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Table of Contents

2 **From the Editor**

Research Articles

3 **Effects of Drainage, Fire Exclusion, and Time-Since-Fire on Endemic Cutthroat Grass Communities in Central Florida**
Rebecca Yahr, Eric S. Menges, Dawn Berry

12 **Restoration of Florida Pine Savanna: Flowering Response of *Lilium catesbaei* to Fire and Roller-Chopping**
Jean M. Huffman, Patricia A. Werner

24 **Disturbance History of Two Natural Areas in Wisconsin: Implications for Management**
James E. Cook

36 **Plant Community Responses to Prescribed Burning in Wisconsin Sedge Meadows**
Michael A. Kost, Diane De Steven

46 **Changes in the Forest Landscape of the Charles C. Deam Wilderness, Southern Indiana, 1939–1990**
Michael A. Jenkins, George R. Parker

56 **Road Effects on Amphibian Movements in a Forested Landscape**
Phillip G. deMaynadier, Malcolm L. Hunter Jr.

66 **Assessing Floristic Quality in Southern Ontario Woodlands**
Charles M. Francis, Madeline J. W. Austen, Jane M. Bowles, William B. Draper

78 **An Evaluation of the Use of The Nature Conservancy Vegetation Classification for Mapping Bird Distributions at Chincoteague National Wildlife Refuge**
David I. King, Curtice R. Griffin, Paul J. Champlin, Tracey B. Champlin

Research Note

85 **A Common Species at the Edge of its Range: Conservation of Bear Oak (*Quercus ilicifolia*) and its Low-Elevation Rocky Summit Community in North Carolina**
Lawrence S. Barden

90 Book Reviews

92 Conferences and Workshops

95 NAA Membership Directory

RESEARCH ARTICLE

Changes in the Forest Landscape of the Charles C. Deam Wilderness, Southern Indiana, 1939–1990

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ABSTRACT: We used aerial photographs from 1939, 1974, and 1990 to examine how land cover has changed on the 5,286-ha Charles C. Deam Wilderness of Hoosier National Forest over this time span. Digital elevation models were used to examine changes in land-cover class (closed-canopy forest, open forest, agriculture/old-field, clearcut, and pine plantation) within each land type (flat uplands, bottoms, mesic slopes, and dry slopes). In 1939 most of the Deam Wilderness lands consisted of agriculture/old fields (33%) and open forest (25.6%), with only 41.4 % in closed-canopy forest. By 1974 most of the wilderness area had become closed-canopy forest (77.5%), and an even greater proportion was in that class in 1990 (86.3%). Since 1939 the forest of the Deam Wilderness has become much less fragmented with larger patch sizes, less total edge, and more total core area (area within a patch > 50 m from the edge). In 1939 land cover varied by land type: flat uplands and bottoms had the highest proportion of agriculture and old-fields (42.4% and 63.3%, respectively), and dry and mesic slopes had the highest proportion of open forests (30.2% and 28.3%, respectively). By 1974 closed-canopy forest was the dominant land cover on all land types.

Index terms: forest fragmentation, GIS, land use, ecosystem recovery, spatial analysis

INTRODUCTION

The past 70 years have been an era of critical change in the forests of eastern North America. During the 100 years prior to this period, much of the forest in eastern North America consisted of small areas of cutover timber with highly degraded native plant communities interspersed within a matrix of farmed fields and pasture (MacCleery 1992, Whitney 1994). Mayr (1890) described the forests of eastern North America as "scattered saplings, or a collection of isolated, gnarled, blighted trees, sometimes the last of their race." In 1911 the U.S. Congress enacted the Weeks Act, which authorized the federal purchase of forestland to protect watersheds (U.S. Department of Agriculture 1958). In 1924 the Clarke-McNary Act broadened this purchase plan to include lands with potential value for timber production (U.S. Department of Agriculture 1958). As a result of these laws, 23 national forests totaling approximately 3.7 million ha were established in the eastern United States between 1930 and the early 1950s (Cox et al. 1985).

Much of the land that became national forests in the eastern United States was highly eroded farmland (Shands and Healy 1977) that had been abandoned because it was physically unsuitable for continued agriculture (Hart 1968). According to the Soil Conservation Service (as cited by Sieber and Munson 1994), overall topsoil erosion ranged from 75% to 99% on these steep hillside farms that were often referred to as "ten-year land" because they could only be tilled for 10 years before enough soil eroded away to make further tillage unprofitable (Visher 1937). Because of their highly degraded condition, these lands were willingly sold for incorporation into forest reserves. During the 1930s, the Civilian Conservation Corps replanted large tracts of abandoned farmland in the new national forests with pine species, including *Pinus strobus* L., *Pinus resinosa* Ait., and *Pinus echinata* Mill. (Otis et al. 1986, Sieber and Munson 1994).

Within Indiana, incorporation of Hoosier National Forest (HNF) began in 1935 when the Indiana General Assembly determined that Indiana's national forest would be located in the southern uplands of the state (Sieber and Munson 1994). By 1951 enough land had been acquired to officially designate the HNF (Sieber and Munson 1994). Incorporation of land into the HNF continues today. As of 1999, the HNF is approximately 63,500 ha in size. Clearcutting has been the most common harvest method in the HNF since the 1960s, although there has been little harvesting since the mid-1980s. The Charles C. Deam Wilderness (5,235 ha within HNF) was creat-

ed in December 1982 and is located in Monroe, Brown, Lawrence, and Jackson Counties within the HNF (Figure 1). Since it did not receive a wilderness designation until 1982, the area that became the Deam Wilderness was in active forest management during the period when clearcutting was prevalent in the HNF. The oldest clearcut within the present-day wilderness was created in 1967, and the last clearcut occurred in 1978. Like the rest of the HNF, land parcels that became the Deam Wilderness were highly degraded by agriculture and contained highly disturbed plant communities. There has been no active forest management in the Deam Wilderness since its creation.

The Deam Wilderness is representative of the land use history of most eastern hardwood forests prior to their incorporation as national forests. In addition, prior to its designation as the Deam Wilderness, the area's management regime was typical of eastern national forests. For this reason, the Deam Wilderness offers an excellent opportunity to examine how land cover has changed and how these changes have altered spatial relationships within a forest landscape. This study addressed three questions. First, how was the land that became the Deam Wilderness used in 1939, when incorporation of Hoosier National Forest began? Second, how has the forest landscape changed from 1939 to 1974, and again to 1990, and how have these changes altered the fragmentation and other spatial relationships of the forest landscape? Third, how has land-use change differed with topography?

METHODS

Description of the Region

The Deam Wilderness is located in south-central Indiana (Figure 1) within the Pleasant Run Unit of the Hoosier National Forest (39° 03'N, 86° 20'W). According to Bailey et al. (1994) this area is within the Eastern Broadleaf Forest (Continental) Province of the Hot Continental Division of the Humid Temperate Domain. At a regional scale, the area is within the Brown County Hills Subsection of the Interior Low Plateau, Highland Rim Section (Bailey 1994, Keys et al. 1995) and contains dissected uplands underlain by siltstone, sandstone, shale, and limestone (Homoya et al. 1984). Soils are of the silt-loam type and are among the shallowest in the state (Indiana Department of Natural Resources 1984). Upland forests of the region are typically dominated by oas and hickories (e.g., *Quercus rubra* L., *Quercus alba* L., *Carya cordiformis* K. Koch, and *Carya glabra* Sweet) and *Liriodendron tulipifera* L., with *Acer saccharum* Marsh. and *Fagus grandifolia* Ehrh. common on mesic sites.

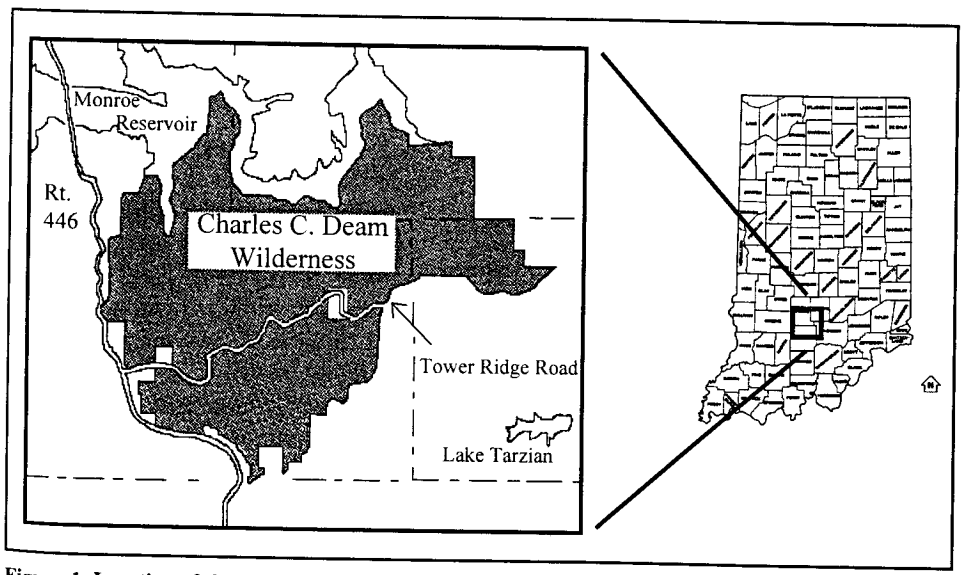

Figure 1. Location of the Charles C. Deam Wilderness within south-central Indiana. Areas not shaded around Tower Ridge Road are not part of the wilderness area.

Data Preparation

In this study, we examined land use in three separate years within a 51-year period—1939, 1974, and 1990—resulting in two intervals of change. Using aerial photos we delineated land use for the three dates into five land-cover classes: (1) closed-canopy forest, (2) open forest, (3) agriculture/old-field, (4) clearcut, and (5) pine plantation. Open forests were defined as forests with a canopy cover of less than 80%. Treeless pastures were included in the agriculture/old-field class. The locations of clearcuts were confirmed with U.S. Forest Service records. Former clearcuts were considered closed-canopy forest when they were no longer discernable from aerial photos. Aerial photos were at the following scales: 1939—1:20,000, 1974—1:15840, and 1990—1:30,000.

To reduce topographic displacement (distortion resulting from topography) and tilt displacement (distortion resulting from tilt in the camera optical axis), we determined the effective area of all photos (Paine 1981) and only delineated land use within these areas. The effective area is the central portion of each photograph bounded by lines bisecting the area of overlap with every adjacent photograph (Lillesand and Kiefer 1994). Distortion is lower in this part of the photo than in areas nearer the edge (Paine 1981). Photos containing areas of steep terrain were viewed stereoscopically during delineation. All land use delineations were converted to a common scale (1:24,000) and transferred to 7.5-minute USGS topographic quadrangle maps using a Bausch and Lomb zoom transfer scope. Separate land use maps were created for each year (1939, 1974, and 1990). Overall, the five land-cover classes were easily discerned from the three sets of aerial photos. The 1939 photos were the most distorted and required more adjustment of the zoom transfer scope. Although the 1990 photos were at a larger scale, land-cover classes were readily discernible because the photos were color infrared. The land use maps were transferred onto tracing paper and digitized by scanning into TIF files. ARC/INFO was used to create georeferenced coverages (UTM projection) in square meters from the TIF files.

Data Analysis

Total coverage and percent coverage of each land-cover class were determined for each year (1939, 1974, and 1990). We used ERDAS IMAGINE to calculate the transition matrix of land-cover classes between 1939 and 1974 and between 1974 and 1990. From this matrix, we assessed how land cover shifted between classes with time across the entire wilderness.

To assess the spatial characteristics of each coverage, we used FRAGSTATS (McGarigal and Marks 1995) to calculate mean patch size (MPS), large patch index (LPI), patch density (PD), total edge (TE), total core-area index (TCAI), and mean proximity index (MPI). MPS equals the sum of the areas (m^2) of all patches of a given class (agriculture/old-field, closed-canopy forest, etc.) divided by the number of patches of the same class, divided by 10,000 (to convert to hectares). LPI is the percentage of the landscape comprised of the largest patch. PD equals the number of patches of the corresponding class divided by total landscape area, multiplied by 10,000, then 100 to convert to 100 ha. TE is the sum of the lengths of all edge segments of patches of a given class. TCAI is the percentage of a

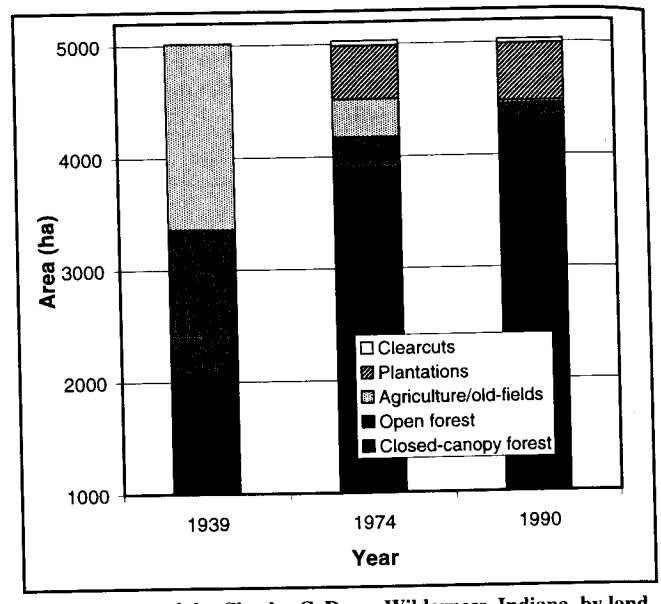

Figure 2. Area of the Charles C. Deam Wilderness, Indiana, by land-cover classes in 1939, 1974, and 1990.

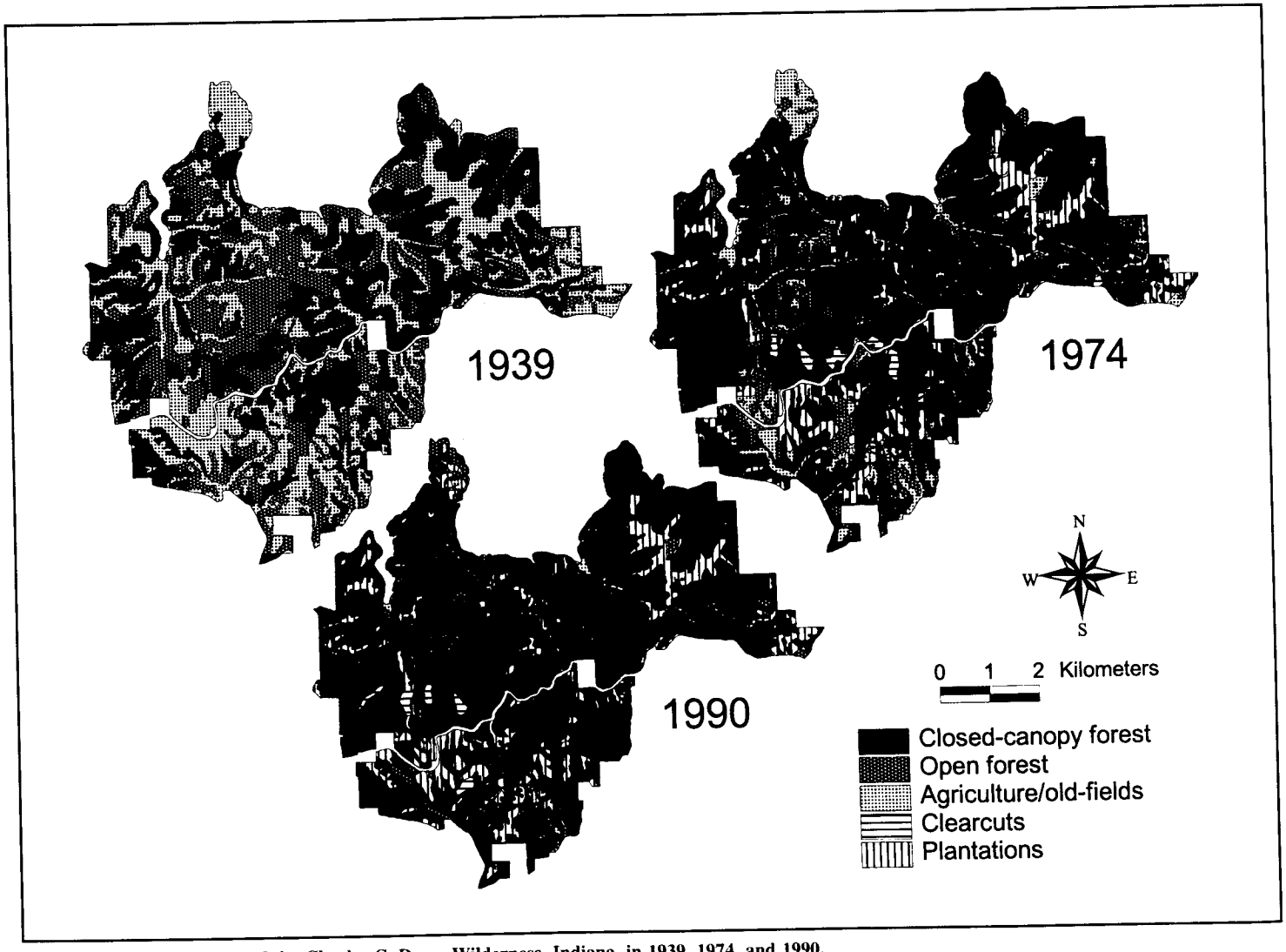

Figure 3. Land-cover maps of the Charles C. Deam Wilderness, Indiana, in 1939, 1974, and 1990.

Table 1. Transition matrix (total hectares and percent area) for different land-cover classes in the Charles C. Deam Wilderness, Indiana, between 1939 and 1974.

Land Cover Class 1974 / 1939	Agriculture/ old-fields		Open forest		Closed-canopy forest		Plantations		Clearcuts	
	ha	%	ha	%	ha	%	ha	%	ha	%
Agriculture/ Old-fields	294	17.5	169	10.1	801	47.7	413	24.6	2	0.1
Open forest	31	2.4	81	6.2	1,126	86.7	43	3.3	18	1.4
Closed-canopy forest	17	0.8	34	1.6	1,975	94.6	36	1.7	25	1.2

Table 2. Transition matrix (total hectares and percent area) among land-cover classes in the Charles C. Deam Wilderness, Indiana, between 1974 and 1990.

Land Cover Class 1990 / 1974	Agriculture/ old-fields		Open forest		Closed-canopy forest		Plantations		Clearcuts	
	ha	%	ha	%	ha	%	ha	%	ha	%
Agriculture/ Old-fields	22	6.3	81	23.6	186	54.4	54	15.7	0	0
Open forest	1	0.4	18	6.5	249	88.1	12	4.3	2	0.7
Closed-canopy forest	2	<0.1	20	0.5	3,800	97.4	55	1.4	23	0.6
Plantations	1	0.1	1	0.1	86	17.4	405	82.2	<1	<0.1
Clearcut	0	0	<1	0.7	32	71.4	0	0	13	27.9

class in the landscape that is core area (area within an individual patch > 50 m from the edge of the patch). MPI (Gustafson and Parker 1994) measures isolation and is calculated for individual patches as the sum of patch area (m^2) divided by the nearest squared edge-to-edge distance (m^2) between the patch and the focal patch for all patches whose edges are within a specified radius from the focal patch (1,000 m was used in this analysis). As MPI values decrease, isolation increases. The mean of these individual patch values was calculated for each land-cover class.

ARCVIEW was used to create a raster land type map of the Deam Wilderness based on 30-m x 30-m digital elevation models (DEMs). We divided the landscape into four topographic classes: (1) flat uplands (areas with elevations ≥ 180 m and slopes ≤ 5 %), (2) bottoms (elevation < 180 m, slope ≤ 5 %), (3) mesic slopes (aspect = 316°–134°, and slope > 5 %), and (4) dry slopes (aspect = 135°–315°, slope > 5%). Hammer et al. (1995) found that a 30-m resolution DEM correctly classified field-measured slopes into 5% slope classes only 25% of the time. However, the same DEM placed field-measured slopes into correct or adjacent slope classes 69% of the time. Because we combined derived slopes into two broad classes (0%–5 % and > 5%), the accuracy in this study probably was sufficient to divide the landscape into four broad classes. Total coverage of each land type was calculated. The coverage of each land-cover class by land type was calculated for 1939, 1974, and 1990.

RESULTS

Land-Cover Classes

Since 1939 there has been a dramatic shift in the percentages of land occupied by different land-cover classes in the Deam Wilderness (Figures 2 and 3). In 1939, 33% (1,656 ha) of the Deam Wilderness was in agriculture or old-fields. By 1974 agriculture and old-fields comprised only 6.7% (336 ha) of the landscape, and by 1990 this cover class comprised only 0.5% (24 ha) of the landscape. Open forest exhibited a similar reduction in area, declining from a peak of 25.6% (1,284 ha) of the landscape in 1939 to 5.5% (274 ha) in 1974 and 2.3% (113 ha) in 1990. Conversely, the percent of the Deam Wilderness in closed-canopy forest increased from a low of 41.4% (2,076 ha) in 1939 to 77.5% (3,897 ha) in 1974 and a maximum of 86.3% (4,339) in 1990. In 1974 and 1990, plantations comprised approximately 10% (479 ha and 513 ha, respectively) of the Deam Wilderness landscape, while clearcuts comprised less than 1% (44 ha and 38 ha, respectively). All clearcuts in the Deam Wilderness were created prior to its designation as wilderness in 1982.

Between 1939 and 1974, areas that were in agriculture/old-fields and open forests showed a large-scale shift to closed-canopy forests (Table 1). During this period, 47.7% of abandoned agriculture/old-fields and 86.7% of open forests reverted to closed-canopy forest. An additional 24.6% of abandoned agricultural areas and old-fields were planted into pine plantations. Between 1974 and 1990, most of the remaining agriculture/old-fields reverted to closed-canopy forest (54.4%, Table 2), and 17.4% of areas that were plantations in 1974 became closed-canopy hardwood

forests in 1990. Whereas some of this change may have resulted from errors in land-cover delineation, we observed several plantations in aerial photos that had senesced and succeeded to hardwoods. Severe winters during the late 1970s caused heavy mortality in these plantations. After 1974, 88.1% of open forests reverted to closed-canopy forest. During this same time period, 71.4% of clearcut areas delineated in 1974 underwent canopy closure and were indistinguishable from older surrounding stands in 1990.

Spatial Relationships

The increased dominance of closed-canopy forest in the Deam Wilderness since 1939 has resulted in a much less fragmented landscape, as measured by patch size (Table 3). Between 1939 and 1974 the mean patch size of closed-canopy forest increased from 47 ha to 491 ha, a tenfold increase, while the mean patch size increased another 50% to 730 ha after 1974. The mean patch sizes of agriculture/old-fields and open-canopy/grazed forests exhibited dramatic decreases between 1939 and 1990. The LPI of closed-canopy forests exhibited similar increases across the three sample dates (Table 3). Between 1939 and 1990 the PD decreased for agriculture/old-fields, open forests, and closed-canopy forests.

As the number and size of agriculture/old-field patches decreased between 1939 and 1990, the total edge of this land-cover class decreased drastically (344,080 m to 11,260 m, a decrease of 96%; Table 3). Open forest exhibited a similar decrease of 84% (247,260 m to 40,660 m). Although the total area of closed-canopy forest increased during this period (Figure 2), the condensing of forest patches decreased total edge from 282,260 m in 1939 to 192,740 m in 1990.

The larger patches of closed-canopy forest in 1974 and 1990 resulted in an increase in TCAI and MPI compared to 1939 (Table 3). In 1974 and 1990 these larger patches had larger core areas and were less isolated because they were no longer scattered within a matrix of agricultural fields and grazed forests. While the MPI of closed-canopy for-

Table 3. Landscape metrics for the Charles C. Deam Wilderness, Indiana, in 1939, 1974, and 1990.

	1939	1974	1990
Mean Patch Size (ha)			
Agriculture/old-fields	12	3	1
Open forest	13	3	2
Closed-canopy forest	47	491	730
Plantations	0	3	3
Clearcuts	0	7	8
Total Area	18	14	21
Large Patch Index (% of landscape comprised of largest patch)			
Agriculture/old-fields	3.8	1.4	0.1
Open forest	8.5	0.6	0.5
Closed-canopy forest	8.7	58.8	64.5
Plantations	0	2.1	2.1
Clearcuts	0	0.2	0.2
Total Area	8.7	58.8	64.5
Patch Density (# of patches/ 100 ha)			
Agriculture/old-fields	2.7	2.4	0.5
Open forest	2.0	1.8	1.2
Closed-canopy forest	0.9	0.2	0.1
Plantations	0	2.8	3.0
Clearcuts	0	0.1	0.1
Total Area	5.6	7.3	4.9
Total Edge (m)			
Agriculture/old-fields	344,080	103,360	11,260
Open forest	247,260	88,360	40,660
Closed-canopy forest	282,260	256,440	192,740
Plantations	0	144,540	152,220
Clearcuts	0	8,900	8,280
Total Area	436,800	300,800	202,580
Total Core Area Index (% of patch type that is core area)			
Agriculture/old-fields	27	16	4
Open forest	31	10	6
Closed-canopy forest	45	66	74
Plantations	0	15	15
Clearcuts	0	21	24
Total Area	35	55	66
Mean Proximity Index			
Agriculture/old-fields	449	18	1
Open forest	268	18	3
Closed-canopy forest	1,221	21,314	16,969
Plantations	0	55	42
Clearcuts	0	3	1
Total Area	503	492	438

as increased between 1939 and 1974, the MPI of the entire wilderness decreased as a result of the sharp decrease in the MPI of agriculture/old-fields and open forests. Patches of agriculture/old-fields and open forests were highly scattered in a matrix of closed-canopy forest in 1974 and 1990. The MPI of closed-canopy forest decreased slightly between 1974 and 1990, most likely as a result of isolated patches of old-fields and open forest succeeding to closed-canopy forest in the extreme northwestern portion of the wilderness. Since these patches were spatially isolated from the dominant closed-canopy forest patch, their MPI values were low.

Topography

Past land use in the Deam Wilderness was influenced by physiography. Flat uplands dominate the topography of the wilderness area, followed by mesic and dry slopes; bottoms comprise the smallest area (Table 4, Figure 4). In 1939, 42.4% of all flat uplands were in agriculture/old-fields and 23.5% were covered by open forests (Table 4). Closed-canopy forests covered only 34.1% of all flat uplands. By 1974 only 7.4% of flat uplands were in agriculture/old-fields and only 6.9% were in open forest. By 1990 the proportion of flat uplands in agriculture/old-fields and open forest decreased further to 0.4% and 2.5%, respectively. Over the same time period, the coverage of closed-canopy forests on flat uplands increased to a maximum of 79.3% in 1990. The total area of pine plantations and clearcuts did not change greatly between 1974 and 1990. Flat uplands contained the greatest percent and total area of pine plantations. Bottoms were primarily occupied by agriculture/old-fields in 1939 (63%, Table 4). Only 20.9% of bottoms were in closed-canopy forests in 1939, the lowest percentage of any land type. However, this percentage increased to 76.9% in 1990, the greatest increase in closed-canopy forest of any land type.

Mesic and dry slopes were used less frequently for agriculture and grazing than flat uplands and bottoms (Table 4). In 1939 over 50% of each of these two land types were in closed-canopy forests, compared to 34.1% of flat uplands and 20.9% of bottoms. However, mesic and dry slopes exhibited higher percent coverage of open forest in 1939 than bottoms or flat uplands. These mesic slopes and dry slopes were highly unsuited for agriculture, yet both types had relatively high coverage of agriculture/old-fields in 1939 (19.6% and 17.1%, respectively). These two land types have exhibited greater reforestation than flat uplands or bottoms; in 1990, 95.3% of mesic slopes and 95.8% of dry slopes were covered with closed-canopy forest.

DISCUSSION

Land-Cover Classes and Land-Use History

The results of our study show that at the landscape level there has been a considerable increase in forest cover in the Deam Wilderness. Cover of closed-canopy for-

Table 4. Total area (ha) and percent area of land cover classes in 1939, 1974, and 1990 by land type on the Charles C. Deam Wilderness, Indiana. Slight differences in total area between years resulted from small differences in the delineated wilderness boundary between years. Percent of the Deam Wilderness in each land type is given in parentheses.

	1939		1974		1990	
	Area	Percent	Area	Percent	Area	Percent
Flat Uplands (50.3%)						
Agriculture/old-fields	1,064	42.4	186	7.4	10	0.4
Open forest	590	23.5	174	6.9	63	2.5
Closed-canopy forest	855	34.1	1,732	68.9	2,000	79.3
Plantations	0	0	402	16.0	426	16.9
Clearcuts	0	0	21	0.8	23	0.9
Total area	2,509		2,515		2,521	
Bottoms (5.6%)						
Agriculture/old-fields	175	63.3	92	33.0	10	3.6
Open forest	44	15.9	24	8.6	32	11.3
Closed-canopy forest	58	20.9	154	55.2	216	76.9
Plantations	0	0	9	3.1	23	8.2
Clearcuts	0	0	0	<0.1	0	0.0
Total area	277		279		281	
Mesic Slopes (23.8%)						
Agriculture/old-fields	233	19.6	34	2.8	2	0.1
Open forest	335	28.3	46	3.9	9	0.7
Closed-canopy forest	617	52.1	1,054	88.9	1,136	95.3
Plantations	0	0	38	3.2	38	3.2
Clearcuts	0	0	15	1.3	9	0.7
Total area	1,185		1,186		1,192	
Dry Slopes (20.3%)						
Agriculture/old-fields	174	17.1	21	2.1	1	0.1
Open forest	306	30.2	28	2.8	10	1.0
Closed-canopy forest	535	52.7	931	91.6	975	95.8
Plantations	0	0	28	2.8	25	2.4
Clearcuts	0	0	8	0.8	7	0.6
Total area	1,014		1,015		1,017	

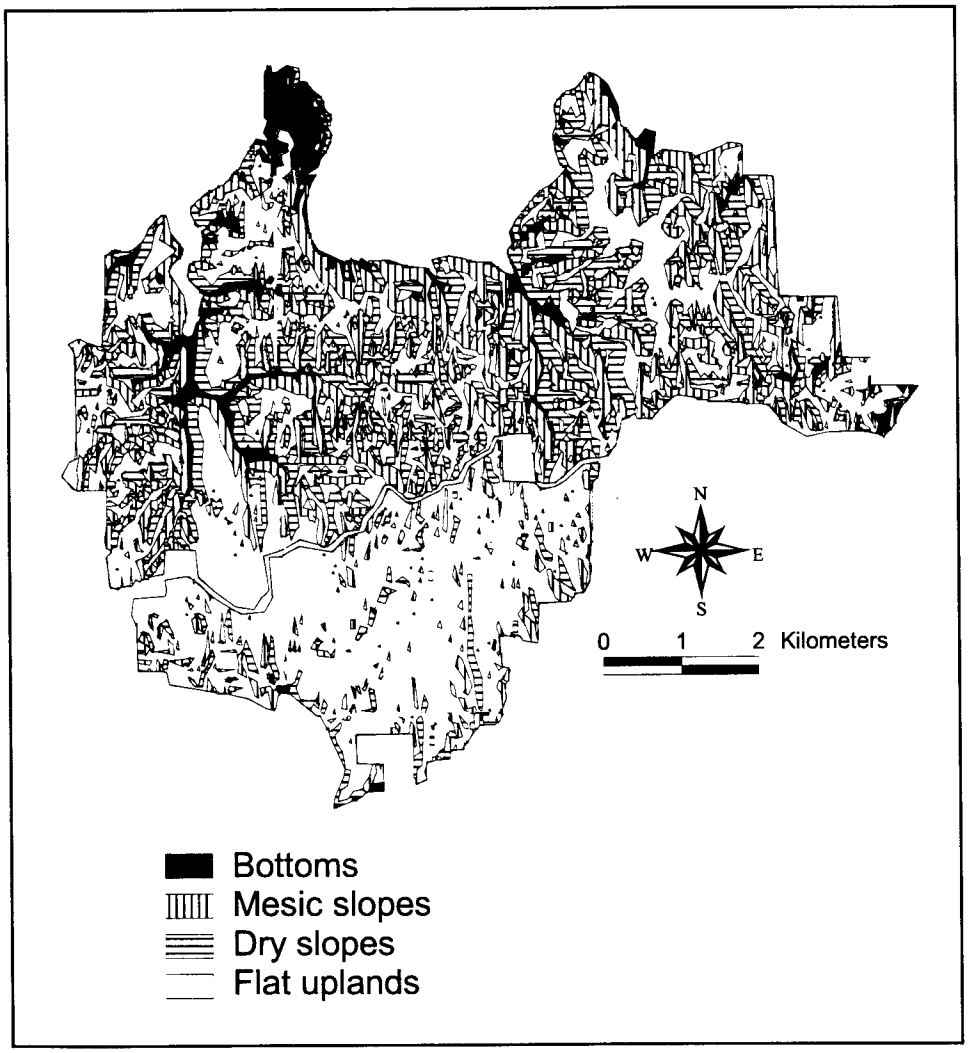

Figure 4. Land type map of the Charles C. Deam Wilderness, Indiana, derived from digital elevation models.

est doubled (from 41.4% to 86.3%) between 1939 and 1990. Similar reforestation has been observed in other parts of the Midwest (MacCleery 1992, Powell et al. 1992, Ebinger 1997). In Illinois forest cover has increased 41% from 1.22 million ha in the early 1900s (Telford 1926) to 1.72 million ha in 1985 (Iverson and Schwartz 1994). The creation of state and national forests in the first half of this century was the impetus for this recovery of forest cover across much of the region. However, these "new" forests were mostly relegated to areas of abandoned marginal farmland with steep terrain and highly eroded soils (Hart 1968, Otto 1983, Iverson 1988). Within the Midwest, there has been little change in forest cover on more level areas that remain profitable for agriculture or development (Parker and Merritt 1994, Smith et al. 1994). Consequently, there remains far less forest cover on the modern landscape of the Midwest than existed prior to European settlement (Ebinger 1997).

Although a third of the Deam Wilderness was in agriculture in 1939, it is unlikely that all of these cleared areas were cultivated in any single year. At the turn of the century in eastern Kentucky, only half of all cleared land was cultivated at any one time (Otto 1983). The remaining abandoned fields were used as rough pasture until they succeeded to forest. Davis (1930) described this system of shifting land use as "forest fallowing," whereby cultivation was constantly shifted, allowing fields to fallow for two or more decades before they were cultivated again. Fallowed areas and surrounding woodlands were also burned periodically to decrease insect populations and reduce brush cover. Historically, this method of farming represents a synthesis of American and Scots-Irish agricultural techniques (Otto 1983). While much of northern Indiana was settled by immigrants from New York and Pennsylvania, most of the early immigrants into southern Indiana originated from Kentucky (Hart 1972). These early settlers of southern Indiana brought with them the cultural farming practices of the Scots-Irish. In eastern Kentucky, Otto (1983) observed that this pattern of shifting cultivation began to break down as growing population and economic hardship during the Great Depression forced the continuous farming of marginal land without a fallow period. This pattern also occurred in southern Indiana. In many areas that later became the Hoosier National Forest, the population steadily increased until about 1900, but began to decrease during the early 1900s. For example, after reaching a peak around 1900, the population of Brown County (which contains part of the Deam Wilderness) had decreased by 40% before 1940, the time when incorporation of the Hoosier National Forest was initiated (Visher 1943).

Farming practices in southern Indiana included not only shifting cultivation but also the grazing of cattle and the running of hogs in woodlands (Whitney 1994, Parker 1997). Grazing cattle and rooting by hogs can have drastic effects on forest structure and composition by reducing woody regeneration and damaging the roots of existing trees (Lutz 1930, Diller 1937, Bratton 1975). Because over 25% of the Deam Wilderness consisted of open forest in 1939, it is probably safe to assume that livestock grazing was common throughout the Hoosier National Forest prior to incorporation. While common on all land types, open forest had greater percent cover on dry and mesic slopes than on any other land type.

Day and DenUyl (1932) identified four stages of grazing damage in Indiana woodlands: the early stage, the transition stage,

the open park stage, and the final stage. During the early stage, native understory vegetation and woody regeneration are greatly reduced. In the transition stage, years of successive grazing have reduced the canopy to less than 80% closure and removed the indigenous ground cover. The park and final stages occur when there are only scattered trees and the ground cover consists almost exclusively of weeds. Since we defined open forests as forests with less than 80% canopy cover, areas delineated in the 1930 photos were in these latter three stages of damage. The forest canopy in most of these areas had closed by 1974 and these areas were identified as closed-canopy forests. Additional research is needed to better understand how plant communities have recovered from grazing disturbance.

Spatial and Ecological Relationships

In recent years, numerous studies have examined the relationship between landscape structure and populations of forest-interior songbirds (Wilcove 1985, Robbins et al. 1989, Gustafson and Crow 1994, Askins 1995, Donovan et al. 1997). Generally, these studies have shown that sharp boundaries between forest and surrounding pastures and croplands result in increased rates of nest predation and parasitism by the brown-headed cowbird, *Molothrus ater* (Askins 1995). These nest problems are most severe in landscapes where small tracts of forest are embedded in a matrix of cropland and pasture (Wilcove 1985, van Dorp and Opdam 1987, Donovan et al. 1997). According to our results, the Deam Wilderness in 1939 was a highly fragmented landscape with islands of forest surrounded by agriculture, old-fields, and grazed forests. According to Mumford and Keller (1984), at the turn of the century the brown-headed cowbird was an abundant summer resident throughout the state of Indiana and remained in some parts of southern Indiana during the winter. Although precise estimations of songbird populations in southern Indiana are not available for the 1930s and 1940s (Dr. Harmon P. Weeks, Purdue University, West Lafayette, Indiana, pers. com.), our results show that nest parasitism during this period in southern Indiana may have been considerable. Our results also show that fragmentation of the forest landscape in southern Indiana has decreased considerably since 1939 following the creation of the Hoosier National Forest and surrounding state forests and parks.

The widespread intensive land use we observed in our study was typical of large parts of eastern North America in the early twentieth century and has had lasting effects on the region's forests. Throughout much of the region, contemporary vegetation composition and structure, successional dynamics, and landscape pattern are legacies of past intensive land use (Siccama 1970, Hart 1980, Foster et al. 1992, MacCleery 1992, Franklin 1994, Parker and Merritt 1994, Whitney 1994, Motzkin et al. 1996, Ebinger 1997, Foster et al. 1998). While historic land use explains the majority of variation in the present-day vegetation of some areas (Motzkin et al. 1996), historic land use and physical conditions typically have combined to determine present-day species composition and distribution (Niering 1953, Hunter and Swisher 1983, Glitzenstein et al. 1990, Nowacki et al. 1990, Nowacki and Abrams 1992). Our study suggests that the present-day composition and distribution of species in the Deam Wilderness have been determined by a combination of factors relating to past disturbance, topography, and edaphic characteristics. While predictive models of the relationships between plant species distribution and existing environmental conditions have been constructed, such models often account for a relatively small amount of variation across forest landscapes due to the inherent difficulty of incorporating past disturbance (Christensen and Peet 1984, Christensen 1989).

To improve understanding of the interrelationships between vegetation communities and their physical environment, numerous multifactor ecological classifications have been developed in the eastern United States (Spies and Barnes 1985, Hix 1988, Van Kley et al. 1994). The multifactor approach integrates both environmental and vegetation data into the development of classification units (Barnes et al. 1982). Some classification systems, such as that for forests of northern Wisconsin (Kotar et al. 1988), have included potential successional pathways of ecological classification units. However, in landscapes that have experienced varied land use history, such as southern Indiana, the impacts of past disturbance, in addition to physiography and edaphic characteristics, must be incorporated into ecological classification systems. It is difficult to classify sites where the typical relationships between vegetation and physiographic/edaphic characteristics have been altered by past disturbance without a clear understanding of past land use.

Although the increase in closed-canopy forest cover on the Deam Wilderness and throughout southern Indiana since 1939 has been dramatic, the native plant communities on severely degraded and abandoned farmland have not fully recovered. Herbaceous and understory plant communities in 20- to 50-year-old forests regenerated on abandoned farmland in wet-mesic bottomlands are still very different in composition from forest stands with similar topography that have not undergone major disturbance in 80–100 years (Jenkins 1998). While past agriculture in southern Indiana has shifted forest communities away from that typical of mature stands, silvicultural management without site preparation generally has not constituted a severe enough disturbance to shift the overall composition of ground-layer species (Jenkins 1998, Jenkins and Parker 1999). However, tree species composition in silvicultural openings differs from that of mature stands, which are typically oak-dominated and were initiated under more severe disturbance regimes (including unregulated logging, frequent burning, grazing, and cultivation) prior to the creation of Hoosier National Forest (Jenkins and Parker 1998). In the absence of major disturbance, mature stands are succeeding to shade-tolerant species, such as *Acer saccharum*. As past and contemporary disturbances continue to impact vegetation within southern Indiana, information about how these impacts have altered and continue to alter vegetation should be incorporated into the ecological classification system for the Hoosier National Forest (Van Kley et al. 1994). Improving our understanding of

how plant communities within forest landscapes have been shaped by past land use and respond to present-day management practices is essential to the successful management of forest ecosystems.

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LITERATURE CITED

Askins, R.A. 1995. Hostile landscapes and the decline of migratory birds. BioScience 67:1956-1957.

Bailey, R.G., P.E. Avers, T. King, and W.H. McNab (eds.). 1994. Ecoregions and subregions of the United States (map). U.S Geological Survey, Washington, D.C.

Barnes, B.V., K.S. Pregitzer, T.A. Spies, and V.H. Spooner. 1982. Ecological forest site classification. Journal of Forestry 80:493-498.

Bratton, S.P. 1975. The effect of the European wild boar *Sus scrofa* on gray beech forests in the Great Smoky Mountains. Ecology 56:1356-1366.

Christensen, N.L. 1989. Landscape history and ecological change. Journal of Forest History 33:116-124.

Christensen, N.L. and R.K. Peet. 1984. Convergence during secondary forest succession. Journal of Ecology 72:25-36.

Cox, T.R., R.S. Maxwell, P.D. Thomas, and J.J. Malone. 1985. This Well-Wooded Land: Americans and their Forests from Colonial Times to the Present. University of Nebraska Press, Lincoln. 325 pp.

Davis, D.H. 1930. A study of the succession of human activities in the Kentucky Mountains, a dissected highland area. Journal of Geography 29:95-99.

Day, R.K. and D. DenUyl. 1932. The natural regeneration of farm woods following the exclusion of livestock. Bulletin 368, Purdue University Agriculture Experiment Station, West Lafayette, Ind.

Diller, O.D. 1937. The forage cover in heavily grazed farm woods of northern Indiana. Journal of the American Society of Agronomy 29:924-933.

Donovan, T.M., P.W. Jones, E.M. Annand, and F.R. Thompson III. 1997. Variations in local-scale edge effects: mechanisms and landscape context. Ecology 78:2064-2075.

Ebinger, J.E. 1997. Forest communities of the Midwestern United States. Pp. 3-23 *in* M.W. Schwartz, ed., Conservation in Highly Fragmented Landscapes. Chapman and Hall, New York.

Foster, D.R., T. Zebryk, P. Schoonmaker, and A. Lezberg. 1992. Post-settlement history of human land-use and vegetation dynamics of a *Tsuga canadensis* (hemlock) woodlot in central New England. Journal of Ecology 80:773-86.

Foster, D.R., G. Motzkin, and B. Slater. 1998. Land-use history as long-term broad-scale disturbance: regional forest dynamics in central New England. Ecosystems 1:96-119.

Franklin, S.B. 1994. Late Pleistocene and Holocene vegetation history of Land Between the Lakes, Kentucky and Tennessee. Transactions of the Kentucky Academy of Science 55:6-19.

Glitzenstein, J.S., C.D. Canham, M.J. McDonnell, and D.R. Streng. 1990. Effects of environment and land-use history on upland forests of the Cary Arboretum, Hudson Valley, New York. Bulletin of the Torrey Botanical Club 117:106-122.

Gustafson, E.J. and T.R. Crow. 1994. Modeling the effects of forest harvesting on landscape structure and the spatial distribution of cowbird brood parasitism. Landscape Ecology 9:237-248.

Gustafson, E.J. and G.R. Parker. 1994. Using an index of habitat patch proximity for landscape design. Landscape and Urban Planning 29:117-130.

Hammer, R.D., F.J. Young, N.C. Wollenhaupt, T.L. Barney, and T.W. Haithcoate. 1995. Slope class maps from soil survey and digital elevation models. Soil Society of America Journal 59:509-519.

Hart, J.F. 1968. Loss and abandonment of cleared farm land in the eastern United States. Annals of the Association of American Geographers 58:417-440.

Hart, J.F. 1972. The middle west. Annals of the Association of American Geographers 62:258-282.

Hart, J.F. 1980. Land use in a Piedmont County. Annals of the Association of American Geographers 70:492-527.

Hix, D.M. 1988. Multifactor classification and analysis of upland hardwood forest ecosystems of the Kickapoo River watershed, south-western Wisconsin. Canadian Journal of Forest Research 18:1405-1415.

Homoya, M.A., D.B. Abrwel, J.R. Aldrich, and T.R. Post. 1984. The natural regions of Indiana. Proceedings of the Indiana Academy of Science 94:245-268.

Hunter, N.B. and K.J. Swisher. 1983. Arboreal composition of a Pennsylvania natural area: past, present and future. Bulletin of the Torrey Botanical Club 110:507-518.

Indiana Department of Natural Resources. 1984. Indiana forest soils handbook. Indiana Department of Natural Resources, Division of Forestry, Indianapolis.

Iverson, L.R. 1988. Land-use changes in Illinois, USA: the influence of landscape attributes on current and historic land use. Landscape Ecology 2:45-61.

Iverson, L.R. and M.W. Schwartz. 1994. Forests. Pp. 33-66 *in* J.P. Ballenot, ed., The Changing Illinois Environment: Critical Trends. Technical report of the Critical Trends Assessment Project, Vol. 3: Ecological Resources. Illinois Department of Natural Resources, Springfield.

Jenkins, M.A. 1998. The response of plant communities to human disturbance in southern Indiana forests. Ph.D. diss., Purdue University, West Lafayette, Ind. 280 pp.

Jenkins, M.A. and G.R. Parker. 1998. Composition and diversity of woody vegetation in silvicultural openings of southern Indiana forests. Forest Ecology and Management 109:57-74.

Jenkins, M.A. and G. R. Parker. 1999. Composition and diversity of ground-layer vegetation in silvicultural openings of southern Indiana forests. American Midland Naturalist 142:1-16.

Keys, Jr., J.E., C. Carpenter, S. Hooks, F. Koenig, W.H. McNab, W.E. Russell, and M.L. Smith. 1995. Ecological units of the eastern United States—first approximation (map and booklet of map unit tables). U.S. Department of Agriculture, Forest Service, Atlanta, Ga. 83 pp.

Kotar, J., J.A. Kovach, and C.T. Locey. 1988. Field Guide to Forest Habitats of Northern Wisconsin. Wisconsin Department of Natural Resources, Madison. 28 pp.

Lillesand, T.M. and R.W. Kiefer. 1994. Remote Sensing and Image Interpretation. John Wiley and Sons, New York. 750 pp.

Lutz, H.J. 1930. Effects of cattle grazing on vegetation of a virgin forest in northwest Pennsylvania. Journal of Agricultural Research 41:561-570.

MacCleery, D.W. 1992. American forests, a history of resiliency and recovery. Publication FS-540, U.S. Department of Agriculture, Forest Service, Washington, D.C. 58 pp.

Mayr, H. 1890. The general condition of North American forests. Garden and Forest 3:445-447.

McGarigal, K. and B.J. Marks. 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. General Technical Report PNW-GTR-351, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Ore. 122 pp.

Motzkin, G., D. Foster, A. Allen, J. Harrod, and R. Boone. 1996. Controlling site to evaluate history: vegetation patterns of a New England sand plain. Ecological Monographs 66:345-365.

Mumford, R.E. and C.E. Keller. 1984. Birds of Indiana. Indiana University Press, Bloomington. 400 pp.

Niering, W.A. 1953. The past and present vegetation of High Point State Park, New Jersey. Ecological Monographs 23:127-148.

Nowacki, G.J., M.D. Abrams, and C.G. Lorimer. 1990. Composition, structure, and historical development of northern red oak stands along an edaphic gradient in north-central Wisconsin. Forest Science 36:276-292.

Nowacki, G. J. and M.D. Abrams. 1992. Community, edaphic, and historical analysis of mixed oak forests of the Ridge and Valley Province in central Pennsylvania. Canadian Journal of Forest Research 22:790-800.

Otis, A.T., W.D. Honey, T.C. Hogg, and K.K. Lakin. 1986. The Forest Service and the Civilian Conservation Corps: 1933–42. Publication FS-395, U.S. Department of Agriculture, Forest Service, Washington, D.C. 219 pp.

Otto, J.S. 1983. The decline of forest farming in southern Appalachia. Journal of Forest History 27:18-27.

Paine, D.P. 1981. Aerial Photography and Image Interpretation for Resource Management. John Wiley and Sons, New York. 571 pp.

Parker, G.R. 1997. The wave of settlement. Pp. 369-381 *in* M.T. Jackson, ed., The Natural Heritage of Indiana. Indiana University Press, Bloomington.

Parker, G.R. and C. Merritt. 1994. The Central Region. Pp. 129-171 *in* J.W. Barrett, ed., Regional Silviculture of the United States. John Wiley and Sons, New York.

Powell, D.S., J.L. Faulkner, D.R. Darr, Z. Zhu, and D.W. MacCleery. 1992. Forest resources of the United States, 1992. General Technical Report RM-234, U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colo. 132 pp.

Robbins, C.S., J.R. Sauer, R.S. Greenberg, and S. Droege. 1989. Population declines in North American birds that migrate to the neotropics. Proceedings of the National Academy of Sciences 86:7658-7662.

Shands, W.E. and R.G. Healy. 1977. The Lands Nobody Wanted. The Conservation Foundation, Washington, D.C. 282 pp.

Siccama, T.G. 1970. Presettlement and present forest vegetation in northern Vermont with special reference to Chittenden County. American Midland Naturalist 85:153-172.

Sieber, E. and C.A. Munson. 1994. Looking at History, Indiana's Hoosier National Forest Region, 1600 to 1950. Indiana University Press, Bloomington. 131 pp.

Smith, W.B., J.L. Faulkner, and D.S. Powell. 1994. Forest statistics of the United States, 1992. General Technical Report NC-168, U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, Minn.

Spies, T.A. and B.V. Barnes. 1985. A multifactor classification of the northern hardwood and conifer ecosystems of the Sylvania Recreation Area, Upper Peninsula, Michigan. Canadian Journal of Forest Research 15:949-960.

Telford, C.J. 1926. Third report on a forest survey of Illinois. Bulletin 16, Illinois Natural History Survey.

U.S. Department of Agriculture. 1958. Land: the Yearbook of Agriculture 1958. U.S. Department of Agriculture, Washington, D.C. 605 pp.

van Dorp, D. and P.F.M. Opdam. 1987. Effects of patch size, isolation and regional abundance on forest bird communities. Landscape Ecology 1:59-73.

Van Kley, J.E., G.R. Parker, D.P. Franzmeier, and J.C. Randolph. 1994. Field guide: ecological classification of the Hoosier National Forest and surrounding areas of Indiana. U.S. Department of Agriculture, Forest Service, Bedford, Ind. 75 pp.

Visher, S.S. 1937. Indiana regional contrasts in soil erosion and their chief causes. Proceedings of the Indiana Academy of Science 46:143-159.

Visher, S.S. 1943. Indiana county contrasts in population changes. Proceedings of the Indiana Academy of Science 53:139-143.

Whitney, G.G. 1994. From Coastal Wilderness to Fruited Plain: A History of Environmental Change in Temperate North America from 1500 to the Present. Cambridge University Press, Cambridge, England. 451 pp.

Wilcove, D.S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66:1211-1214.