receiving value from the mere existence of wildlife. The primary source of use value estimates is the National Survey of Fishing, Hunting, and Wildlife-Associated Recreation conducted every 5 years by the U.S. Department of Commerce in cooperation with the U.S. Fish and Wildlife Service (U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, Census Bureau 2008).

Given the absence of a market, existence value must be measured indirectly, typically using a contingent valuation method (CVM). Weisbrod (1964) first developed the notion that non-users of wildlife may still get value from it and would be willing to pay an option price to retain the possibility of using the resource in the future. Krutilla (1967) introduced the theory that non-users may value a natural resource they have no desire to use, giving it an existence value. An existence value may also derive from a wish to endow future generations; this value is an altruistic value from knowing others can enjoy the resource, while others place an intrinsic value on a resource.

The 2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, Census Bureau 2008) estimated that 3.1 million U.S. residents fished, hunted, and watched wildlife in Indiana (Table 3). The total overestimates the number of individuals because some carry out more than one activity. Expenditures per hectare for the state can be obtained by applying these estimates to the area of land in Indiana suitable for each activity. The per-hectare values can then be applied to the area of the MMYC to get rough estimates of expenditures attributable at least in part to activities in the MMYC.

Table 3.—Activities in Indiana by U. S. residents. Source: 2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, Census Bureau 2008) and extrapolations by author. TEENR = trip and equipment expenditures by non-residents.

	Fishing	Hunting	Wildlife watching	Total
Participants	768,000	272,000	2,042,000	1,795,000
Days	9,805,000	4,808,000	24,013,000	38,626,000
Trip related expenditures	$242,624,000	$65,550,000	$143,615,000	$451,792,000
Equipment expenditures	$384,543,000	$157,470,000	$790,305,000	$1,335,318,000
TEENR	$120,464,000	$31,639,000	$45,511,000	$197,614,000
Estimates by author				
$/ha of water or land	$444.31	$3.27	$25.19	$28.40 for land
MMYC area	$66,650	$64,000	$495,800	$626,600

The 2007 National Resources Inventory (NRI) (U.S. Department of Agriculture 2009) estimated that there were 8,264,102 ha of non-federal rural land, plus 196,506 ha of federal land in Indiana. Because most of the federal land is open to hunting, the total private and federal land theoretically available for hunting is 8,460,607 ha. Assuming that one-half of this land is actually hunted, the area hunted would be 4,230,304 ha. The 2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation estimates that resident and nonresident hunters spent $254,662,000 for travel and equipment in 2006. This makes the expenditure per ha for hunting $60.20 ($254,662,000/4,230,000 ha). Based on the 13,837 ha used for timber production on the MMSF, the hunting value would be $833,000.

The same survey estimated that resident and nonresident wildlife watchers spent $979,413,000 in Indiana in 2006. Assuming wildlife watching is possible on all 8,264,102 ha of rural land plus on a portion of urban land, the expenditure per ha is $119 ($979,431,000/8,264,102). If this expenditure is applied to the entire 16,279 ha of the MMYC, the wildlife-watching value would be $2,269,824. This value would capture a majority of the day hikers using the MMYC.

The 2007 NRI (U.S. Department of Agriculture 2009) estimated that there are 155,200 ha of water in Indiana. Assuming that 50 percent is available for fishing, the hectares fished would be 77,600 ha. The Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, Census Bureau 2008: 4) estimated that fishing-related expenditures by residents and nonresidents totaled $747,631,000. This gives a per-hectare value of $9,634 for water fished ($747,631,000/77,600 ha). This assumption would make the value of the 61 ha of lakes in the MMYC $587,701. Multipliers for determining total economic impact of wildlife are needed to make these values comparable to the estimated impact of timber revenue.

Ritz and Ready (n.d.) conducted a comprehensive analysis of the marginal cost and benefit of a deer in Pennsylvania. Costs included Lyme disease; deer-vehicle collisions; damage to landscaping and nurseries; crop damage; and damage to forests, the timber industry, and ecosystems. They estimated that 1 out of 12 state residents takes out a hunting license. They estimated marginal benefits for a one-deer increase in the total deer population from hunting of $91 to $107 per deer and from viewing of $7 to $116; total costs ranged from $150 to $232. The implication is that the size of the deer herd should not be increased, because at the high end of the estimates costs exceed benefits. At the low end of the estimates, the benefits from adding an additional deer outweigh the costs.

The IDNR-Division of Fish and Wildlife reports the deer kill by county (Stewart 2010). The deer harvest was 1,480 and 1,968 in Monroe and Brown Counties, respectively. Reports by MMYC property personnel of nearly bumper-to-bumper parking on MMYC roads during hunting season indicate that state forest land is hunted more heavily than private land because no permission is needed to hunt on State Forests and the area of private land open for hunting has declined, in part due to leasing. Monroe County is 1,020 km^2 and Brown County is 821 km^2. The MMYC constitutes about 10 percent of the land area of Brown and Monroe Counties combined. It constitutes about 15 percent of the forested area of these counties. Applying the forest cover percentages for each county to the deer take yields a deer take in the MMYC of 400 to 500 per year.

Contingent Valuation

The most likely alternative for measuring non-market values is contingent valuation. Stevens et al. (1991) developed a utility function to test the validity of the contingent valuation method for estimating the existence value of four species introduced or reintroduced to New England—bald eagle (*Haliaeetus leucocephalus*), Atlantic salmon

(*Salmo salar*), wild turkey (*Meleagris gallopavo*), and coyote (*Canis latrans*). They found that their results were consistent with previous work, but noted the frequency with which respondents to their survey instrument expressed moral beliefs which are not measurable in monetary terms. They adopted a general form of a utility function that includes an individual's own consumption of a bundle of private goods, knowledge of others' use, personal existence value, and knowledge that others derive satisfaction from knowing of the existence of the resource. These independent variables fall into three categories: (1) personal use values, (2) use by others, and (3) non-use value. Economists' main concern is the possibility of bias in responses and the extent to which economic theory applies.

Harris et al. (1998) questioned the decisionmaking process of respondents to surveys. Specifically, they questioned whether respondents consider the price of substitutes available in the market or non-market, whether income constrains responses, whether the process of responding to questions creates a stressful circumstance, whether respondents are familiar with the resource in question, or whether any possible decision is inconsequential to a respondent. Finally, a respondent may consider a resource priceless because of spiritual or moral rights he or she assigns to the resource. A further complication noted by Mitchell and Carson (1989) is that wildlife existence is a public good for which the supply cannot be explained by neoclassical economics. Stevens et al. (1991) found that 79 percent of respondents to their salmon survey agreed with the statement "All species of wildlife have a right to exist independent of any benefit or harm to people." Seventy percent of respondents gave this statement as one of the three most important reasons for the existence of bald eagles, wild turkeys, and coyotes.

Stevens et al. (1991) found a positive relationship between respondents' level of formal education and willingness to pay for bald eagles, wild turkeys, and coyotes. They also found a positive relationship between respondents' expectation that they would fish for Atlantic salmon in the future and their willingness to pay for salmon restoration. The option value and willingness to pay for coyote were the lowest of the four species. Their estimated equivalent surplus willingness to pay was $7.93 for Atlantic salmon. Boyle and Bishop (1987) estimated $1.00 to $5.00 for striped shiner (*Luxilus chrysocephalus*), a non-game fish. Kay et al. (1987) estimated Atlantic salmon existence value of $10 to $30 above the willingness to pay for fishing licenses. Only 6 percent of respondents to the salmon survey selected either a current or future use as one of the reasons for preservation of this species. Only 12 percent of those responding to the surveys for the other three species selected either current or future use as reason for preservation in New England. Stevens et al. (1991) estimated bald eagle equivalent surplus of $19.28 per person per year; estimates within the $10.62 and $75.31 range were reported in other studies they cited.

Research

Though difficult to value in dollar terms, the use of the MMYC for research is arguably its greatest value to the nation. Its location less than 25 km north of Bloomington, IN, home of Indiana University, has led to extensive use for research. Researchers from Ball State University and Purdue University, among others, have conducted research here. The complex provides unique opportunities to conduct research because of the large areas of contiguous forest cover.

Terrestrial Data

The MMYC plays a critical role in the study of the biochemical systems of deciduous forests using data from NASA's "moderate resolution imaging spectroradiometer" (MODIS) satellite imagery focused on representative vegetative sites throughout the world. The MODIS data are correlated with complex ground-level data collected from AmeriFlux towers, one of which is located on the MMYC. "The core of the MMYC AmeriFlux project is a joint assessment

of the role of this forest ecosystem as a net carbon sink by micrometeorological (eddy-flux, EC[3]) and biometric (carbon-pool increments) methods. The site regularly collects data about CO_2 exchange between the atmosphere and the forest, as well as many other variables necessary for understanding the carbon dynamics in deciduous forests" (Indiana University 2009: homepage).

Data from the MMYC are compared with those of other MODIS-AmeriFlux sites to assess the role of forests globally to sequester carbon. Ground, vegetation layers, air composition, and meteorological data are collected at these sites and correlated with spectral signatures to provide the relationships required to draw conclusions on carbon flux and other critical environmental factors determined primarily by vegetative cover. The ability to access these data has led to millions of dollars in research grants to Indiana universities.

A brief search of the academic literature revealed that more than 100 scientific journal articles have resulted from research projects using data from the MMYC. Journal articles started to appear 6 years after the founding of MMYSF in 1929.[4] Research reported includes remote sensing and carbon flux modeling, extent of and control of invasive species, wildlife, forest regeneration, entomology, landscape modeling, botany, mensuration, and soils.

Timber Sales

The MMYC primarily supports the wood products industry within 58 km of the center of the forest. This estimate is based on the 41 percent of the volume sold from 1993 to 2010 for which the buyer's city was recorded. The longest haul was 172 km for a white oak (*Quercus alba*) veneer sale. Hoover and Preston (2010) reported that statewide the average haul distance is about 80 km. In 2005 sawlog production of all species in Brown, Monroe, and Morgan Counties was 33.2 million board feet (Piva and Gallion 2007). The volume sold from the MMYC forests in 2005 was 1.6 million board feet. Thus, the MMYC accounted for 4.8 percent of total sawlog production in these counties in 2005. The gross annual economic impact of timber sales for the 17-year period from 1993 to 2010 averaged $94.9 million (Table 4). This total assumes a gross economic multiplier of $51 per board foot (Hoover and Settle 2010).

ECONOMICS OF SILVICULTURAL SYSTEMS

Forest management regimes are selected primarily by owners' objectives and the evaluation of tradeoffs. The MMYC is heavily used for recreation and research, as

Table 4.—Economic impact of timber volume sold from MMYC State Forests, 1993 to 2010.

Year	Volume sold (MBF)	Gross economic impact
1993	1,340	$68,330,259
1994	1,490	$75,983,115
1995	1,809	$92,246,658
1996	875	$44,608,323
1997	1,390	$70,898,517
1998	1,114	$56,803,188
1999	1,184	$60,377,880
2000	545	$27,795,255
2001	545	$27,795,255
2002	1,081	$55,120,494
2003	1,182	$60,303,624
2004	1,461	$74,514,570
2005	1,794	$91,498,794
2006	5,345	$272,597,958
2007	2,771	$141,301,773
2008	4,453	$227,080,152
2009	3,012	$153,602,106
2010	2,089	$106,515,642
Total	33,480	$1,707,373,563

[3] Eddy covariance

[4] Duncan, W. 1935. Root systems of woody plants of old fields of Indiana. Ecology. 16(4): 554-567.

well as for generating revenue from the sale of timber. The statutory guidelines for management of State Forests provide that:

> Sec. 1. (a) It is the public policy of Indiana to protect and conserve the timber, water resources, wildlife, and topsoil in the forests owned and operated by the division of forestry for the equal enjoyment and guaranteed use of future generations. However, by the employment of good husbandry, timber that has a substantial commercial value may be removed in a manner that benefits the growth of saplings and other trees by thinnings, improvement cutting, and harvest processes and at the same time provide a source of revenue to the state and counties and provide local markets with a further source of building material (Indiana Code 2011).

The selection of silvicultural systems involves significant tradeoffs among timber revenue, biological diversity, and forest users' perception of aesthetic value. The tradeoffs between oak-hickory (*Quercus-Carya*) and maple-beech-birch (*Acer-Fagus-Betula*) forest types are a major theme of the HEE project. These tradeoffs are discussed by Jenkins (this publication).

Composition of Indiana's Timberland

The loss in timber value resulting from the movement away from oak-hickory may be used as justification for creating disturbances sufficient to regenerate oak and associated species. Indiana's primary and secondary forest products industry developed in large part based on the availability of high-quality timber of oak species and black walnut (*Juglans nigra*). Many other species are used, however.

Forest-type groups defined by the USFS were used for analysis of changes in forest type. These are aggregates of forest cover types (Eyre 1980). The oak-hickory forest type group includes 20 forest cover types. The maple-beech-birch forest type group includes four forest cover types (USFS 2007). In 2010 these two forest types accounted for 80 percent of the timberland in Indiana (USFS 2011).

Oak species as a percent of the statewide net tree volume on timberland classified in the oak-hickory forest type decreased from 45.5 percent in 1986 to 30.9 percent in 2010, while hard maple increased from 4.9 percent to 8.4 percent (Fig. 2). The change for the volume of oak species classified in the maple-beech-birch forest type was from 17.7 to 7.5 percent of total volume, while hard maple increased from 22.6 percent to 58.6 percent (Fig. 3). While the percentage of oak species in both forest types declined from 1986 to 2010, the area of timberland classified as oak-hickory forest type increased by 544,989 ha; the area classified as maple-beech-birch decreased by 266,453 ha (Fig. 4). The increase in the area of oak-hickory forest type group is due to the broad range of forest cover types included. In addition, timber harvesting practices on unmanaged timberland are dominated by harvesting of all merchantable timber down to pulpwood size. This activity creates sufficient disturbance to regenerate intolerant species.

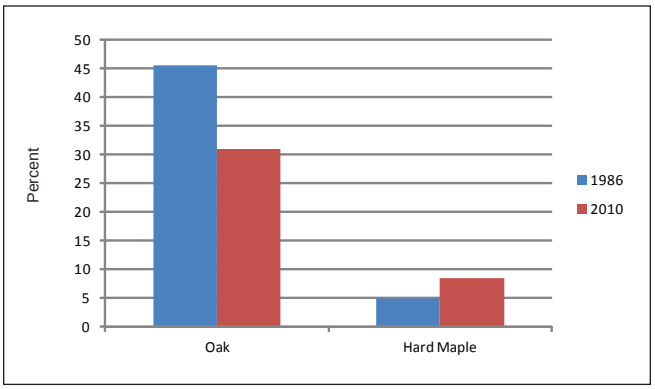


Figure 2.—Oak and hard maple components of oak/hickory forest type group in 1986 and 2010.

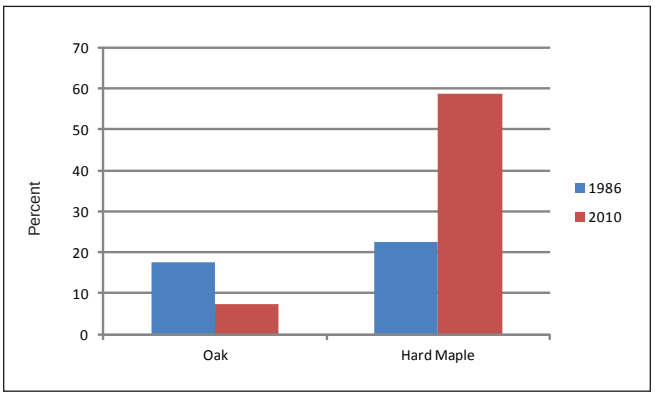


Figure 3.—Oak and hard maple components of maple/beech/birch forest group type in 1986 and 2010.

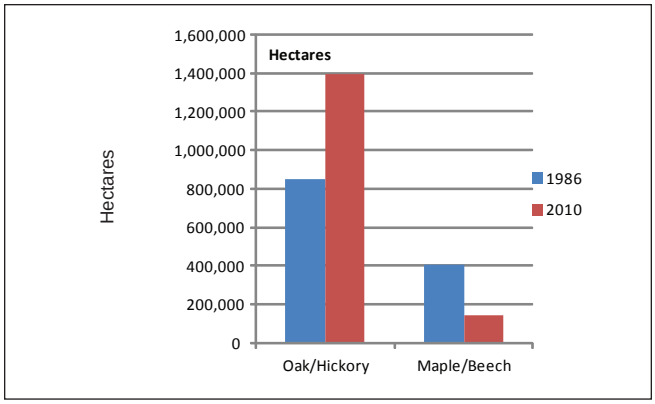


Figure 4.—Hectares of timberland in oak/hickory and maple/beech/birch forest group types.

Timber Value by Forest Type

An indication of the overall economic impact of the change from oak-hickory to maple-beech-birch forest types is a comparison of the historical value of the species mixes for these forest types. The FIA Forest Inventory Data Online calculator for the oak-hickory (400) forest type, and the maple-beech-birch (800) forest type were used to obtain the species mix of these forest types. The species distribution for 1986, the earliest available, was used for the 1957 to 1986 period. From 1986 to 2010 the species weights were based on a straight-line increment between 1986 and 2010. The price data by species and log grade came from the Indiana Forest Products Price Report (Hoover and Preston annual). These reports delivered log prices for 4 grades of sawlogs for 15 species, and 2 grades of veneer logs by log diameter categories using diameter inside bark in inches for 6 species. The average log values used to calculate the price index for an average stand were used; only the species distribution was changed. The values of prime red oak (*Q. rubra*), white oak, and hard maple sawlogs (Fig. 5) follow the same pattern as that for the two forest types (Fig. 6). Stumpage values were not estimated for this analysis because the focus is on the value difference between the two forest types.

The values of these two forest types did not separate until 1974 (Fig. 6), when the rate of inflation increased substantially. That year is also about the time hardwood lumber production cycled up with peak production in 1999 and a decline thereafter (Hoover and Preston 1999). The amount by which the value of oak-hickory forest types exceeded the maple-beech-birch forest type peaked in 1989 at $56 per thousand board feet (MBF). The value of oak-hickory was greater than maple-beech-birch until 2000, bottoming out in 2006 at $125 per MBF (Fig. 7). From 1957 to 1986, the value difference was due to changes in log prices. Thereafter the change was due to both price fluctuations and trends, and species composition changes.

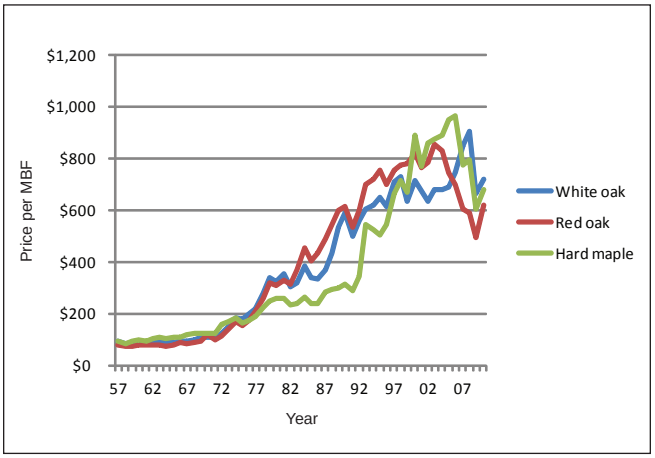


Figure 5.—Price per MBF of prime sawlogs delivered to mills in Indiana, 1957 to 2010. Source: Hoover and Preston (annual).

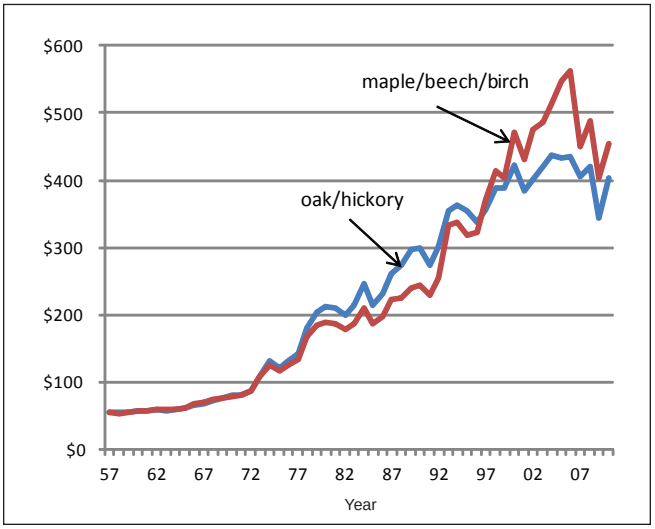


Figure 6.—Value of timber for average stand for oak/hickory and maple/beech/birch forest types.

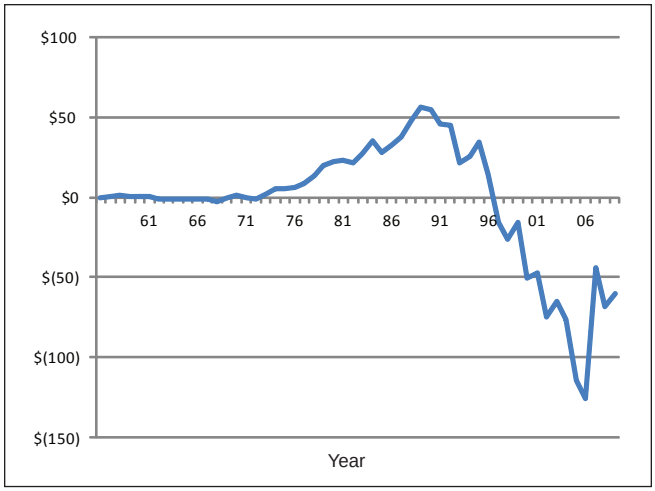


Figure 7.—Price differential between timber value of an average stand classified as oak/hickory and maple/beech/birch forest types.

Relative Price of Oak and Hard Maple

Decisions made today regarding managing for oak-hickory forest types or allowing natural succession are driven by owners' objectives. Long-term relative timber values are most likely not a significant factor for a majority of owners. Butler's (2011) National Woodland Owner Survey Table Maker estimates that the owners of 60 percent of family-owned forest land have "minimal activity to maintain" forest land as an objective. However, the same source estimates that 74 percent of the land is held by owners expecting to "harvest sawlogs or pulpwood" in the future. Thus, it is doubtful that family forest owners seek to maintain oak-hickory forest types. The management objective for state forest land is the maximization of net benefits to society, including biological diversity. Thus, relative timber values inform decisionmakers regarding the net financial benefit of managing to maintain oak-hickory forest types.

The higher value of maple-beech-birch forest types compared to oak-hickory determined for this analysis reflects state-wide average species compositions. This is a recent trend and may change back to equal or higher value for oak-hickory as the supply of oak species decreases due to harvest levels in excess of growth (Woodall et al. 2005), as well as natural succession to maple. As long as the market value of hard maple exceeds that of oaks, there is no financial incentive for private forest owners to manage for oak-hickory.

Oak removals exceeding growth should eventually reverse relative prices unless there is a significant and long-term decline in oak demand. On the other hand, increasing average temperatures will favor oak-hickory while maple decline continues to impact maple volumes in both urban and forest sites (Bauce and Allan 1991, Houston 1999). Considering these numerous interacting factors, this analysis assumes that the relative oak and maple prices will return to the long-run pattern of parity.

Capital Value of Silvicultural Systems

The potential reduction in stumpage value resulting from a change from the oak-hickory forest type group to the maple-beech-birch forest type group does not necessarily mean a reduction in the return on investment when natural succession occurs. Both market and non-market benefits should be considered if the total return to society is the management objective. Management of the MMYC requires balancing the tradeoffs between market and non-market benefits because of uncompensated recreational use. The focus in this section is traditional measures of investment returns from timber production.

The standard approach to evaluating alternative timber management regimes is estimation of the year zero net present value (NPV_0) for each alternative. If the land is to be permanently managed for timber, the appropriate measure is the net present value of a perpetual series of periodic payments (Faustmann 1849), as given in equation (1). The numerator is NPV as of the end of a rotation. The denominator is the multiplier for the year zero present value of a perpetual series of constant NPVs as of the end of the period:

$$LEV = \sum_{t=0}^{n}\left[\left(R_t - E_t\right)\left(1+i\right)^t\right] \Big/ \left[\left(1+i\right)^n - 1\right] \quad (1)$$

where R = revenues, E = expenses, i = nominal discount rate, t = index on years, and n = rotation length. This measure is referred to as soil/land expectation value (SEV/LEV). It reflects the amount that could be paid for bare land if the land is used to generate the net revenues assumed. Although formulated for a continuous series of even-aged rotations, it is applicable to uneven-aged regimes by substituting cutting cycles for rotations (Straka and Bullard 1996). This application requires assuming a continuous series of harvests of equal periodicity, net revenue, and costs, as shown in equation (2) (Straka and Bullard 1996, equation (h)). Equation 2 reflects the amount that could be paid for the land and the level of stocking needed to generate the periodic net revenue:

$$LTEV = \left[NTR - \left(a\left(1+i\right)^{C} - 1 \right) / i \right] \Big/ \left[\left(1+i\right)^{C} - 1 \right] \quad (2)$$

where NTR = net timber revenue received every c years, a = annual management and tax costs, c = number of years in the cutting cycle, and i = nominal discount rate. Given that NTR in equation (2) can be generated only with the land and a base level of timber stocking, "land expectation value" is not an appropriate term. Countryman and Miller (1989) refer to this measure as land and timber expectation value (LTEV).

Countryman and Miller (1989) estimated the investment returns for upland oak stands with sugar maple (*A. saccharum*) understory compared to stands that succeed to sugar maple. At the time of their analysis in 1989 oak stumpage prices were at least 80 percent higher than hard maple, based on the 1989 Indiana Timber Price Report (Hoover 1989). They assumed revenues per ha of $648 (low), $7,770 (medium), and $23,310 (high) for the final harvest of oak clearcutting systems. The comparable values for upland oak stands succeeding to hard maple uneven-aged stands were $130, $389, and $1,554 per ha on a perpetual 10-year cutting cycle. They also modeled an oak shelterwood system. The comparable values for the final cut were $433, $5,180, and $15,540 per ha. They concluded that (1) managing to shorten rotation length increases LTEV because of the end-loading of revenues and front-loading of costs; (2) for higher discount rates, minimizing costs contributes more to LTEV than does a higher-quality stand; (3) in stands with medium to high initial stand values, sugar maple has the highest LTEV because of higher costs associated with oak alternatives; (4) in stands with low initial stand values, oak may have a higher LTEV because these stands generally have low site indexes; and (5) when existing oak stands can be treated before they reach economic maturity, oak LTEV may exceed hard maple LTEV because stand value growth exceeds carrying costs.

The literature lacks clarity regarding the financial analysis of converting among silvicultural systems. Countryman and Miller (1989) assumed that for existing stands with adequate stocking for conversion to uneven-age management, the one-time revenues and costs for conversion were added to the LTEV. For stands with inadequate stocking they added the revenue from the initial clearcutting to the LTEV. These one-time revenues increase LTEV above that for a perpetual series of NPVs. Similarly, Chang (1981) and Rideout (1985) used the term "manage forest value" for the NPV of an uneven-aged stand in its steady state. They added to this amount the NPV of the conversion-period cash flows.

Guldin's (1996) approach to dealing with the analysis of conversions among southern conifer systems is keyed to the landowner's objectives. In the case of landowners not willing to liquidate the residual stand required to perpetuate uneven-aged systems, the liquidation value is not carried forward. In other words, the value of the residual stocking that must be carried in perpetuity is not considered as a capital cost, unlike the Countryman and Miller (1989) approach. Not including the capital value of residual stocking would be justified when private landowners receive non-monetized benefits from the continuous presence of stocking.

Silvicultural Model Assumptions

The primary purpose of this analysis is to compare alternative silvicultural systems used in Indiana generally, and the MMYC in particular. These models are relevant only for making per-hectare comparisons. A given model is not relevant to the value of the MMYC as a whole because a mix of silvicultural systems is used.

Even-aged: LEV as presented in equation (1) was used with all costs and revenues compounded to the end of an assumed 80-year rotation.

Shelterwood: Straka and Bullard (1996) do not adapt the LEV model for shelterwood. Countryman and Miller (1989) analyzed shelterwood models by applying equation (1) to each individual revenue and expense with a final shelterwood cut in year 60. This approach is equivalent to modeling a shelterwood system over an 80-year period and applying equation (1) to NPV_{80}.

Single-tree selection model: The formulation of Straka and Bullard (1996) for determination of LTEV is based on a fixed cutting cycle with all revenues and costs replicated in perpetuity. Equation (1) is applied to the NPV at the end of the cycle. This is also equivalent to modeling a shelterwood system over an 80-year period and applying equation (1) to NPV_{80}.

Analysis

Estimates of NPV_0 were made for even-aged, shelterwood, and single-tree selection silvicultural systems using Microsoft Excel (Microsoft Corporation, Redmond, WA). I reported LEV for even-aged systems and LTEV for shelterwood and single-tree selection systems. The spreadsheet was formulated to allow sensitivity analyses for all variables. The measures used to compare systems were NPV_0 and equation (1) applied to NPV_{80}. Assuming an 80-year rotation or cycle makes the specific model used less important because of the small contribution to NPV of revenues and expenses past 60 years and the focus on comparison of LTEVs. The impact does vary, however, by the interest rate used. For example, for the 3-percent discount rate used in this analysis $1 in year 80 is worth $0.09 in year zero.

Sets of low, medium, and high variables were assumed for productivity, stumpage price, and expenses for each system. Current levels of revenues and expenses were assumed. An adjustment for an overall rate of inflation was incorporated into the calculations, but an increase by an assumed rate of inflation does not change LTEV because the nominal discount rate used, i, must also be adjusted for inflation, i.e., $i = r + f + r \times f$, where i = nominal discount rate, r = real discount rate, and f = overall rate of inflation.

The option for a linear and/or exponential real price increase for stumpage was incorporated. This real price increase does increase LTEV. The low set of assumptions was at the low end of the mean annual increment (MAI) range, low stumpage prices, and low expenses. The high set was at the high end for these variables. The MAIs assumed for all three systems were 518, 648, and 906 board feet per ha per year for low, medium, and high, respectively.

The DoF does not pay federal or state income, sales, or other taxes. However, it does make payments to the counties in which timber sales are made in lieu of tax. The payments are 15 percent of the net revenue from sales. The DoF records costs of labor for marking, paperwork, advertising, showing, access roads, and travel for each sale. Net revenue was assumed to be gross revenue minus an average direct cost of sales. The average cost of sales for the MMYC from 1994 to 2010 was about 8 percent (STDEV = 11.1) of the sales price.

Base Model: The base assumptions for the three silvicultural models were annual costs of $12.79, $25.59, and $38.38 per ha for the low, medium, and high levels. Linear real price increases for stumpage of $1, $2, and $3 per MBF were assumed. The average annual price increase for the trend line for real prices reported by Hoover and Preston (2011) is $2 per MBF. The discount rate used was 3 percent. The other base assumptions for the silvicultural model are presented in Tables 5, 6, and 7.

Table 5.—Even-aged volume harvested, unit price, and TSI expense.

	Low			Medium			High		
Year	Harvest (MBF per ha)	Value per MBF	TSI per ha	Harvest (MBF per ha)	Value per MBF	TSI per ha	Harvest (MBF per ha)	Value per MBF	TSI per ha
0	15.54	$300		20.72	$400		23.31	$500	
1			$259			$648			$1,036
2			$259			$518			$1,036
5			$518			$1,036			$1,554
10			$130			$130			$518
20			$130			$130			$518
30			$130			$130			$518
40	7.77	$150		10.36	$200		14.25	$250	
60	10.36	$200		12.95	$300		16.84	$400	
70				12.95	$350		16.84	$500	
80	23.31	$300		20.72	$400		24.61	$500	

Table 6.—Shelterwood volume harvested, unit price, and TSI expense.

	Low			Medium			High		
Year	Harvest (MBF per ha)	Value per MBF	TSI per ha	Harvest (MBF per ha)	Value per MBF	TSI per ha	Harvest (MBF per ha)	Value per MBF	TSI per ha
0	10.36	$350		15.54	$450		23.31	$550	
2			$518			$777			$1,036
5	2.59	$350		5.18	$450		5.18	$550	
15			$259			$259			$518
25			$259			$259			$518
35	2.59	$300		2.59	$350		5.18	$400	
65	7.77	$300		7.77	$350		14.25	$400	
80	18.13	$400		20.72	$450		24.61	$550	

Table 7.—Single tree selection volume harvested, unit price, and TSI expense.

	Low			Medium			High		
Year	Harvest (MBF per ha)	Value per MBF	TSI per ha	Harvest (MBF per ha)	Value per MBF	TSI per ha	Harvest (MBF per ha)	Value per MBF	TSI per ha
0	7.77	$300		10.36	$425		15.54	$500	
2			$130			$259			$518
15	6.48	$300		7.77	$425		12.95	$500	
17			$130			$259			$518
30	6.48	$325		7.77	$450		12.95	$550	
32			$130			$259			$518
45	5.18	$350		6.48	$500		7.77	$600	
47			$130			$259			$518
60	5.18	$350		6.48	$500		7.77	$600	
62			$130			$259			$518
70	5.18	$350		6.48	$500		7.77	$600	
72			$130			$259			$518
80	5.18	$350		2.5	$500		7.77	$600	

Results

The LEVs (LTEVs) ranged from -$13 per ha for the even-aged model with the low set of assumptions to $7,439 for the single-tree selection model with the high set of assumptions (Table 8). Revenue from the initial harvests for all three treatments was not included in NPV_0 and LTEV. The LTEVs with the initial harvest included can be calculated simply by adding the initial harvest values to the LTEV shown in Table 8. The differences between the low single-tree selection and even-aged LTEV can be interpreted to mean that application of even-aged silviculture reduces the value of forest land by $2,503 to $7,017 per ha. This reduction occurs because of the long waiting period for the intermediate and final harvests with even-aged management.

Table 8.—Land and timber expected values per hectare for base models.

Model	Low	Medium	High
Even-aged			
Initial harvest	$3,823	$6,179	$8,689
NPV_0	($12)	$8	$382
LEV	($13)	$9	$422
Shelterwood			
Initial harvest	$2,703	$5,213	$9,556
NPV_0	$935	$1,627	$2,933
LTEV	$1,032	$1,795	$3,237
Single-tree selection			
Initial harvest	$1,683	$3,282	$5,793
NPV_0	$2,256	$4,260	$6,740
LTEV	$2,490	$4,702	$7,439
Allowable cut			
LTEV	$4,107	$5,857	$7,608
LTEV including recreation	$4,930	$7,216	$9,502

The responsiveness of the models to changes in the specified variables was evaluated by comparing LEV (LTEV) for the base cases with the LEV (LTEV) for a change in one of the variables. The results are presented as elasticities, i.e., the percent change in LEV (LTEV) for a 1-percent change in one of the specified variables, *ceteris paribus* (Tables 9, 10, and 11). All variables except timber stand improvement (TSI) costs were elastic for the even-aged model. As expected, the only elastic variables in the two uneven-aged models were MAI and discount rate.

Table 9.—Sensitivity analysis for even-aged model. LISP = linear increase in stumpage price. EISP = exponential increase in stumpage price.

	Low	Medium	High
LISP	2.46	2.62	2.10
EISP	1.79	1.88	1.74
Discount rate	-10.35	-13.62	-10.03
MAI	8.49	10.34	12.39
Annual costs	-0.70	-0.94	-0.47
TSI costs	-5.74	-7.38	-3.02

Table 10.—Sensitivity analysis for shelterwood model. See Table 9 for abbreviations used.

	Low	Medium	High
LISP	0.18	0.24	0.33
EISP	0.81	0.67	0.93
Discount rate	-2.12	-1.59	-1.69
MAI	2.12	3.53	1.69
Annual costs	-0.14	-0.15	-0.12
TSI costs	-0.66	-0.56	-0.40

Table 11.—Sensitivity analysis for single-tree selection model. See Table 9 for abbreviations used.

	Low	Medium	High
LISP	0.10	0.22	0.19
EISP	0.37	0.46	0.45
Discount rate	-1.22	-1.26	-1.25
MAI	1.17	1.19	1.27
Annual costs	-0.06	-0.06	-0.10
TSI costs	-0.12	-0.12	-0.17

Capital Value Based on Allowable Cut

In addition to the per-hectare values for the three silvicultural treatments, an average value per hectare for the 6,772 ha of the MMYC under timber management was estimated using the current total allowable cut of 5,000 MBF per year assuming that 85 percent of the total area is available for timber production.[5] The capital values (LEV and LTEV) for the three silvicultural treatments should be used for making comparisons for alternative treatments for specific stands, or a mix of treatments to meet wildlife, recreational, and other values.

LTEVs based on the current allowable cut should consider the extent to which even-aged management is currently used. This consideration is important because a substantial shift to even-aged management would reduce the allowable cut in the immediate future. The application of even-aged management is indicated by the size of openings marked for sale over the last 18 years as shown in the detailed timber sale records. Two hundred twenty-three timber sales were conducted on the MMYC from 1993 through May 2010. Average tract size treated was 39.1 ha with timber harvested on 31.6 ha. The total area of openings marked averaged 1.9 ha per sale. Individual openings did not exceed 3.8 ha. Most of these were group-selection cuts. The price per MBF received for the 253 sales from January 1993 to May 2010 averaged $257.00 (STDEV $223.48) and ranged from $2,819.00 for a white oak veneer sale to $19.16 for a commercial TSI.

The average per-hectare capital value as measured by LTEV for the 6,772 ha under timber management was estimated for three sets of assumptions—low, medium and high. The annual harvest volume was 5,000 MBF for all three sets. The specified price received per MBF was $200, $280, and $360. A linear annual increase of $1, $2, and $3 per MBF was specified. An exponential real price increase of zero was specified for the base model. The specified annual management cost per hectare was $5.18, $10.36, and $15.54. Per-hectare LTEVs with annual recreation values of $10.00, $16.50, and $23.00 were estimated for the base model. The discount rate was 3 percent for all three sets. Per-hectare LTEVs obtained for the low, medium, and high models were $4,107, $5,857, and $7,608 respectively (Table 8). Inclusion of annual per-hectare recreation values of $10.00, $16.50, and $23.00 increased the base model LTEVs to $4,930, $7,216, and $9,502.

Sensitivity analyses were conducted by estimating elasticities for a doubling of the base model specifications *ceteris paribus* (Table 12) except for the discount rate. It was increased to 4 percent from the base model specification of 3 percent. Sensitivity to an exponential increase in price was evaluated for the change from 1-percent simple compound interest per year to 2 percent. Gross revenue was reduced by 8.07 percent for the cost of sales. Net revenue was reduced by 5 percent for the in-lieu tax payments to counties.

The increases in LTEV with an estimated value for recreation included were made assuming $24.00, $39.67, and $55.29 per ha per year for low, medium, and high models. The estimate for the medium set of assumptions was based on $41.29 per day-use visit (Table 1), 45 visits per weekend day, and 97 weekend days per year. These assumptions give a total of $180,850 per year and $11.11 per ha per year. This result was increased by $3.27 per ha per year for hunting and $25.19 for bird watching (Table 3). The latter value would account for hiking as most hikers engage in bird watching.

Table 12.—Sensitivity analysis for allowable cut model. See Table 9 for abbreviations used.

	Low	Medium	High
LISP	0.176	0.241	0.275
EISP	0.537	0.557	0.568
Discount rate	-0.454	-0.465	-0.470
Annual cost	-0.123	-0.174	-0.204
Recreation	0.212	0.237	0.251

[5] Personal communications with Jim Allen, MMYC property manager, 30 September 2011

SUMMARY

The MMYC exemplifies public forest land with stakeholders holding a wide range of management expectations. The need to be at least partially self-supporting financially, requiring sale of timber, adds a dimension similar to industrial forest land. Management strategies to meet the broad range of stakeholder expectations require the application of sound science appropriate for the MMYC. Information about how the forest is used and valued for non-timber purposes is also needed, especially in light of increased population density within typical day-use travel distances of the MMYC.

The information needed to evaluate tradeoffs between market and non-market values presents a particular challenge. Much of this challenge comes from a lack of information on the extent to which the MMYC is used for recreation, gathering of non-timber forest products, and scenic drives, and its role in defining the regional culture. Ongoing controversies about how the forest should be managed clearly indicate the significance of non-timber values among vocal stakeholders. The values held by a larger segment of stakeholders need to be quantified, but the results of the work by Rogers et al. (this publication) indicate a loss of non-market value when forest stocking in users' viewscapes is reduced.

Surveys of desired OR opportunities closely match those provided by the MMYC. Deer hunting is of particular significance. The mast crops of the oak-hickory forest type benefit the deer population, but data are lacking on the potential decline as the forest type moves toward maple-beech-birch. By extrapolation of the limited data available an estimated 400 to 500 deer are taken annually on the MMYC. The annual hunting value of the MMYC is in the range of $833,000. The wildlife viewing value is in the $2.3 million range per annum. The value of the lakes and ponds is in the range of $588,000 per annum. The total value is approximately $3.7 million, but multipliers for determining total economic impact are needed to make these values comparable to the estimated impact of timber revenue.

The gross annual economic impact of timber sales for the 17-year period from 1993 to 2010 averaged $94.9 million per annum (Table 4), assuming a gross economic multiplier of $51 per board foot (Hoover and Settle 2010). The fair market value of the MMYC as a unit is not relevant, but the capital value resulting from management for timber production can be used to estimate the opportunity cost of holding the land for timber production. The year-zero value of land managed in perpetuity for timber production under the even-aged, shelterwood, and single-tree selection systems ranged from $156 per ha for the even-aged model with the low set of assumptions to $8,590 for the single-tree selection model with the high set of assumptions (Table 8). These results can be interpreted to mean that application of even-aged silviculture reduces the value of forest land when used only for timber production from $7,770 to $2,590 per ha. Future analyses should add estimates of the reduction in recreational value when even-aged management is used.

Additional data collection and analyses are needed to develop non-timber value estimates comparable to the estimated economic impact of timber production.

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CONCLUDING SYNTHESIS AND LOOKING FORWARD

SCIENCE IN THE HARDWOOD ECOSYSTEM EXPERIMENT: ACCOMPLISHMENTS AND THE ROAD AHEAD

Michael R. Saunders and Robert K. Swihart[1]

Abstract.—Over the next century, the Hardwood Ecosystem Experiment (HEE) in Indiana will provide numerous opportunities for collaborative research on how forest management affects the ecological, economic, and social resources of southern Indiana. Here, we highlight the pre-treatment research conducted at the HEE sites from 2006 through 2008 and discuss the role that pre-treatment differences among experimental units and/or treatments may have on future research activities. We then formulate hypotheses about the effects of the various silvicultural regimes on communities monitored for the duration of the study (i.e., woody plants, terrestrial salamanders, birds), focal communities and species (i.e., box turtles, timber rattlesnakes, cerulean warblers, moths, and wood-boring beetles), and public attitudes toward forest management. HEE researchers will face a number of key challenges, ranging from overcoming constraints in experimental design to maintaining scientific, managerial, and social relevance. However, each of these challenges provides opportunities for high-impact research and outreach that can improve the management of our Central Hardwood Forests for decades to come.

INTRODUCTION

The Hardwood Ecosystem Experiment (HEE) is a multi-faceted, 100-year project in south-central Indiana involving multiple universities, state natural resource agencies, and non-governmental organizations.
The overarching goal of the HEE is to improve the sustainability of forest resources and quality of life of Indiana residents by understanding ecosystem and human responses to forest management (Kalb and Mycroft, this publication). Fundamental to the HEE are ecological and social studies designed to address three key research objectives: (1) development of silvicultural systems for regenerating oak-dominated forests, (2) quantification of the responses of plants and animals to these systems, and (3) determination of the social and economic implications of these systems. This paper summarizes research findings from studies conducted during 2006-08 to establish baseline conditions during the pre-harvest period, considers predictions related to responses following the initial harvests in 2008-09, and identifies opportunities and challenges awaiting future HEE research.

STUDY AREA AND DESIGN

The location and design of the HEE are covered in detail by Kalb and Mycroft (this publication). Briefly, the HEE is located on Morgan-Monroe (9,710 ha) and Yellowwood (9,440 ha) State Forests (hereafter MMYSF) in south-central Indiana. Nine research core areas ($\bar{x} \pm$ s.d. = 89 ± 9 ha) serve as the sites to which one of three silvicultural treatments (uneven-aged, even-aged, or no harvest) was assigned randomly, subject to the constraint that each treatment was assigned to an equal number of (n=3) cores. Each research core was surrounded by a buffer area

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(311 ± 59 ha) to isolate the core areas from effects of ongoing management in other portions of the state forest. Each core and associated buffer constituted a management unit (400 ± 60 ha).

Initial harvests associated with silvicultural prescriptions commenced in July 2008 and concluded in February 2009. The uneven-aged (UEA) cores received patch cutting with single-tree selection; harvesting created four small (0.5 ± 0.1 ha), two medium (1.4 ± 0.2), and two large (2.1 ± 0.3) patches in each UEA core (Kalb and Mycroft, this publication). Even-aged (EA) cores each received two clearcuts (3.9 ± 0.6 ha) and two three-stage shelterwoods. The midstory-removal stage of the shelterwood cuts was completed in the initial entry; the projected size of the shelterwood cuts is equal to the clearcuts.

In general, two types of ecological studies were implemented prior to harvest. Long-term monitoring began for woody plants (Saunders and Arseneault, this publication), breeding birds (Malloy and Dunning, this publication), and terrestrial salamanders (MacNeil and Williams, this publication). These communities were selected for monitoring because they either define the habitat (woody plants) or respond to the habitat at landscape (birds) or local (salamanders) scales.

The second group of ecological studies characterized responses of either focal species or focal communities. During the pre-harvest period, sampling was conducted for populations of cerulean warblers (*Setophaga cerulea*) (Islam et al., this publication), timber rattlesnakes (*Crotalus horridus*) (MacGowan and Walker, this publication), eastern box turtles (*Terrapene carolina Carolina*) (Currylow et al., this publication), and oak acorns (*Quercus alba* and *Q. velutina*) (Kellner et al., this publication) using designs specific to each taxon. Similarly, pre-harvest sampling occurred for assemblages of wood-boring beetles (Holland et al., this publication), moths (Summerville et al., this publication), small mammals (Urban and Swihart, this publication), and bats (Sheets, Duchamp, et al., this publication; Sheets, Whitaker, et al., this publication). These particular species and communities were selected for study based either on their conservation status and suspected sensitivity to one or more of the silvicultural treatments planned for the HEE (eastern box turtles, timber rattlesnakes, cerulean warblers, and bats) or for their anticipated rapid community-level response to changing resources (acorns) or habitat (small mammals, wood-boring beetles, and moths).

Lastly, sociological research was begun before or soon after initial harvests at the HEE. Social surveys of MMYSF recreationists and local residents were conducted during the pre-harvest period (Rogers et al., this publication), and a statewide survey to assess attitudes regarding forest management was completed shortly thereafter (Witter et al., this publication). In addition, a comprehensive assessment of values associated with forest-related activities on MMYSF was undertaken (Hoover, this publication).

This paper summarizes key findings from the ecological and sociological studies mentioned above, offers predictions for post-treatment responses, and concludes with a consideration of challenges and opportunities for HEE research in the future.

RESEARCH HIGHLIGHTS - ECOLOGICAL

Communities for Long-Term Monitoring

Plant communities form the basis for most wildlife habitat and are the most visible representations of forest ecosystem responses to management. Saunders and Arseneault (this publication) assessed the composition and structure of trees and large shrubs at two spatial scales. First, they tested for differences among the nine research cores using data from 683 sampling plots arranged systematically in a 75m × 150m set of arrays. Encouragingly, they found no significant differences among sites slated for different

silvicultural treatments. Comparisons among research cores without regard to assigned treatments revealed that tree diameter and basal area were similar among cores, although Core 7 exhibited fewer trees in the 30- to 50-cm diameter at breast height (d.b.h.) class. Core 7 also had lower stem density than most other cores. Basal area was dominated (44-75 percent of total basal area) by oak and hickory (*Carya*) and was concentrated in a relatively few large overstory trees. Although nearly absent from the overstory class, red maple (*Acer rubrum*) and sugar maple (*A. saccharum*) made up the majority of small stems. Once again, Core 7 differed in composition, with greater dominance of shade-tolerant species, lower basal area of oak and hickory, and lower species diversity than other cores. When importance values were considered from a multivariate perspective at the core level, treatment centroids exhibited little separation relative to the separation demonstrated across all nine cores.

Saunders and Arseneault (this publication) also found considerable variability in composition of woody vegetation within each of the research cores. Core 8 was notable for a large block of plots with a high proportion of white pine (*Pinus strobus*) and Virginia pine (*P. virginiana*) associated with plantings dating to the 1930s. Cores 1-4 in the northern portions of the HEE have larger components of intolerant tree species, likely reflecting the greater levels of natural disturbance experienced by these areas over the past two decades.

How will wildlife assemblages respond to changes in forest environments induced by silvicultural treatments? Terrestrial salamanders can serve as reliable indicators of forest ecosystem health at local scales, as they are sensitive to environmental change (Welsh and Droege 2001). MacNeil and Williams (this publication) sampled 5,092 salamanders of 5 species with 66 cover-object grids during the pre-treatment phase of the HEE. An additional 464 salamanders of 4 species were detected with quadrat surveys. Eastern red-backed (*Plethodon cinereus*) and northern zigzag (*P. dorsalis*) salamanders were ubiquitous and dominant members of the salamander community, constituting 65.2 percent and 28.8 percent of all encounters, respectively. Mean encounter rates based on cover objects showed little variation for units classified by proposed silvicultural treatment. However, salamanders were more likely to be encountered under cover objects on northeast-facing slopes in fall and on southwest-facing slopes in spring, which suggests an interaction of temperature and humidity on abundance, activity, or both. Interestingly, quadrat surveys, which were done only in spring, did not support the pattern seen with cover objects.

Wildlife responses to silvicultural treatments also can occur at landscape scales. Because of their greater mobility and concerns over declining trends in abundance, Neotropical migrant birds were chosen for monitoring as part of the HEE. Malloy and Dunning (this publication) analyzed pre-treatment data on 17,806 detections of 74 bird species collected at 240 survey points distributed systematically across the 9 management units. For a set of 12 indicator species, neither individual management units nor proposed silvicultural treatments explained meaningful levels of variation in detection probability. Moreover, abundance of the 12 species did not differ among the proposed silvicultural treatments. Comparison of results with avian surveys conducted in the Hoosier National Forest (HNF) in southern Indiana demonstrated that the HEE bird community closely resembled the HNF community in composition.

Focal Species and Communities

The eastern box turtle is one of the longest-lived vertebrates in eastern deciduous forests (Stickel 1978). Some evidence suggests that box turtles are sensitive to disturbances that alter local features of the environment, and the species has experienced substantial declines in abundance over the past several decades (Currylow 2011, Hall et al. 1999). Radio-tracking was undertaken in the HEE for 23 of 236 eastern box turtles captured and revealed that

home ranges before harvest were nearly double the largest estimates reported previously for the species (Currylow 2011; Currylow et al., this publication). Home ranges in 2007 (2.6 ha) were on average only 56 percent as large as in 2008 (4.6 ha), suggesting that the hot, dry conditions of 2007 may have depressed movements (Currylow et al., this publication). Size of home ranges did not differ notably between sexes, and only two individuals exhibited seemingly nomadic behavior (Currylow et al., this publication). Annual survival was estimated as 0.964, which provides an important baseline for future studies examining effects of timber harvest or other human disturbance on eastern box turtles (Currylow et al. 2011).

Timber rattlesnakes are endangered in Indiana and exhibit behavioral and demographic traits that could make them vulnerable to timber harvesting. Behaviorally, rattlesnakes, especially gravid females, exhibit philopatry to hibernacula sites (Brown 1991, Brown et al. 1982) and thus create clumped spatial patterns that could be affected negatively by harvesting of timber near den sites. Demographically, females exhibit long inter-birth intervals (3-5 years) and a lengthy developmental period before onset of reproduction (7-11 years, Brown 1991). Radio-tracking was used by MacGowan and Walker (this publication) to study movements of timber rattlesnakes in the HEE. Of 55 rattlesnakes captured, 23 were tracked during the pre-harvest period. Predictably, home ranges of these predators were substantially larger than observed for the omnivorous box turtles. Male rattlesnake home ranges (65.7 ha) also were substantially larger on average than those of females (20.6 ha) (MacGowan and Walker, this publication). Home-range sizes of rattlesnakes varied considerably within sexes (MacGowan and Walker, this publication). A portion of this variation appeared due to variation in body size, with correlations of $r = 0.65$ for 9 non-gravid females and $r = 0.60$ for 10 males ($P < 0.05$ for both one-tailed tests). Interestingly, the 23 rattlesnakes used 13 different den sites, and most of these were within research core areas (MacGowan and Walker, this publication).

Like eastern box turtles, cerulean warblers have experienced notable declines in abundance over the past several decades. Indeed, an annual rate of decline of 2.6 percent over the past 43 years has been estimated, which is a more rapid decline than all but two other woodland bird species covered in the North American Breeding Bird Survey (Sauer et al. 2011). Cerulean warblers are endangered in Indiana. Some evidence indicates that large-scale disturbance of habitat, including timber harvesting, can affect the species negatively (Jones et al. 2001, Wood et al. 2006). Point count surveys by Islam et al. (this publication) in the HEE detected cerulean warblers in all nine management units during the pre-harvest period. Abundance averaged 6.2 males km^{-2}, which is greater than densities found at other sites in southern Indiana (Islam et al., this publication). Mapping of 120 territories during the same period indicated an average territory size of 0.28 ha, within the range of territory sizes reported for other areas in southern Indiana. Most territories in the HEE units were spatially clustered, a pattern observed previously in cerulean warblers (Roth and Islam 2007).

Oaks are experiencing regeneration failure in the Central Hardwood Region, apparently due to suppression of disturbances that favor this shade-intolerant group (Abrams 2003, Aldrich et al. 2005, Larsen and Johnson 1998). Because of oaks' importance to wildlife and local economies (Johnson et al. 2002, McShea et al. 2007), loss of oak from MMYSF would create substantial shifts in natural and human communities. Kellner et al. (this publication) studied mast production by 112 individual black oak (*Quercus velutina*) and white oak (*Q. alba*) trees located in the HEE research cores during the pre-treatment period. They also estimated rates of *Curculio* weevil infestation and removal of acorns by wildlife. Not surprisingly, mast production covaried with tree size and was variable across years.

With the exception of a mast failure for black oaks in 2008, black oaks tended to produce more acorns than white oaks. Black oak acorns also suffered from

greater probability of weevil infestation than white oak acorns, with highest rates of infestation observed during 2008, a year of mast scarcity. Overall, there were no differences in removal rates of black and white oak acorns by wildlife. Vertebrates removed 39 percent of acorns from exclosures and were less likely to remove infested acorns, or those that were damaged or germinating. When white-tailed deer (*Odocoileus virginianus*) and wild turkey (*Meleagris gallopavo*) were denied access to acorns in exclosures, rates of acorn removal did not decline, indicating that these large vertebrates were not important predators of acorns in the HEE during the period of study. In contrast, when squirrels (*Sciurus*) were denied access to acorns, rates of removal appeared to decline somewhat (Kellner et al., this publication).

The Lepidoptera function as key agents of herbivory and pollination in forest systems, in addition to serving as an important prey resource. Summerville et al. (this publication) collected 277 species of macrolepidoptera in 1 year of pre-harvest sampling. Over one-third of these species were oak feeders. Analysis revealed significant correlations between the moth assemblage found within a forest stand and the importance of oaks, tree density, and percent basal area of shrubs and saplings. However, less than 25 percent of the moth community structure could be explained with the environmental variables measured. Moreover, moth communities across 20 forest stands exhibited weak or no spatial autocorrelation. Although niche-assembly processes appear to play a role in structuring forest moth communities, stochastic or other factors may be even more important, at least in the absence of disturbance or successional events that alter overstory composition (Summerville et al., this publication).

Wood-boring beetles (Buprestidae and Cerambycidae) rely on living or recently killed trees for portions of their life cycles. They serve as decomposers, ecosystem engineers, pollinators, and food resources within forest ecosystems (Gutowski 1987; Holland 2009; Linsley 1961, cited in Holland et al., this publication). Regrettably, some species in these groups also damage valuable hardwood species and thus function as pests. Holland et al. (this publication) captured 102 species of wood-boring beetles during the pre-treatment phase of the HEE, including several new state and county records. Fortunately, no exotic invasives were collected in the sample. Analysis revealed no differences in measures of species diversity among sites scheduled for future treatments. However, differences were noted between assemblages of proposed harvest sites and sites within the units that were not slated for the initial harvest. These differences may have been due to sampling of harvest and non-harvest sites in separate years, coupled with differences in weather in the 2 years. Roughly half of the species responsible for the difference appear to have generation times >1 year, which could cause annual variation in abundance that would have been reflected in the harvest versus non-harvest site comparisons.

Many species of temperate bats depend on forests for roosting and foraging habitat, and silvicultural practices have the capacity to alter the quality of these habitats. Concern also exists regarding the conservation status of bats, especially so for the federally endangered Indiana bat (*Myotis sodalis*). Sheets, Duchamp, et al. (this publication) and Sheets, Whitaker, et al. (this publication) assessed bat communities across the nine HEE management units. Mist netting yielded 342 bats of 8 species from 18 locations over 3 years (Sheets, Whitaker, et al., this publication), and acoustic sampling with bat detectors produced 3,759 min of recordings with bat calls of 7 species from 157 locations over 2 years (Sheets, Duchamp, et al., this publication). A single silver-haired bat (*Lasionycteris noctivagans*) was captured via mist netting in 2008 and likely was a migrant (Mumford and Whitaker 1982).

Based on acoustic sampling, bat activity was similar among sites designated for the three silvicultural treatments. Bat diversity was slightly higher at sites designated for EA management. Sites designated for UEA treatment exhibited greater probability of

high levels of bat activity than other sites. Levels of activity in four habitats were interior forest < corridors < forest edge ≤ pre-harvest openings, corresponding to reductions in structural density of vegetation. Among species, northern long-eared bats (*Myotis septentrionalis*) were more likely than other bats to exhibit high levels of activity within interior forest sites. Tri-colored bats (*Perimyotis subflavus*) also used interior forest locations at relatively high levels. Northern long-eared and Indiana bats used forest openings less than other species did.

Small mammals form an important prey base for forest predators, serve as important predators of insects and seeds, and promote dispersal of seeds and mycorrhizal fungi in some circumstances (Maser et al. 1978, Moore et al. 2007, Whitaker and Hamilton 1998). Urban and Swihart (this publication) captured 3,422 small mammals of 7 species from 32 trapping grids during the pre-treatment phase of the HEE. Site occupancy probability did not differ as a function of proposed silvicultural treatment for any of the small mammal species. Relative abundance of eastern chipmunks (*Tamias striatus*) was lower on sites proposed for EA treatment but did not differ among proposed treatments for white-footed mice (*Peromyscus leucopus*), pine voles (*Microtus pinetorum*), or short-tailed shrews (*Blarina brevicauda*). Survival of local pine vole populations was greater at sites with northeastern aspects, as was relative abundance of white-footed mice and short-tailed shrews. Amount of herbaceous cover was positively associated with relative abundance of pine voles and short-tailed shrews, and pine voles also were more abundant at trap sites with greater amounts of coarse woody debris.

Baseline Comparisons

Despite the wide array of taxa surveyed, few differences were reported in the pre-treatment data (Table 1). Differences fell into three broad categories. For those studies where inventories were taken over successive years, temporal differences were often reported either for the community as a whole, or for individual species within that community. Malloy and Dunning (this publication), for example, reported an increase in relative abundance of several focal bird species over the 3 years of pre-treatment data. Likewise, MacNeil and Williams (this publication) reported an increase in salamander encounters from 2007 to 2008. The exact causes of many of these increases are unknown, but they may be attributed to temporal variation induced by weather either on the HEE sites or, in the case of birds, on wintering grounds.

The second group of differences was related to local neighborhood habitat features (Table 1). Although Saunders and Arseneault (this publication; Figs. 7 to 15) did not statistically test for intra-core differences, they used spatially explicit maps to display the large variability in the woody plant community across each research core. They effectively demonstrated that habitat differed widely within cores, often varying by slope position and aspect. Corresponding to this variation in habitat, differences were noted for terrestrial salamander (MacNeil and Williams, this publication) and small mammal (Urban and Swihart, this publication) communities on northeast versus southwest aspects.

Other researchers alluded to intra-core differences within their communities but did not formally test for them, often because experimental design was not appropriate (Table 1). For example, Holland et al. (this publication) observed significantly different beetle assemblages between areas designated for future harvest and matrix sites (i.e., the remaining areas within each research core). However, as sampling of these two areas was conducted in different years, intra-core variability and temporal variability were confounded. Holland et al. (this publication) postulated that the synchronized emergence of several species with generations spanning multiple years would naturally lead to different community assemblages across years, and hence, pre-treatment spatial variability may not be that important for the wood-boring beetle community.

Table 1.—Summary of pre-treatment data results from ecological studies in the Hardwood Ecosystem Experiment. Significant temporal (i.e., across years) or spatial (i.e., within or across cores or treatments) comparisons tested by the authors of the source are denoted by capital letters; differences mentioned but not significantly tested are in lowercase. Comparisons not tested or mentioned are denoted by a question mark. In several cases, the comparisons were not applicable (denoted by n/a), usually because the sampling design did not allow temporal comparisons or spatial comparisons at that scale.

Group	Units	Number of inventories	Differences tested or mentioned				Source[†]
			Temporal	Intra-core	Inter-core	Inter-treatment	
Monitoring communities							
Woody vegetation	1-9	1	n/a	y	Y	N	8
Breeding birds	1-9	3	Y	n/a			7
Salamanders							
Cover objects	1-9	2	Y	Y	?	N	6
Quadrat surveys	1-9	1	n/a	Y	?	Y	6
Focal groups							
Eastern box turtles	4-9	2	N	n/a	Y	N	1
Timber rattlesnakes	2,5,6,9	2	n	?	y	?	5
Cerulean warblers	1-9	2	N	n/a	N	N	3
Oak mast	1-9	3	Y	?	?	N	4
Bats							
Mist netting	1-9	3	N	?	y	?	10
Acoustic	1-9	2	Y	y	?	Y	9
Small mammals	1-9	2	N	Y	?	Y	12
Moths	1,2,3	1	n/a	y	Y	n/a	11
Wood-boring beetles	1-9	3	Y	y	?	N	2

[†] In alphabetical order, sources (all in this publication) are: 1) Currylow et al., 2) Holland et al., 3) Islam et al., 4) Kellner et al., 5) MacGowan and Walker, 6) MacNeil and Williams, 7) Malloy and Dunning, 8) Saunders and Arseneault, 9) Sheets, Duchamp, et al., 10) Sheets, Whitaker, et al., 11) Summerville et al., and 12) Urban and Swihart.

The last group of differences occurred at the broader landscape scale, either between units or between treatments (Table 1). There was no consistent pattern in reported differences at this scale; EA cores were highlighted as different for chipmunk abundance by Urban and Swihart (this publication), whereas UEA cores had lower terrestrial salamander encounter rates in quadrat surveys (MacNeil and Williams, this publication) and higher-level bat use (Sheets, Whitaker, et al., this publication). These differences may be due to landscape-level environmental gradients and/or differences in land use history. Jenkins (this publication) noted that the northernmost units (i.e., cores 1-3) lie within the pre-Wisconsin glacial limit; therefore, they have less topographic relief than the more southern units. Likewise, Carman (this publication) noted that Yellowwood State Forest was created in 1947, some 18 years after Morgan-Monroe. This difference in ownership history contributes to the legacy of the Great Depression-era planting of conifers throughout the Yellowwood State Forest by the Civilian Conservation Corps; two of these plantations exist in research core 8 (Saunders and Arseneault, this publication).

Given the variability in pre-treatment habitats (Saunders and Arseneault, this publication) and the inherent environmental gradients of such a large-scale study, even minor, nonsignificant, pre-treatment community differences could partially mask treatment effects (Monserud 2002). This outcome may be particularly troublesome for those species

or communities that have inherently high inter-annual variability in abundance and/or composition. Our hope is that the treatments are severe enough perturbations to the study species and communities to overcome these limitations of the HEE design. Nevertheless, mean responses may not convey the true treatment effect, and researchers should specifically look for non-uniformity in response by research unit within a given treatment, relating those differences back to unit-specific, pre-treatment data and known environmental variables whenever possible. Therefore, researchers should consider using mixed-modeling approaches for analysis, treating individual treatment units as random effects and tying the random effects to longitudinally measured, environmental covariables whenever possible.

Lastly, potential inter-unit differences may limit the ability to generalize results from studies using a subsample of units for comparison. Summerville et al. (this publication), for example, used only research cores 1-3 for their sampling and observed slight differences in moth communities across these cores (Table 1). Given their use of these data in broader, regional studies (e.g., Summerville et al. 2009), this experimental approach is an appropriate one. However, it would be improper to infer that the observed harvest treatment effects on these most northerly of the units would be representative of the response on the Yellowwood units, given the known differences in overstory communities (Saunders and Arseneault, this publication).

RESEARCH HIGHLIGHTS – SOCIOLOGICAL

As public property, state forests serve a wide array of stakeholders. Thus, it is useful to consider attitudes regarding forest management at multiple scales. Witter et al. (this publication) surveyed 1,402 Indiana residents from across the state to assess their opinions about forest management. Ninety percent of respondents professed to be somewhat or very interested in the state's fish, forests, and wildlife and the out-of-doors. Nearly 80 percent of respondents indicated that they had visited a state forest, although most did not distinguish among properties managed by the Department of Natural Resources-Division of Forestry as opposed to other state agencies. In keeping with this generic sense of state forests as any publicly owned land, 58 percent of respondents confessed to not being familiar with the Indiana Division of Forestry. Although 85 percent of Indiana forestlands actually are held in private ownership, 73 percent of respondents indicated they believed that government ownership accounted for half or more of Indiana's forestlands. When informed that state forests make up only 3 percent of Indiana's forests, 55 percent of respondents felt that this percentage was not enough.

Concern over the long-term health of Indiana's forests was greater for residents who were woodland landowners, had recently purchased lumber/wood products for home improvement, lived in southern Indiana, were >44 years of age, or professed liberal-leaning political views. When asked about harvesting of timber as a forest management tool, the approval rate was 95 percent if the goal was to protect woodlands from disease or wildfire, 85 percent if harvesting was overseen by professional foresters, 82 percent to improve wildlife habitat, and 61 percent to make lumber or other wood products. The vast majority (88 percent) of respondents agreed that Indiana forests should be managed for multiple uses. Lifestyles, early life experiences, and influences from resource professionals were important in shaping 48 percent of respondents' values about forest land.

To gain a clearer understanding of attitudes held by individuals directly affected by forest management on Morgan-Monroe State Forest, Rogers et al. (this publication) conducted an on-site survey of 165 recreational users and a mail survey of 1,239 neighboring landowners. The surveys contained an informational intervention designed to assess the degree to which enhanced knowledge about forest

management would change attitudes. Hiking was the most common form of recreation mentioned. Of landowners, 78 percent indicated they felt uninformed about forest management activities occurring on Morgan-Monroe State Forest.

Respondents compared images of mature unharvested stands as well as single-tree selection, shelterwood, and clearcut harvests. Forest stand density was correlated positively with the acceptability of management practices, desirability of forest scenes, and likelihood of visiting managed forests. In general, order of preference was intact stands > single-tree selection > shelterwood > clearcut. Recreationists varied in their support for harvesting timber from forests to derive ecological benefits. Not surprisingly, opponents of harvesting were 13 times more likely to find images of shelterwood cuts unacceptable. Providing information on the benefits of forest management increased acceptability scores for all treatments except the no-harvest treatment. Acceptability of the no-harvest option tended to decline following information intervention, which is consistent with a greater appreciation of the value of forest management. Providing information to respondents who initially held neutral or undecided attitudes about forest management for human benefits resulted in levels of acceptability for all practices except clearcutting that were comparable to attitudes held toward no harvest.

The multiple uses of MMYSF complicate attempts at valuation, as many of the benefits derived from these properties cannot be monetized readily, if at all. Hoover (this publication) considered intrinsic, cultural, and aesthetic values as they relate to human ties to forests generally and specifically to the role of forest management in creating a mix of successional stages. No monetary values were assigned for these attributes, or for the values of ecosystem services or research. Hoover (this publication) estimated monetary values associated with forest recreation, wildlife, and timber sales. Estimates for the first two categories were derived indirectly from secondary data. Using data for the HNF, day trips to MMYSF were estimated to make up >60 percent of the total visits. Expenditures for day trips likely ranged from $30 to $40 per party per trip (2003 dollars). With regard to fish and wildlife on MMYSF, annual hunting value was estimated at about $148,000, fishing value was estimated at roughly $246,500, and viewing value was estimated at $2,164,000.

Direct comparison of these values to impacts estimated from timber harvesting is not appropriate, because multipliers for determining total economic impact of wildlife are not available. The gross annual economic impact of timber sales from MMYSF for 1993-2010 averaged $94.9 million, assuming a gross economic multiplier of $51 per board foot (Hoover and Settle 2010). With regard to the various silvicultural treatments considered in the HEE, application of EA silviculture was estimated to reduce the value of forest land by approximately $2,500-$7,000 per ha. This reduction is due to the long waiting period for intermediate and final harvests with EA management.

PREDICTING POST-HARVEST RESPONSES

The studies summarized above provide a baseline from which to examine how populations of plants, wildlife, and humans respond to the silvicultural treatments. Theory and prior studies can guide predictions and form the basis for the material presented in this section. The focus here is on predicting short-term responses, as the uncertainty associated with responses increases at least proportionately with time since disturbance. Of course the timespan encompassed by short-term responses will differ according to the taxa considered. For instance, the length of time over which a short-term response is operative for overstory tree communities is likely an order of magnitude longer than the time frame of response for insects.

In the absence of overstory disturbances resulting from a widespread windstorm or from timber harvesting, the woody plant community of the HEE can be expected to continue its compositional shift from oak-hickory dominance to maple dominance that began decades ago. This shift is well documented in the literature (Abrams and Nowacki 1992, Fei et al. 2011, Johnson et al. 2003, Nowacki et al. 1990, Rentch et al. 2003). For example, Aldrich et al. (2005) reported that oaks had largely failed to recruit into the overstory of an Indiana old-growth stand for over 75 years in absence of grazing, fire, or other stand-level disturbances. Both basal area and stem density in undisturbed sites are also likely to increase in the foreseeable future, as shade-tolerant species continue to regenerate and recruit into multiple understory strata (Lorimer 1993, Nowacki and Abrams 2008).

Harvesting will produce obvious and dramatic changes in stem density and species composition. Following harvest, timber stand improvement within the clearcuts in the EA cores and patch cuts in the UEA cores removed most woody competitors >2.5 cm from those sites (Kalb and Mycroft, this publication). Future composition of these sites, therefore, will be determined by the combination of propagules from seed fall, advanced regeneration and sprouting of harvested stumps. Given that there was little advanced regeneration of oak, outside of chestnut oak (*Q. prinus* L.) on very dry ridgetops, and that the harvests were not timed with a mast year, clearcuts and patch cuts will likely be dominated for several years by shade-intolerant species such as tulip poplar (*Liriodendron tulipifera* L.) and sassafras (*Sassafras albidum* [Nutt.] Nees) that seed in from adjoining stands, with a few stems of oak and hickory arising from stump sprouts (Johnson et al. 2002). As is typical during stand initiation (sensu Oliver and Larson 1996), stem densities will increase until all growing space is occupied within these cuts, which will occur within 20 years in most oak-dominated forests (Johnson et al. 2002). Stem densities will then decline rapidly with competition-induced mortality during stem exclusion (Oliver and Larson 1996), with the survivors likely being the more numerous shade-intolerant species (Jenkins and Parker 1998, Morrissey et al. 2010).

In the UEA cores outside of the patch cuts, the single-tree selection harvests may accelerate successional development toward maple dominance (Abrams and Nowacki 1992, Jenkins and Parker 1998). Oak has not been regenerating in these stands for several years; therefore, most understory advanced regeneration consists of shade-tolerant species. Removing individual oaks across the site does not create large enough openings in the canopy to regenerate and sustain oak populations, even if shade-tolerant stems are removed as well (Johnson et al. 2002, Lorimer 1993). Continual single-tree selection likely will just thin overstory oaks more rapidly than the overstory oaks decline in the control stands, reducing oak dominance in the UEA cores until the maple and beech in the subcanopy strata recruit into the overstory (Sander and Graney 1993).

Within the shelterwood areas in the EA cores, removal of the midstory and many small overstory canopy trees through a preparatory harvest increased light levels at the forest floor and may allow development of a seedling bank of oak and hickory if overstory trees mast repeatedly before the understory fills again with shade-tolerant regeneration (Johnson et al. 2002). However, success of the shelterwood treatment can be hampered by deer herbivory to a greater extent than for clearcuts or patch cuts. Repeated browsing from high deer populations may quickly overwhelm the lower seedling densities that establish under an overstory as compared to the higher densities in full-light environments. In any case, researchers will need to be patient and schedule the next harvest of the shelterwood sequence (the establishment cut) when there is an established bank of oak seedlings throughout much of each site and, ideally, a bumper crop of mast in the overstory (Brose 2011, Loftis 1990).

Terrestrial salamanders generally are predicted to respond negatively to timber harvest, due to the hotter and drier conditions created by opening of the canopy and reduction in leaf litter (Herbeck and Larsen 1999) coupled with limited dispersal ability (Welsh and Droege 2001). However, negative effects may be ameliorated by leaving some basal area at a site (Knapp et al. 2003). Thus, salamanders are likely to be less negatively affected by initial entries into three-stage shelterwoods, and by smaller patch cuts. However, additional entries into shelterwoods could interfere with recovery (Morneault et al. 2004). Secondarily, site aspect may play a role, with a reduced effect of a particular silvicultural treatment predicted for northeast-facing sites.

Similar responses are predicted for species of small mammals that rely on humid sites with well-developed litter layers, including shrews and pine voles (Urban and Swihart 2011). In contrast, eastern chipmunks are predicted to increase in the dense, shrubby conditions characterizing southwest-facing openings with low basal areas shortly after harvest, irrespective of opening size (Urban and Swihart 2011). The presence of basking sites and abundant prey could also make these sites popular for timber rattlesnakes.

Following harvest, declines in the vicinity of openings are expected in abundance of birds and bats characterized by mature, interior forest habitat, including acadian flycatcher (*Empidonax virescens*), cerulean warbler (*Setophaga cerulea*), ovenbird (*Seiurus aurocapilla*), red-eyed vireo (*Vireo olivaceus*), scarlet tanager (*Piranga olivacea*), worm-eating warbler (*Helmitheros vermivorum*), wood thrush (*Hylocichla mustelina*), and northern long-eared bat (Malloy and Dunning, this publication; Sheets, Duchamp, et al., this publication; Sheets, Whitaker, et al., this publication). Increases in abundance or activity are predicted in early successional species or those with preferences for foraging in open areas, including brown-headed cowbird (*Molothrus ater*), Carolina wren (*Thryothorus ludovicianus*), indigo bunting (*Passerina cyanea*), eastern towhee (*Pipilo erythrophthalmus*), Indiana bat, little brown myotis (*Myotis lucifugus*), eastern red bat (*Lasiurus borealis*), hoary bat (*L. cinereus*), big brown bat (*Eptesicus fuscus*), and possibly tri-colored bat. Area effects are likely for most of these species, although some of the bats (e.g., Indiana bat) may be more responsive to the amount of edge created than to the size of an opening.

Moth communities did not vary predictably during the pre-treatment phase of the HEE. However, forest disturbances that alter microclimates and facilitate shifts in plant species composition can provide colonization opportunities for moth species that previously were absent from a site (Summerville et al. 2009). Shifts in moth assemblages thus are expected to follow shifts in plant species composition and/or microclimates. Short-term shifts could include addition of species associated with forbs and early successional woody vegetation. Short-term shifts also are anticipated in wood-boring beetle assemblages, as down logs and snags of various species attract beetles that rely on these resources for larval habitat. As with bird and bat communities, these responses are likely to be area-dependent, as the zone of attraction created is presumed to be correlated with the volume of harvested trees.

Nearly all of the respondents to the statewide survey conducted by Witter et al. (this publication) approved of timber harvest to improve forest health (defined as fire or disease). However, one-third of respondents preferred that the state's woodlands be untouched by humans. Ratings of visual acceptability also indicated that users of Morgan-Monroe State Forest prefer forest scenes with dense, unburned stands (Rogers et al. this publication). Thus, we predict negative responses to silvicultural prescriptions that deviate most from the unmanaged norm, i.e., greatest opposition to clearcutting, followed by patch cutting and then the midstory removal stage of shelterwood harvest. We also predict that educational efforts about the ecological and other benefits associated with

these silvicultural treatments will reduce negative perceptions. However, we suspect that effects of education will be greatest for those individuals who have the least emotional connection to MMYSF.

CHALLENGES AND OPPORTUNITIES

The HEE has progressed from infancy to toddler stage, with post-treatment responses beginning to be quantified. Still, a great deal of work remains to characterize both the biotic and abiotic variables related to community responses. Spatially explicit databases need to be built from pre-treatment data, and additional data layers, such as soils, canopy cover, and land-use history, need to be summarized from existing management records and available remotely sensed data. These data need to include not only research cores and units, but also the landscape context within which MMYSF lies. HEE databases can then more effectively inform the design of new studies, increase the efficiency of sampling, and allow more adaptive management in response to the harvest treatments.

As is the case for many large-scale, long-term forest management studies, the scope of inference for HEE—to describe the effects of forest management in the Central Hardwood Region—is broad and complex and involves investigation of ecological, economic, and social responses at an operational scale (Ganio and Puettmann 2008). A broad scale of inference is a strength of studies such as the HEE, because results collected are directly applicable at the scale of management activities, thereby leading to relatively rapid transfer of knowledge (Curtis et al. 2004, Puettmann 2005) and increasing the credibility of the research (Seymour et al. 2006) to land managers. Furthermore, large-scale, long-term studies allow the opportunity to superimpose smaller-scale, shorter-term studies of components or processes of the ecosystem that cannot be investigated across larger extents. If these fine-scale studies are done in a manner that explicitly recognizes the broader spatial context, they can be exceedingly valuable to quantify the scale dependency in process (Levin 1992) and identify emergent properties of the system (Puettmann et al. 2009).

Unfortunately, large-scale, long-term studies create a set of challenges, and we discuss five important challenges here. First, shortsightedness can mortgage the long-term value of studies such as the HEE. The short-term, ancillary experiments described above could have been superimposed onto the HEE project in a way that compromised the original objectives of the larger study. Shortsightedness can occur in research programs funded partially or entirely by "soft money"; a sudden infusion of cash from a large grant can direct the focus of researchers and administrators from the original objectives of the larger study. Similarly, political or social pressure to focus on taxa that are of most management concern at the time can distract attention or effort from longer-term objectives. Finally, monitoring programs can be tempting targets to cut during periods of fiscal duress, even though they are critical to describing the long-term trends of the larger study. While there is no one "right" or "wrong" approach to the influence of external pressures, development of a strong objective tree is critical to long-term success of large-scale forest management studies (Ganio and Puettmann 2008). The HEE has been proactive in this respect; the full HEE objective tree is provided as an appendix in Kalb and Mycroft (this publication).

Second, minimal replication of landscape treatments results in low statistical power in large-scale studies. The size of experimental units, and costs of maintaining monitoring programs and other ecological studies, limited HEE researchers to installation of three replicates. Consequently, investigators may have difficulty detecting small or moderate treatment effects for the various taxa (Ganio and Puettmann 2008, Gram

et al. 1997). Harrington and Carey (1997[2], as cited in Monserud 2002), for example, estimated that it would take 8-10 replications to detect a 20-percent treatment effect in small animal populations within their study ecosystem. Further, several research units in the HEE abut one another, so there is a possibility that responses for wide-ranging taxa (e.g., birds) might be pseudoreplicated as populations within each unit may not be spatially independent (Hurlbert 1984, Monserud 2002).

Although these deficiencies in experimental design can limit inference, researchers studying silvicultural systems and ecological phenomena at an operational scale often have no choice (Seymour et al. 2006). Use of mixed modeling, Bayesian hierarchical models, or other statistical approaches that explicitly account for spatial and/or temporal autocorrelation may help in this regard. Furthermore, HEE researchers should pursue the use of meta-analysis across multiple similar, but independent, large-scale studies to increase statistical power and inference (Monserud 2002). One such sister study is the Missouri Ozark Forest Ecosystem Project (Brookshire and Shifley 1997, Shifley and Kabrick 2002), which is similar to HEE in forest type, scale, and research objectives. Other opportunities likely exist for comparisons that extend beyond the boundaries of the HEE. For example, a post-treatment study of white-tailed deer impacts on herbaceous communities and tree regeneration included the installation of thirty-two 22m × 22m paired plots in clearcuts, shelterwoods, patch cuts, single-tree selection, and uncut matrix. In each pair, a 2.5-m tall exclosure was constructed. These exclosure plots are proposed to be included within a meta-analysis of deer impacts across the eastern United States.[3]

A third and imminent challenge facing the HEE team is to sustain momentum and commitment. Studies of manipulations to ecosystems need to persist at least long enough for transient dynamics to fade and the system to stabilize (Tilman 1989). In forests, transient dynamics can last for decades, with long-term treatment outcomes differing from or even contradicting the short-term treatment responses (Monserud 2002). Maintaining investigator and institutional momentum over this period can be difficult even with adequate funding sources. Academic demands for short-term, sharply focused, reductionist studies can discourage long-term participation by early-career scientists (Monserud 2002). Since forest development trajectories change slowly soon after the initial treatment, meaningful change in forest ecosystem structure or processes may not occur for a decade—well beyond a 3-year granting cycle or even a 7-year faculty tenure window. Consequently, university scientists often must relegate the study to a minor portion of their research portfolio.

Likewise, educational and outreach opportunities risk decline over this period, if prior events satiate public and professional consumers of project-related information. The "adolescent" stage of projects such as the HEE can be a dangerous period because the natural flow of products associated with long-term objectives can ebb or be perceived as stagnant before the study has matured enough to show long-term treatment effects and non-transient dynamics. Addressing the challenge of project adolescence will require project administrators to be actively engaged with researchers

[2] Harrington, C.A.; Carey, A.B. 1997. The Olympic Habitat Development Study: conceptual study plan. Unpublished manuscript on file at the Olympia Forestry Sciences Laboratory. Olympia, WA: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 38 p.

[3] Chris Webster (professor, Michigan Technological University) and Mike Jenkins (associate professor, Purdue University), personal communication

in identifying emerging research topics and new opportunities for collaboration.

Flexibility is a fourth challenge to the future success of the HEE. Planning documents for many large-scale forest management studies, including the HEE, state that an adaptive management framework will be used to guide future treatment prescriptions (e.g., Curtis et al. 2004, Franklin et al. 1999). However, adaptive management can become a nightmare for individuals asked to analyze results from the study. For instance, researchers and project administrators may choose to split experimental units to study a new treatment array that is being superimposed on the original study. From a management viewpoint, splitting can provide a powerful demonstration of individual treatment effects within a silvicultural system. From a research perspective, however, these splits can further reduce the statistical power of an already "power-starved" study, thereby weakening inference.

We are not asserting that an adaptive framework is unnecessary. On the contrary, researchers must be able to change treatment prescriptions when those treatments do not achieve their respective goals. This is the hallmark of a study comparing management or silvicultural "systems", which are planned sets of "treatments" used to accomplish some desired structural condition (Smith et al. 1997). By establishing benchmarks for individual treatment success a priori (e.g., Curtis et al. 2004), follow-up treatments can be done in a manner that does not compromise the original objectives of the study (Ganio and Puettmann 2008).

Unfortunately, these decisions often happen well after study establishment (Abbott et al. 1999), making application of follow-up treatments a contentious issue among researchers on an interdisciplinary team. For example, the use of prescribed fire in the HEE was deemed both too costly and administratively difficult to coordinate immediately after the first harvest entry across multiple research cores. However, advanced oak regeneration within shelterwoods is almost nonexistent, even 2 years after midstory removal and a subsequent mast year in those areas. Thus, project administrators and researchers will need to decide on the use of prescribed fire in areas slated for harvest during the next entry in 2028, largely as a way to build up a seedling bank of regeneration well ahead of overstory harvests. The use of prescribed fire will have to be carefully considered against the costs of increasing sampling effort to document its effects.

A fifth challenge of a long-term study is to maintain scientific, managerial, and social relevance. Many large-scale forest management studies, like the HEE, are often installed to fill a knowledge gap for management. The widely-cited Demonstration of Ecosystem Management Options study established in Washington and Oregon is a good example; Franklin et al. (1999: 4) explain that the study was conceived "to integrate ecological and commodity objectives." Studies that are inherently applied can still address fundamental research questions, but not without ingenuity and progressive thinking. The "hot-button" topic in science that may have helped drive the conception of a study will, in all likelihood, have long faded from scientific concern by the time the long-term results are known. HEE researchers will thus need to "re-think, re-analyze, and re-package" their long-term results to frame and answer emerging questions of basic interest.

Maintaining science of high relevance is a necessary but not sufficient condition for success of projects such as the HEE. Relevance to land managers and the public must also be maintained. Monserud (2002: 176) wrote: "As scientists, it is tempting to believe that the world will wait patiently for our well-designed experiments to be completed before conclusions can be reached." Practicing foresters, wildlife biologists, and other resource managers recognize that sound ecological and social science is only one small piece of a large list of information used to make policy and management decisions. Regulatory, economic, social,

and political pressures often drive the decisionmaking process in the management of publicly owned natural resources. Managers and the public often do not have the luxury of waiting on definitive answers and thus routinely make decisions based on short-term results.

In this regard, the HEE has the potential to be an exemplar for integrating ecologic and social research. The HEE's primary objectives explicitly state that economic and social ramifications of timber harvesting will be quantified, and that the HEE will be an outdoor laboratory for public engagement (Kalb and Mycroft, this publication). We have established a baseline in public attitudes regarding forest harvesting (Rogers et al., this publication; Witter et al., this publication); we now need to track changes in social attitudes and norms in response to ongoing research, education, and outreach efforts. Incorporating a longitudinal social science component into the HEE is perhaps the biggest challenge, and biggest opportunity, of all.

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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.

THE HARDWOOD ECOSYSTEM EXPERIMENT: EXTENSION AND OUTREACH

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Abstract.—The Hardwood Ecosystem Experiment (HEE) in Indiana is a long-term, large-scale experimental study of forest management and its impacts on plants and animals. Information from the HEE should and will be made available to a diverse group of potential users. This paper summarizes educational efforts during the pre-treatment period and highlights potential mechanisms and considerations for future efforts.

INTRODUCTION

Forests provide a multitude of environmental (e.g., carbon sequestration, enhanced water quality, wildlife habitat), economic (e.g., timber, wood products manufacturing, tourism), and social (e.g., recreation, aesthetics) benefits to society. The sustainability of these benefits is strongly tied to the stability of the resource. Although the number of hectares of forest area may describe the amount of forest base available, it fails to account for loss of forest values as a function of fragmentation and parcelization (Carman, this publication). In fact, the 2010 National Report on Sustainable Forests (U.S. Forest Service 2011) reported the loss of working (actively managed) forests as one of three overarching issues of "crucial importance" to our forests.

Forest land in the eastern United States is mostly (83 percent) privately owned, and landowners continue to grow older while the average parcel size decreases (Butler and Leatherberry 2004). Eighty-six percent of the 1.88 million ha (4.65 million acres) of forest land in Indiana is privately owned (U.S. Forest Service 2007) with an estimated 151,300 private landowners (Birch 1996). Most of these landowners control small areas with very few (3 percent) owning ≥40 ha (100 acres). Large ownerships of forests, such as state and federal forests, offer the best (and usually only) opportunity to manage for a wide diversity of forest life (Prugh et al. 2008). The Hardwood Ecosystem Experiment (HEE) provides a test of management options and offers a unique opportunity to learn how ecosystem responses differ as a function of the type of forest management.

The HEE takes place on Morgan-Monroe State Forest (9,710 ha) and Yellowwood State Forest (9,440 ha) in Morgan, Monroe, and Brown Counties in south-central Indiana. The study area and HEE study design are described in detail by Kalb and Mycroft (this publication). The general focus of the HEE is to experimentally study how timber harvesting affects plants and animals. This information can in turn be used to develop management prescriptions that maintain oak (*Quercus*)-dominated forest communities and the services they provide. A primary goal of the HEE also is to provide demonstration sites and educational programs to engage the public concerning forest management (Kalb and Mycroft, this publication). Here, we summarize educational efforts during the pre-treatment period of the HEE and highlight potential mechanisms and considerations for future efforts.

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PRE-TREATMENT EXTENSION EDUCATION

During the pre-treatment phase of the HEE, educational programming focused on the sustainability of oak forest ecosystems and how the HEE can help to inform management decisions within oak forest ecosystems. Outputs developed during the pre-treatment phase consisted of a project Web site, informational pamphlet, and newsletter. The HEE Web site (www.HEEForestStudy.org) was launched in 2007 and included a project overview, general study design and research activities, project partners and personnel with contact information, and job and meeting announcements. The HEE Web site also included a portal to a secure site where researchers could post and share information with each other. A quarterly newsletter, the HEE Update, was initiated in April 2008. Its purpose was to inform those directly involved with the HEE and those interested in learning about ongoing activities associated with the HEE. Individuals could sign up on the HEE Web site to receive the newsletter by e-mail, and all issues were posted on the HEE Web site. Both the Web site and newsletter offered an open door for stakeholders to learn more about the HEE.

Web statistics from 2007 through 2010 were not available. During 2011, visitors had 3,613 page views and an average page view time of 2 minutes per page. The initial HEE Update distribution was limited to project sponsors and investigators. By March 2009, the distribution list totaled 104. Ninety-two percent were university faculty, staff, and graduate students (44 percent); government agency personnel (30 percent); and HEE student workers (18 percent). More than half (56 percent) of university faculty, staff, and graduate students were directly involved with the HEE. Unfortunately, we were unable to separate traffic sources for the HEE site from the Forestry and Natural Resources Web site. We theorize that most Internet traffic was either from referral links from project sponsors and cooperators or direct traffic and Web searches from people who were familiar with the project. However, one must be at least aware of the HEE before seeking these information sources.

FUTURE OUTREACH EFFORTS

As the HEE progresses, it will provide much data to inform forest management policy for public forests in Indiana and the Central Hardwood Region. However, land management decisions are not strictly science-based. Studies have demonstrated that stakeholders do not disagree that sound science is necessary to solve natural resource problems, but they can disagree on the appropriate roles that science and scientific experts play in making management or policy decisions (Vogel and Lowham 2007). Because the Indiana Department of Natural Resources-Division of Forestry and many other natural resources agencies across the country seek stakeholder input on strategic planning and management prescriptions through public comment, it is beneficial to understand how stakeholders and the general public view woodlands and their management.

The public has a general lack of knowledge and awareness of woodlands and their management. For example, in a telephone survey of Indiana adults, Witter et al. (this publication) found that although 90 percent of respondents were either “very” or “somewhat” interested in Indiana’s forests and other natural resources, most (58 percent) were not familiar with the Indiana Department of Natural Resources-Division of Forestry, and 18 percent thought most of Indiana’s woodlands were privately owned. People generally are not opposed to timber harvesting (Schaaf et al. 2006), especially for specific reasons such as disease control and wildlife habitat (Witter et al., this publication), yet they may not understand the scientific basis of timber harvesting and other forest management decisions.

Information that improves the public’s understanding of the scientific basis of forest management may

change public opinion. For example, adjacent landowners and recreationists of Morgan-Monroe State Forest had mixed views regarding the same forest management practices used on the HEE (i.e., group selection, clearcut, shelterwood, no harvest). However, a minimal level of informational intervention improved acceptability of all forest management practices, although the increase was nominally small (Rogers et al., this publication). Educational programming developed by the HEE could provide recreationists, adjacent landowners, and other stakeholder groups a better understanding of the scientific basis of forest management practices and their associated tradeoffs. Specifically, programs that help people experience nature or link them to a natural resource professional (Witter et al., this publication) may offer the best potential for impacting their views about woodland management and timber harvesting.

In addition to informing state forest policy, the HEE also can serve to inform management decisions on private forests. Eighty-six percent of the 1.88 million hectares of forest land in Indiana is owned by private citizens or corporations (U.S. Forest Service 2007), including 141,300 non-industrial private forest landowners (Birch 1996). Thus, programs that target forest land in Indiana as a whole must involve private landowners. How do we reach such a large and diverse group? Indiana and other states already offer a variety of resources through land grant institutions and public resource agencies. Nonetheless, most landowners do not seek out those resources largely because they were unaware of assistance and programs available (Broussard and Lamprecht 2006, Hodge 1996, Measells et al. 2005).

Beliefs and values generally shared by private forest landowners and specific types of forest landowners may also offer insights into how to approach educational and outreach efforts associated with the HEE. Most people who own forest land indicate they believe timber is very important, but protection of natural beauty and wildlife ranks higher for reasons of ownership (Sampson and DeCoster 1997). Non-industrial private forest landowners share a deeply rooted stewardship ethic and are open to receiving more information about managing their land (Sampson and DeCoster 1997).

Information about forest management may be better received if consideration is given to the type of landowner because all forest landowners do not share the same goals and objectives for their property. Ross-Davis and Broussard (2007) found that forest landowners in northern Indiana could be divided into three distinct groups—new forest owners, forest managers, and passive forest owners. These groups differed in whether they considered their land to be managed, harvested timber on their land, and used information from the Department of Natural Resources and foresters (Ross-Davis and Broussard 2007). The HEE is focused on timber management and its impacts on plants and animals. However, woodland owners in Indiana (Ross-Davis and Broussard 2007) and across the country (Kilgore 2004) have a declining interest in timber production, and many own forests for reasons other than producing timber. Indiana woodland owners considered timber production to be "not important" as a motivation for ownership and ranked timber production much lower than enjoyment of scenery, privacy, wildlife habitat, and learning from nature among other motivations (Ross-Davis and Broussard 2007).

Even though the HEE focuses upon the use of timber harvesting as a management tool, educational programs associated with the HEE can present the costs and benefits of management activities (or disturbance in general) that will enhance wildlife habitat and other recreational and ecological values of forests. Hoover (this publication) reviewed the broad range of values that should be considered when making management decisions for forests in the Morgan-Monroe-Yellowwood Complex (MMYC). To the extent possible based on available data, he estimated a $2.6 million total annual value for outdoor

recreation within the MMYC. When the forest was used only for timber production, the gross annual economic impact of timber sales averaged $94.9 million from 1993 to 2010 (Hoover, this publication). However, he estimated a reduced value of between $2,590 and $7,770 per ha of forest land in the MMYC when even-age management is used. Although these numbers are not directly comparable (see Hoover, this publication), they demonstrate the importance of identifying recreational and ecological values of forests and how management decisions may influence them.

Clearly the HEE will provide much science-based information that can inform forest management on both public and private forests in Indiana and beyond. However, determining how different types of harvesting impact oak regeneration or how wildlife utilize harvest sites is not the end result. We need to better define what types of impacts we want to achieve and how we will measure them. The Renewable Resources Extension Act (RREA) offers federal funding to states with a focus on sustaining the nation's forests and rangelands. Dovetailing the goals, strategies, and performance measures of the HEE into existing programs such as RREA may offer a solution. For example, the RREA strategic issue of forest stewardship and health aims to increase the health and sustainability of privately owned forests by various ways, including educational programs that help landowners develop management strategies and implement stewardship practices (U.S. Department of Agriculture 2004). Information gained from the HEE can inform these management strategies and provide the background for demonstration of management practices.

Use of demonstration sites has been a cornerstone of traditional forest and wildlife extension programming. Adoption of management practices can be influenced, in part, by attending educational programs (Rasamoelina et al. 2010) and learning by example from peers (Snyder and Broderick 1992). These traditional in-person educational programs have much value for those who attend, but they also have limitations. Generally, in-person programs are expensive (advertisement, travel, and refreshments) and offer low potential for reaching large numbers of landowners given that attendees come from a limited geographic area near the location of the program. The HEE field sites have been used for media events, landowner workshops, professional training, and teacher summer courses, among others.

We certainly advocate for continuing these types of events, especially for specific target audiences such as foresters who provide technical assistance and advice to forest landowners, teachers who can instruct a new cohort of students annually, model forest landowners who can serve as an example to their peers, and media who relay information to the general public. However, truly understanding the impacts of the HEE resulting from these efforts requires the assessment of workshop participants in terms of how they used the information they learned and what type of difference it made for the people with whom they work. Adapting and expanding the HEE Web site to provide information on how to manage for specific plants and animals, the types of technical assistance and cost-share available to forest owners, and a means to track how visitors use the information would be of value.

Forest landowners will not seek out the HEE to find information relevant to their woodlands. Rather, HEE scientists must design educational programs specifically to engage this group. Creating new partnerships and enhancing existing partnerships with stakeholder groups offer much potential. Professional societies (e.g., The Wildlife Society, Society of American Foresters), non-governmental organizations (e.g., The Nature Conservancy, Indiana Woodland Steward), landowner groups (e.g., Indiana Forestry and Woodland Owners Association), and trade associations (e.g., Indiana Hardwood Lumbermen's Association) all have an interest in sustaining the ecological and biological integrity of forests. Many have extensive communication networks that include annual meetings, newsletters and circulars, and training/educational workshops.

These groups and many more also serve on the Indiana Forest Stewardship Coordinating Committee (IFSCC). This group brings together and maintains communication among stakeholder groups and individuals concerned with the management of Indiana's forest resources. Stakeholder groups represented on the IFSCC could communicate findings and management implications gained from the HEE to a broad audience. The same groups could also provide membership feedback to HEE scientists that could be used to identify key issues and optimal ways to reach end users (e.g., electronic publications, Internet seminars, and blogs). A formal survey instrument or focus groups would facilitate this process.

In summary, the HEE offers a unique informational resource for forestry professionals, educators, and landowners in Indiana. To maximize its value, we need to specifically outline and implement a plan that addresses desired impacts for each target audience that includes ways to evaluate success.

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PAST IS PROLOGUE: A SYNTHESIS OF STATE FOREST MANAGEMENT ACTIVITIES AND HARDWOOD ECOSYSTEM EXPERIMENT PRE-TREATMENT RESULTS

G. Scott Haulton[1]

Abstract.—Disturbance plays an important role in forest development processes. Present-day forest condition can be viewed as the cumulative result of various historical disturbance events; therefore, an understanding of disturbance history is important when describing overall forest condition. Pre-treatment studies of the Hardwood Ecosystem Experiment (HEE) have described various ecological and social aspects of portions of two State Forests in south-central Indiana that serve as the study area for the HEE. This paper describes the timber harvesting history for the HEE study area and how this history relates to HEE studies and current forest condition. Additionally, pre-treatment results of HEE studies are discussed in the context of past state forest management activities to highlight important emerging issues in state forest management that could be addressed by future HEE research.

INTRODUCTION

Disturbance events can have a dramatic, sometimes long-lasting, effect on forest structure and composition. The overall impact on forest development, ecosystem processes, community structure, and recovery from these events varies depending on such attributes as disturbance intensity, frequency, and spatial scale. Disturbances play a particularly important role in shaping the development of managed forests, where anthropogenic disturbance agents are prescribed alongside natural events to achieve predictable outcomes. Often, forest managers base this predictability upon an understanding of how events and manipulations in the past have affected the present-day condition of a forest. Furthermore, a thorough understanding of how historical disturbance events have shaped a forest may better inform future decisionmaking and help adapt management trajectories accordingly.

The Hardwood Ecosystem Experiment (HEE) is a long-term, landscape-scale collaborative project designed to explore the ecological effects and social aspects of forest management in the heavily forested region of south-central Indiana (Kalb and Mycroft, this publication). The HEE is based upon a comparative before-and-after experimental design using replicated treatments defined by even-age and uneven-age silvicultural methods and unharvested control areas. A "pre-treatment" study period began in 2006 to allow researchers to collect baseline data until the treatment harvests were implemented starting in the summer of 2008. Post-treatment data collected after harvesting will be compared to baseline conditions to quantitatively evaluate the response of various attributes to the harvests. Researchers have also used baseline data to evaluate the pre-treatment condition of the HEE research sites, known as "cores" (Kalb and Mycroft, this publication). Similarity among cores, and particularly among the three treatment types (i.e.,

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cores designated for uneven-age, even-age, or control treatments), is important in a before-and-after study because post-treatment effects will be easier to identify if pre-treatment conditions were similar.

Given that HEE study areas exist on two state forests, each with an extensive management history, an assessment of each core's timber harvest history would help to provide a more complete understanding of its condition before implementation of the HEE experimental treatments. In this paper, I summarize a review of state forest records that documented inventories and harvests at the present-day HEE research cores. My goals for this review were to: 1) evaluate the degree of similarity in timber harvest history among the HEE research cores and treatment types, 2) provide an historical context to the HEE pre-treatment results, and 3) offer suggestions for future research to address important questions raised during the pre-treatment assessment process regarding forest management on state forests and in southern Indiana in general.

STUDY AREA

The study area and experimental design of the HEE project is described in detail by Kalb and Mycroft (this publication). The historical documents reviewed pertained to inventories and harvests that took place at Morgan-Monroe (MMSF) and Yellowwood State Forests (YSF), where the HEE study areas (i.e., research cores) have been established.

State forest stands are classified using the forest type nomenclature of the U.S. Department of Agriculture, Forest Service (U.S. Forest Service 2011: Appendix 2). Indiana Department of Natural Resources-Division of Forestry continuous forest inventory data[2] indicate the Mixed Upland Hardwood forest type accounts for 46 percent of both MMSF and YSF, and is the single largest forest type at each property. On MMSF and YSF, stands are often dominated by a mix of species that may include yellow-poplar, sugar maple, and at least one oak species (see Appendix 1 for listing of scientific names). Oak-hickory forest types account for 43 percent of both properties, with white oak-red oak-hickory the largest single forest type in the oak-hickory group at the MMSF (32 percent) and YSF (22 percent). Conifer or mixed forest types account for <5 percent of either property.

Division of Forestry inventory data[3] also indicate approximately 2 percent of MMSF and YSF is <20 years old, and stands >80 years old occur across >50 percent of these two state forests, combined. Across MMSF and YSF, 74 percent of stands classified as oak-hickory are >80 years old, and <0.5 percent are <20 years old.

MATERIALS AND METHODS

Inventory and harvest records, as well as other supporting documents, were reviewed and summarized for each of the state forest management tracts occurring within a HEE research core. HEE cores were established across 27 management tracts in 2006; cores contain two or three tracts, each ranging from 17 to 53 ha. Records reviewed dated from 1971 to when pre-treatment surveys began on HEE research cores in 2006.

Although records were largely complete (with the exception of one missing file for a tract in core 8), inconsistencies in how information was recorded over the 35-year period limited the scope of the review and made certain compromises necessary in data analysis. For instance, though some records indicated the number and size of each regeneration opening (i.e., patch cut or group selection) associated with a harvest, other records simply provided a range in sizes

[2] Data on file with the Indiana Department of Natural Resources-Division of Forestry, Indianapolis, IN

[3] Data on file with the Indiana Department of Natural Resources-Division of Forestry, Indianapolis, IN

and an overall number of acres that were included in openings. Therefore, rather than estimating mean opening size associated with each harvest, the mean size of the largest opening is presented.

Inconsistencies in data recording also limited the scope of analysis, especially considering the relatively few harvests that had occurred within cores (mean 2.3, range 1-4). Due to low sample sizes within cores, differences were tested among treatment types (i.e., uneven-age, even-age, and control) using pooled tract-level harvest data within each treatment type. Mean differences were tested using multi-response permutation procedures in program Blossom (Cade and Richards 2005). These procedures are analogous to one-way analysis of variance when comparing differences among >2 groups, and are more effective for testing small sample sizes than many parametric and nonparametric alternatives. A significance level of alpha = 0.05 was used for analyses.

RESULTS

A review of management records spanning 35 years indicated 21 harvests have taken place in the MMSF and YSF tracts that became HEE research cores in 2006 (Table 1). Seven harvests occurred in each treatment type. The overall proportion of core area harvested was similar across treatment types, with 60, 56, and 61 percent harvested in uneven-aged, even-aged, and control cores, respectively (Table 1). Among cores, historical harvest rates ranged from 14 percent of the area in core 4 to 95 percent in core 2, both designated as control cores (Table 1).

Even-aged and control cores received most of their harvests from 1985 through 1995, with no harvests occurring after 1995 (Table 1, Fig. 1). In uneven-aged cores 60 percent of core area had been harvested by 1995, similar to the other two treatment types; however, 30 percent of core 7 was harvested again in 2003, 3 years before the HEE pre-treatment period (Table 1). Overall, the mean number of years between a tract's last harvest and the start of the HEE pre-treatment period was similar among the three treatment types, ranging from 18 years at even-aged and control cores to 22 years at uneven-aged cores (Table 1).

Three state forest management tracts within HEE cores received >1 harvest during the 35-year review period,

Table 1.—Number of harvests, proportion of unit harvested by time period, and the mean age of harvests in cores (range of observations) and treatment type (standard error, range of observations) at the initiation of the HEE.

HEE unit	No. of harvests	Proportion (%) harvested				Harvest age (years)
		1971-1984	1985-1995	1996-2006	Total (1971-2006)	
1	1	28.3	0.0	0.0	28.3	35
7	4	73.5	20.1	30.7	93.7	18.8 (3-23)
8	2	0.0	58.3	0.0	58.3	15 (14-16)
Uneven-age	7	35.0	24.7	10.6	59.7	20 (4.2, 3-35)
3	4	0.0	78.9	0.0	78.9	15.5 (15-17)
6	2	0.0	46.9	0.0	46.9	13
9	1	32.5	0.0	0.0	32.5	22
Even-age	7	9.2	46.4	0.0	55.7	15.7 (1.2, 13-22)
2	3	0.0	94.5	0.0	94.5	15.7 (15-17)
4	1	13.5	0.0	0.0	13.5	24
5	3	0.0	78.5	0.0	78.5	13.7 (11-17)
Control	7	4.7	56.3	0.0	61.0	16 (1.6, 11-24)
HEE overall	21	16.4	42.4	3.5	62.3	17.2 (1.5, 3-35)

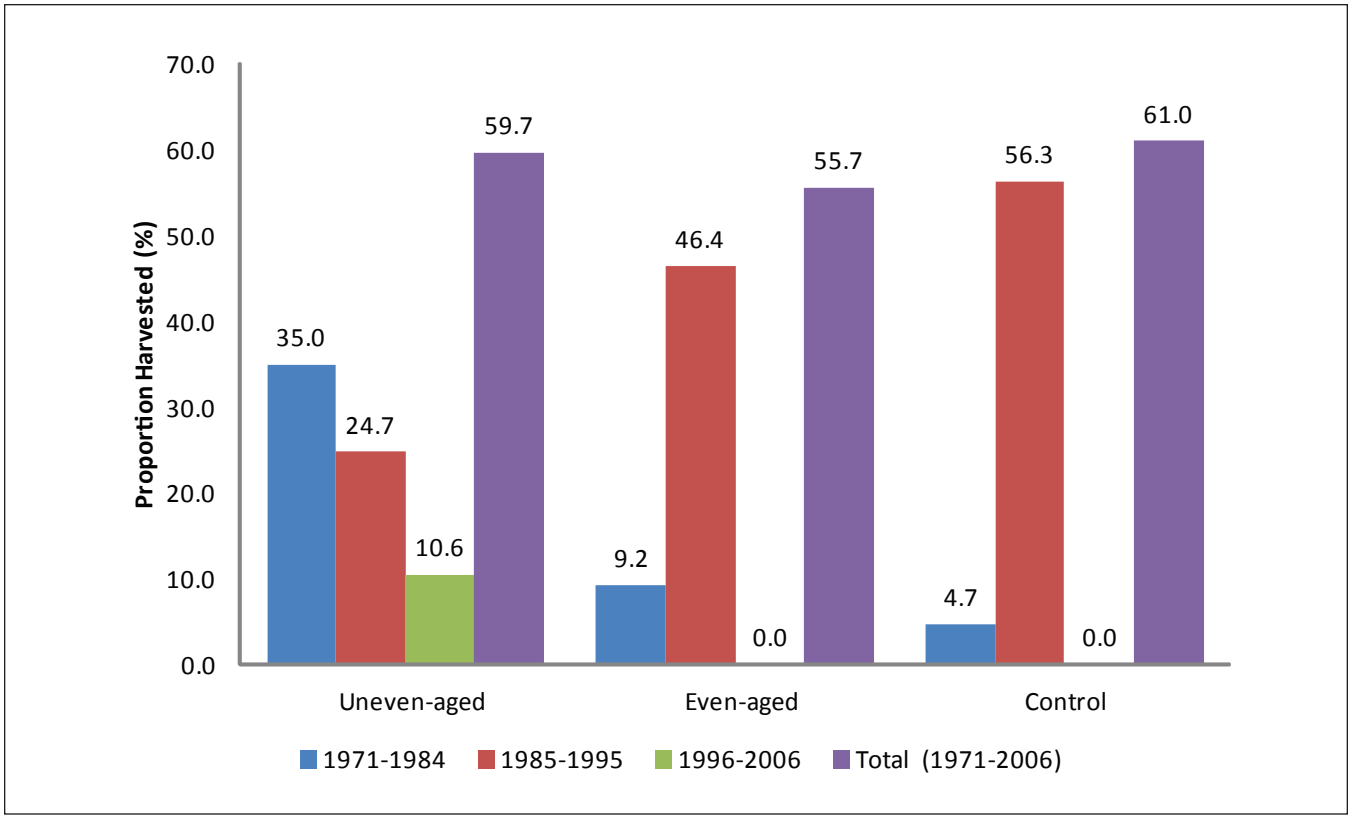


Figure 1.—Proportion of HEE treatment types harvested by time period. Labels over bars indicate treatment value.

and with the exception of the 2003 improvement harvest in core 7, additional harvests were due to salvage harvesting in previously harvested tracts that had experienced high wind damage at MMSF during the summer of 1990 (Table 2). Salvage harvesting was limited to areas in cores 2 and 3, affecting 17 percent of the overall HEE study area (Table 2).

Historically, non-regenerative, intermediate harvests (e.g., selection thinning, improvement harvests) were the most often employed harvesting method in core tracts, with 96, 89, and 92 percent of harvests in uneven-aged, even-aged, and control cores, respectively, using this cutting method (Table 2). Harvest records indicated that these techniques were typically prescribed to thin overstocked stands, release suppressed crop trees, and remove poorly developed individuals to reduce competition with trees of higher quality. Additionally, similar selective techniques were used during salvage harvests to remove damaged trees with commercial value. Most improvement harvest prescriptions mentioned removing fire-scarred trees of poor quality, indicating that, historically, the occurrence of fire was not uncommon on HEE cores.

Across all cores, regeneration harvests (e.g., patch cut and group selection) accounted for 7.4 percent of harvesting (Table 2). Among treatment types, even-aged cores had the highest proportion of regeneration harvests (11.7 percent), mostly associated with portions of salvage harvests where heavy damage resulted in a total loss of standing timber. The size of the largest patch cuts and group openings ("regeneration openings") was similar across treatment types, ranging from 0.9 hectares in uneven-aged cores to 1.4 hectares in even-aged cores (Table 3). Additionally, the mean age of regeneration openings was similar among treatment types, ranging from 15 years in uneven-aged and even-aged cores to 16 years in control cores (Table 3).

Table 2.—Proportion of HEE units salvage harvested and the proportion of harvests using patch cutting or group selection (Patch/Group) and single-tree harvesting techniques. Harvesting occurred at present-day HEE research cores, 1971-2003. n = number of harvests in unit; NA = no determination made due to incomplete records.

HEE unit (core/treatment)	n	Salvage harvested[a] (%)	Proportion of harvests (%)	
			Patch/Group	Single-tree
1	1	0.0	NA	NA
7	4	0	5.1	94.9
8	2	0	2.3	97.7
Uneven-age	7	0	4.3	95.7
3	4	66.6	13.9	86.1
6	2	0	5.5	94.5
9	1	0	NA	NA
Even-age	7	26.7	11.7	88.3
2	3	72.2	11.7	88.3
4	1	0	1.0	99.0
5	3	0	4.6	95.4
Control	7	23.4	8.3	91.7
HEE overall	21	16.7	7.4	92.6

[a] Includes patch cuts, group harvests, and single-tree harvesting done during salvage harvests.

Table 3.—Mean maximum regeneration opening area and the mean age of regeneration openings at the initiation of the HEE. For both attributes, means are presented by core (range of observations) and treatment type (standard error, range of observations). Harvesting occurred at present-day HEE research cores, 1980-2003. n = number of harvests in unit; NA = no determination made due to incomplete records.

HEE unit (core/treatment)	n	Maximum regeneration opening[a] area (ha)	Regeneration opening age (years)
1	0	NA	NA
7	3	1.2 (0.5-1.5)	14.3 (3-23)
8	2	0.4 (0.3-0.4)	15 (14-16)
Uneven-age	5	0.9 (0.24,0.3-1.5)	14.6 (3.26,3-23)
3	4	2.0 (0.8-2.8)	15.5 (15-17)
6	2	0.4 (0.3-0.4)	13
9	0	NA	NA
Even-age	6	1.4 (0.45,0.3-2.8)	14.7 (0.61,13-17)
2	2	2.3 (0.8-3.7)	16 (15-17)
4	1	0.1	24
5	2	0.5	12 (11-13)
Control	5	1.1 (0.65,0.1-3.7)	16 (2.23,11-24)
HEE overall	16	1.1 (0.24, 0.1-3.7)	15.2 (1.17,5-26)

[a] The largest opening created during a single harvest when multiple openings were created.

Mean volume harvest rates (i.e., mean proportion of pre-harvest volume harvested) differed among treatment types (T = -1.93, A = 0.14, p = 0.048), with uneven-aged cores having higher rates of volume removal than control cores, and even-aged cores intermediate to and not different from the other two treatment types (Table 4). Across cores, mean proportion of volume harvested ranged from 10 to 40 percent, with the highest tract-level volume harvest rate (52 percent) occurring in core 7 (Table 4).

The mean density of trees harvested per hectare was similar across treatment types, ranging from 21.6 trees ha^{-1} in control cores to 29.5 trees ha^{-1} in even-aged cores. In all treatment types, the oak-hickory species group provided the majority of trees harvested (Table 5; Fig. 2). Along with scarlet oak, a shade-intolerant species, moderately shade-intolerant oaks, such as chestnut oak, black oak, and white oak were among the most harvested species in each treatment type (Table 5). Relatively abundant non-oak species harvested across all HEE cores included yellow-poplar, large-toothed aspen, hickories, American beech, and sugar maple (Table 5).

Table 4.—Mean proportion of volume harvested by core (range of observations) and treatment type (standard error, range of observations). Harvesting occurred at present-day HEE research cores, 1971-2006. n = number of harvests in unit; NA = no determination made due to incomplete records.

HEE unit (core/treatment)	n	Volume harvested[a] (%)
1	0	NA
7	4	40.2 (24.7-52.9)
8	2	26.9 (24.7-29.1
Uneven-age	6	38.8 (4.9,24.7-52.9)
3	4	21.7 (15.5-26.7)
6	1	29.8
9	0	NA
Even-age	5	23.3 (3.2,15.5-26.7)
2	3	19.0 (13.7-27.5)
4	1	30.9
5	1	9.8
Control	5	19.6 (4.0,9.8-30.9)
HEE overall	16	26.8 (2.9,9.8-52.9)

[a] Proportion of pre-harvest volume removed during harvest.

DISCUSSION

Historically, cores in each of the HEE treatment types were similarly managed, with the exception of marginally higher rates of volume removal in uneven-aged cores compared to control cores. Driving this difference were harvests in core 7, where the mean proportion of volume harvested was 40 percent and harvesting had occurred as recently as 2003, just 3 years before the pre-treatment research period began. Interestingly, Saunders and Arseneault (this publication) reported core 7 had lower total and overstory tree density, and lower total and overstory basal area than most other cores. They also reported that relative to the other HEE cores, core 7 had

Table 5.—Relative density (trees ha^{-1}) of trees harvested 1982-2003 at present-day HEE cores by treatment type. Species ranked by relative density pooled across all HEE cores. Scientific names presented in Appendix 1.

	Relative density of trees harvested (%)[†]			
Common name	HEE overall	Uneven-age	Even-age	Control
Chestnut oak	22.9[ab]	10.2[ab]	24.2[ab]	36.7[ab]
Scarlet oak	17.5[ab]	10.9[ab]	27.5[ab]	10.3[ab]
Black oak	14.8[ab]	13.3[ab]	13.1[a]	12.7[ab]
White oak	8.7[a]	10.6[ab]	7.8[a]	7.0[a]
Northern red oak	6.8[a]	4.2	7.5[a]	5.5[a]
Yellow- poplar	6.6[a]	5.8[a]	5.6	4.2[a]
Bigtooth aspen	4.5[a]	6.4[a]	2.8	4.2[a]
Hickory spp.	4.3	7.0[a]	3.0	2.9
American beech	3.6	10.0[ab]	0.5	4.0
Sugar maple	3.4	9.9[a]	2.2	2.0
White ash	1.7	2.3	0.8	1.6
Sassafras	1.2	1.1	0.9	2.5
Red maple	1.1	1.7	0.6	2.2
American basswood	1.1	0.6	0.8	1.9
Blackgum	0.9	1.9	0.1	1.2
Black cherry	0.4	0.3	0.3	0.5
Red elm	0.1	0.2	0.1	0.2
Black walnut	0.1	----	0.1	0.1
Black locust	0.1	----	0.2	0.1
Elm spp.	0.1	----	----	0.2
American elm	< -0.1	< -0.1	----	----
Kentucky coffee-tree	< -0.1	----	< -0.1	----
Hackberry	< -0.1	< -0.1	----	----
Virginia pine	< -0.1	----	< -0.1	----

[†] Superscript "a" ([a]) denotes species that collectively account for 80 percent of treatment type total, superscript "b" ([b]) denotes species that collectively account for 50 percent of total.

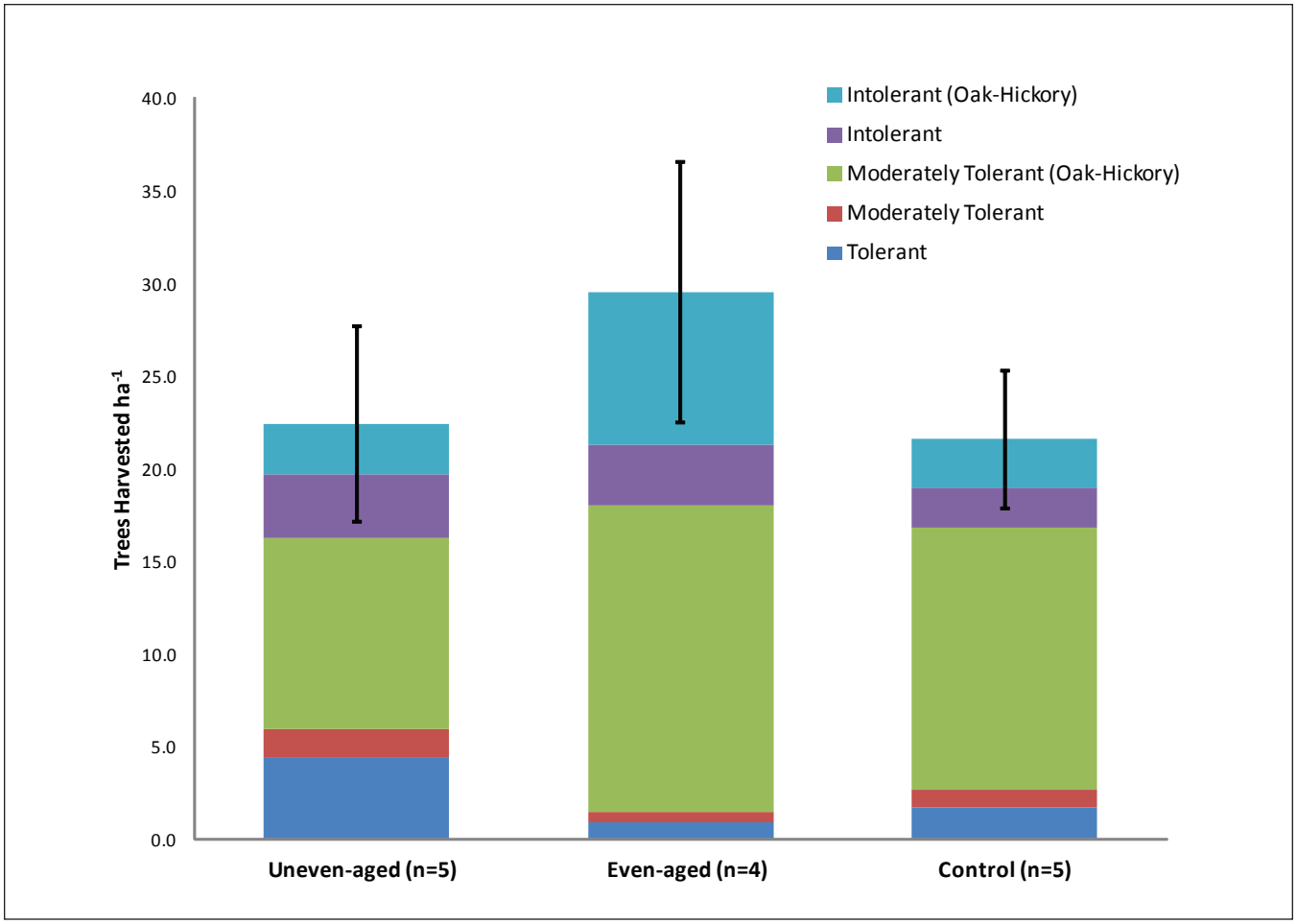


Figure 2.—Density (trees ha^{-1}) of trees harvested by HEE treatment type. Error bars approximate ±1 standard error of the total. n = number of harvests. Definitions of shade tolerance are given in Appendix 1.

irregular diameter and basal area distributions, suggesting the core contained fewer small sawlog-sized trees (i.e., 30 to 50 cm diameter at breast height). The findings of Saunders and Arseneault (this publication) are consistent with the effects expected from harvest records, and it is likely that during the pre-treatment study period stands in core 7 were still recovering from relatively recent and heavier-intensity harvests than stands had experienced in the other HEE cores.

Other authors in this volume found differences in pre-treatment conditions between uneven-aged cores and those of the other treatment types. Sheets, Duchamp, et al. (this publication) found HEE cores destined for uneven-aged treatments had a greater probability of high bat activity, compared to cores of other treatment types. Although these authors were unable to explain this result due to the limited scope of their study, they predicted forest structure likely had strong implications on bat activity and habitat use in the HEE study area (Sheets, Duchamp, et al., this publication). MacNeil and Williams (this publication) found mean encounter rates for salamanders were lower in uneven-aged cores compared to control and even-aged cores when a quadrat survey technique was used (but similar when cover object arrays were used), a result they attributed to relatively low encounter rates for two species. Future HEE research will provide a better understanding of how these species are affected by timber harvesting and whether pre-treatment results reflect historical harvest activities or are more strongly

associated with other factors. Despite these few noted differences, however, most authors in this volume found no difference among the treatment types, suggesting harvests may affect species differently and providing further justification for simultaneously studying harvest effects across communities, rather than just at the single-species level.

Although timber harvesting has occurred on all HEE research cores over the last 35 years, authors in this volume report that pre-treatment conditions reflect mature forest suitable for experimental, before-and-after comparisons of timber harvest effects. Because managers have primarily relied on selection silviculture and intermediate harvesting methods, as opposed to stand-replacing even-age techniques, cores have generally maintained overstory structure characteristic of mature forest (Saunders and Arseneault, this publication). Malloy and Dunning (this publication) report the most abundant avian species encountered on HEE cores were those that use mature forest habitat, whereas species dependent on early-successional habitats were found less often. Other studies, too, have found that bird species associated with mature forest habitats are typically abundant in selectively harvested forest (Haulton 2008), further suggesting that historical management of core tracts was consistent with the habitat requirements of mature forest bird species.

Malloy and Dunning (this publication) also report that two mature forest species listed in Indiana as "species of special concern," Hooded and worm-eating warblers, were among the most abundant species found on HEE cores during the pre-treatment period. The state-endangered cerulean warbler was found on all HEE cores during the pre-treatment study period, and densities were similar or slightly higher than those reported from other Indiana locations (Islam et al., this publication). Earlier research at MMSF and YSF also indicated that state forest harvesting practices appear to be compatible with the breeding habitat needs of cerulean warblers (Register and Islam 2008).

Bat studies on HEE sites indicated that pre-treatment conditions favor the northern myotis, which is typically found in mature forests (Sheets, Duchamp, et al. this publication; Sheets, Whitaker, et al., this publication). Sheets, Whitaker, et al. (this publication) suggested that the forest structure produced by selection harvesting at MMSF and YSF created interior forest conditions suitable for this "clutter-adapted" species. Indiana myotis, a federally endangered species that also uses interior forests, was also found on several HEE cores but prefers more open woodland conditions, which may explain why it was found less frequently than other forest bat species (Sheets, Whitaker, et al., this publication). Future HEE research will provide more insight into Indiana myotis' habitat use and preferences relative to various treatment types and harvesting techniques.

HEE cores historically were managed primarily using intermediate harvesting techniques, such as selection thinning and improvement harvests. Reliance on these methods over the history of MMSF and YSF reflects the developmental stage of most stands at these state forests. Much of the forest land composing these two properties was acquired in a depleted, and often cut-over, condition when it was placed into public ownership beginning around 1930 and continuing through the 1940s (Carman, this publication). Many of these second-growth stands were coming into maturity in the decades preceding the HEE pre-treatment period, requiring selective thinning and improvement harvests to release better-quality trees from competition and to remove damaged and poorly developed trees from the mature stand.

In these developing forests, stand improvement, rather than regeneration, was the priority; however, when regeneration opportunities were available, foresters typically prescribed group selection or larger patch cuts. Pre-treatment forest conditions at HEE sites reflect this philosophy; Saunders and Arseneault (this publication) found that even though oak and hickory species dominated core overstories, more

shade-tolerant species—often red and sugar maple—accounted for most of the smaller stems found in cores. Lack of large forest canopy openings to encourage the regeneration of less shade-tolerant species, such as oaks, has had ramifications for wildlife habitat as well; species requiring early-successional forest were encountered far less frequently than those that use late-successional, interior forest (Malloy and Dunning, this publication; Sheets, Duchamp, et al., this publication; Sheets, Whitaker, et al., this publication). Furthermore, given that the mean age of openings across all HEE cores was 15 years, many regeneration openings that were created on HEE cores have since succeeded beyond the stage where they would be attractive to early-successional species that use only recently created openings.

These results illustrate a major challenge for Indiana state forests in the coming years: as these forests continue to mature and require less maintenance through intermediate techniques, new management systems will need to be employed to regenerate and maintain oak-dominated forests. Summerville et al. (this publication) and Holland et al. (this publication) reported that oak was an important predictor of invertebrate species assemblage and diversity on HEE cores; loss of oak would undoubtedly have long-reaching effects throughout all levels of the forest community (Kellner et al., this publication). The HEE will play an important role in the transition on state forests from primarily single-tree management to a system that incorporates more regeneration methods and balances the habitat requirements of species that use late-successional interior forests with those that use early-successional forest communities.

FUTURE CONSIDERATIONS

Development of Oak Regeneration System

One of the primary research objectives of the HEE is to develop silvicultural systems that maintain oak-dominated communities and landscapes (Kalb and Mycroft, this publication). Even-age techniques, such as clearcutting and shelterwoods, have traditionally been favored for oak regeneration (Johnson et al. 2002); however, they are currently not widely used in Indiana and their effectiveness and ecological impact have not been thoroughly studied in the state. HEE research will provide a better understanding of how these cutting methods affect forest regeneration response on Indiana state forests and in Midwestern forests in general while also providing information on the social and ecological effects of these harvest types. Given the historical reliance on single-tree harvesting at HEE cores, and the pre-treatment conditions that suggest an overall lack of oak regeneration in cores, these prescriptions will provide much needed guidance toward the development of management systems that work to sustain oak as a dominant and ecologically important species group in Indiana forests.

Fire has played an important role in the development of Indiana's forests, and is believed to have influenced the dominance by oak and hickory prior to European-American settlement (Jenkins, this publication). Historical state forest records indicated that many improvement harvests were prescribed on core tracts to remove fire-scarred trees, suggesting that fires were not uncommon in these oak-dominated forests prior to public ownership. Other research has indicated that use of prescribed fire may provide an efficient and effective way to reduce competition from less fire-tolerant species, such as red maple and beech (Johnson et al. 2002). Given the history of fire in southern Indiana forests and its purported role in oak regeneration, HEE researchers should investigate further how prescribed fire might fit into an oak regeneration system for southern Indiana and what the community-level effects of such a system might be.

Balancing Habitat Requirements

Indiana state forests are managed to provide habitat for all native forest species. HEE pre-treatment results indicate, however, that mature forest communities dominate core tracts, conditions that reflect the overall trend across MMSF and YSF and, more generally, all

Indiana state forests. Recent data from the Division of Forestry's continuous forest inventory indicate approximately 2 percent of state forests is <20 years old.[4] These conditions have developed partly due to a reliance on intermediate harvesting methods that maintain canopy cover, enhance stand-level structural heterogeneity, and facilitate the development of old forest characteristics (Franklin et al. 2007), but they are also due to decades of actively limiting the effects of other disturbance agents through both fire suppression and insect pest control.

Although relatively homogenous mature forest conditions are preferred by many species and are desirable for pre-treatment cores in a comparative before-and-after harvesting-effects study such as the HEE, such landscape-scale homogeneity could occur at the expense of excluded assemblages and communities (Summerville et al., this publication). Forest managers would benefit from a better understanding of where appropriate thresholds exist between landscape heterogeneity and homogeneity. Specifically, at what point does the creation of early-successional forest habitat across a landscape negatively affect habitat suitability for interior, mature forest species (e.g., Becker et al. 2011)? Given the landscape-scale approach of the HEE and opportunities for community-level research, future HEE studies should provide a better understanding of impact thresholds, allowing forest managers to balance habitat opportunities for all species.

Maintaining Structural Habitat

During pre-treatment studies, HEE researchers identified several structural habitat features that may be especially important to forest species on HEE cores, including snags (Sheets, Whitaker, et al., this publication); unmerchantable harvest residue, such as tops and limbs (slash) (MacGowan and Walker, this publication); and other down coarse woody debris (Holland et al., this publication; Urban and Swihart, this publication). Sheets, Whitaker, et al. (this publication) indicated that retention of snags suitable for Indiana myotis' day-roosting will help reduce negative impacts that may occur during harvests. Similarly, MacGowan and Walker (this publication) reported that creation of early-successional habitat and retention of woody debris following harvest could enhance habitat for timber rattlesnakes. Holland et al. (this publication) noted that coarse woody debris serves as habitat to many wood-boring beetles, and retention of this structural component should be a consideration for sustainable forest management.

In the coming years, emerging technologies, such as the use of harvest residues and other forms of woody biomass for alternative energy sources, may compete with forest communities for these important structural resources (Indiana Department of Natural Resources-Division of Forestry 2011). Forest managers facing choices between the utilization of harvest residues and pre-existing coarse woody debris for energy production and retention of these materials for community sustainability and resilience will look to researchers for guidance on how to balance all needs sustainably. Although guidelines have been established in Indiana for the retention of harvest residue (Indiana Department of Natural Resources-Division of Forestry 2011) and structural forest habitat elements (Indiana Department of Natural Resources-Division of Forestry 2008), there are opportunities to rigorously test their impacts and effectiveness on managed forest communities. The long-term, multi-disciplinary, experimental approach of the HEE makes it well suited to investigate where appropriate structural retention thresholds exist in Indiana forests and what effects these new harvesting technologies may have on forest communities in the years to come.

[4] Data on file with the Indiana Department of Natural Resources-Division of Forestry, Indianapolis, IN

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APPENDIX 1.

Species' scientific names used in the text. Shade tolerance groups shown for tree species; tolerance groups were defined based on the shade tolerance scale of Niinemets and Valladares (2006) with intolerant and intolerant oaks and hickories <2.5, moderately tolerant and moderately tolerant oaks and hickories 2.5-3.5, and tolerant >3.5.

Common name	Scientific name	Shade tolerance	Tolerance group
Red maple	*Acer rubrum*	3.4	Moderately Tolerant
Sugar maple	*Acer saccharum*	4.8	Tolerant
Hickory spp.	*Carya* spp.	2.7*	Moderately Tolerant (Oak-Hickory)
Hackberry	*Celtis occidentalis*	3.2	Moderately Tolerant
American beech	*Fagus grandifolia*	4.8	Tolerant
White ash	*Fraxinus americana*	2.5	Moderately Tolerant
Kentucky coffee-tree	*Gymnocladus dioicus*	2.5	Moderately Tolerant
Black walnut	*Juglans nigra*	1.9	Intolerant
Yellow-poplar	*Liriodendron tulipifera*	2.1	Intolerant
Blackgum	*Nyssa sylvatica*	3.5	Tolerant
Virginia pine	*Pinus virginiana*	2.0	Intolerant
Bigtooth aspen	*Populus grandidentata*	1.2	Intolerant
Black cherry	*Prunus serotina*	2.5	Moderately Tolerant
White oak	*Quercus alba*	2.9	Moderately Tolerant (Oak-Hickory)
Scarlet oak	*Quercus coccinea*	2.1	Intolerant (Oak-Hickory)
Chestnut oak	*Quercus prinus*	2.9	Moderately Tolerant (Oak-Hickory)
Northern red oak	*Quercus rubra*	2.8	Moderately Tolerant (Oak-Hickory)
Black oak	*Quercus velutina*	2.7	Moderately Tolerant (Oak-Hickory)
Black locust	*Robinia pseudoacacia*	1.7	Intolerant
Sassafras	*Sassafras albidum*	1.7	Intolerant
American basswood	*Tilia americana*	4.0	Tolerant
American elm	*Ulmus americana*	3.1	Moderately Tolerant
Red elm	*Ulmus rubra*	3.3	Moderately Tolerant
Hooded warbler	*Setophaga citrina*		
Worm-eating warbler	*Helmitheros vermivorum*		
Cerulean warbler	*Setophaga cerulea*		
Northern myotis	*Myotis septentrionalis*		
Indiana myotis	*Myotis sodalis*		
Timber rattlesnake	*Crotalus horridus*		

* Average shade tolerance value for all species in genus occurring on HEE

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.

Swihart, Robert K.; Saunders, Michael R.; Kalb, Rebecca A.; Haulton, G. Scott; Michler, Charles H., eds. 2013. **The Hardwood Ecosystem Experiment: a framework for studying responses to forest management.** Gen. Tech. Rep. NRS-P-108. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 350 p.

Conditions in forested ecosystems of southern Indiana are described before initiation of silvicultural treatments for the Hardwood Ecosystem Experiment (HEE). The HEE is a 100-year study begun in 2006 in Morgan-Monroe and Yellowwood State Forests to improve the sustainability of forest resources and quality of life of Indiana residents by understanding ecosystem and human responses to forest management. Human-focused articles detail the history of disturbance from pre-Columbian times, the evolution of forest policy and management, values associated with the state forests that host the HEE, human perceptions of modern forest management in Indiana, and outreach efforts designed to inform stakeholders about the HEE. Interest in forest, wildlife, and fish resources is high in Indiana, and four of five residents have visited a state forest. Support for timber harvest as a management tool also is high if done professionally and not exclusively to supply lumber. Among users and neighbors of the forests that contain the HEE, educational intervention increased the acceptability of forest management and reduced the acceptability of not harvesting.

To study ecological responses to silviculture, nine research core areas were randomly assigned one of three treatments implemented in 2008 (uneven-aged, even-aged, or no harvest). Ecologically focused articles describe pre-treatment conditions during 2006-08 for targeted species (cerulean warbler, eastern box turtle, timber rattlesnake) and communities (woody plants, breeding birds, terrestrial salamanders, small mammals, bats, wood-boring beetles, and moths). Despite the broad diversity of taxa studied, few differences were noted during the pre-treatment period. Differences that did exist were related either to inter-annual variation, variation associated with local habitat features, or variation tied to differences occurring at the landscape scale. Predicted responses of plants and animals to silvicultural treatments are provided, and future challenges and opportunities associated with the HEE are discussed.

KEY WORDS: bats, beetles, birds, Central Hardwoods, experiment, forest management, human attitudes, Indiana, moths, oak, reptiles, salamanders, silviculture, small mammals, wildlife

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