Prescribed Burning to Restore Mixed-Oak Communities in Southern Ohio: Effects on Breeding-Bird Populations

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Abstract: *Fire is being experimentally reintroduced to the forests of southern Ohio to determine its effectiveness in restoring and maintaining mixed-oak (*Quercus *spp.) forest communities. We studied the effects of repeated burning (1–4 years of annual burning) and recovery (1 year after burning) on the breeding bird community. Burning resulted in incremental but temporary reductions in the availability of leaf litter, shrubs, and saplings, but it did not affect trees, snags, or understory vegetation cover. Of 30 bird species monitored, 4 were affected negatively and 2 were affected positively by burning. Population densities of Ovenbirds (*Seiurus aurocapillus*), Worm-eating Warblers (*Helmitheros vermivorus*), and Hooded Warblers (*Wilsonia citrina*) declined incrementally in response to repeated burning and did not recover within 1 year after burning, suggesting a lag time in response to the changes in habitat conditions. Densities of Northern Cardinals (*Cardinalis cardinalis*) fluctuated among years in the control units, but remained low in the burned units. Densities of American Robins (*Turdus migratorius*) and Eastern Wood-Pewees (*Contopus virens*) increased in response to burning, but these increases were apparent only after several years of repeated burning. In general, burning resulted in short-term reductions in the suitability of habitat for ground- and low-shrub-nesting birds, but it improved habitat for ground- and aerial-foraging birds. Overall, there were no changes in the composition of the breeding-bird community. Total breeding bird population levels were also unaffected by burning. Our results suggest that prescribed burning applied on a long-term basis or across large spatial scales is likely to have adverse effects on ground- and low-shrub-nesting bird species, but other changes in the composition of the breeding-bird community are likely to be minimal as long as the closed-canopy forest structure is maintained within the context of prescribed burning.*

Quema Prescrita para Restaurar Comunidades Mixtas de Encino en el Sur de Ohio: Efectos sobre Poblaciones de Aves Reproductivas

Resumen: *Se está reintroduciendo fuego artificialmente en los bosque del sur de Ohio para determinar su efectividad para restaurar y mantener comunidades de bosques mixtos de encino (*Quercus *spp.). Estudiamos los efectos de quemas repetidas (1–4 años de quema anual) y de recuperación (1 año después de la quema) sobre la comunidad de aves reproductivas. La quema resultó en reducciones temporales en la disponibilidad de hojarasca, arbustos y renuevos, pero no afectó a los árboles, tocones o la cubierta vegetal del sotobosque. De 30 especies de aves monitoredas, 4 fueron afectadas negativamente por la quema y 2 fueron afectadas positivamente. Las densidades de población de *Seiurus aurocapillus*, de *Helmitheros vermivorus *y de *Wilsonia citrina *declinaron incrementalmente en respuesta a quemas repetidas y no se recuperaron en un año después de la quema, sugiriendo un retraso en el tiempo de respuesta a los cambios en las condiciones del hábitat. Las densidades de *Cardinalis cardinalis *fluctuaron entre años en las unidades control, pero permanecieron bajas en las unidades quemadas. Las densidades de *Turdus migratorius *y de *Contopus virens *aumentaron en re-*

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spuesta a la quema, pero estos incrementos fueron evidentes sólo hasta varios años después de quemas repetidas. En general, en el corto plazo la quema resultó en reducciones en la calidad del hábitat para aves que anidan sobre el suelo y en arbustos bajos, pero mejoró el hábitat para aves que forrajean en el suelo y el aire. En general, no hubo cambios en la composición de la comunidad de aves reproductivas. Los niveles totales de poblaciones de aves reproductivas tampoco fueron afectados por la quema. Nuestros resultados sugieren la posibilidad de que la quema prescrita aplicada a largo plazo o en escalas espaciales grandes tenga efectos adversos sobre especies de aves que anidan sobre el suelo y en arbustos bajos, pero la posibilidad de cambios en la composición de la comunidad de aves reproductivas es mínima.

Introduction

The composition of deciduous forests in the eastern United States is gradually shifting from dominance by oaks (*Quercus* spp.) to increasing proportions of red maple (*Acer rubrum*) and other tree species (Griffith et al. 1993; Abrams 1998). Oaks have been an important component of eastern deciduous forests for 10,000 years (Abrams 1992). Periodic fires apparently favored regeneration and growth of oaks over fire-intolerant maples and beech (*Fagus grandifolia*), but with the onset of fire suppression in the 1940s, forest composition has been shifting gradually to fire-intolerant species (Abrams 1998). Prescribed burning treatments are being applied on an experimental basis to determine their effectiveness in restoring and maintaining oak-dominated forests (Abrams 1992; Van Lear & Watt 1993). Fires reduce competition from other plant species and thus redirect nutrients and growing space to oaks. There is concern not only about maintaining oaks as the dominant tree species in the forest, but also about maintaining interconnected processes and characteristics relating to oaks and fire, including soil productivity, wildlife population viability, and understory vegetation diversity (Sutherland 1994; Healy et al. 1997). If the oak forest type is declining, then other components of the community are probably being affected as well.

One aspect of particular concern is the forest-bird community. Populations of a number of forest-bird species, particularly Neotropical migrants, have been declining over the last 25 years (Peterjohn & Sauer 1994). Large tracts of forest habitat are important for a number of bird species (Robbins et al. 1989), but the type of forest appears to be less critical. Analyses of Breeding Bird Census data have shown that the composition of breeding bird communities does not differ between mixed-oak and beech-maple forest types (Artman 2000) and that bird population levels tend to be lower in oak forests than in beech-maple forests (Udvardy 1957; Probst 1979). Although some resident bird species such as the Wild Turkey and Blue Jay consume acorns, most Neotropical migratory species are insectivorous and are not dependent on acorns or other resources provided specifically by oaks (scientific names of bird species provided in Table 4). Instead, different bird species require specific structural features within the forest, such as large trees, dense understory vegetation, or thick leaf litter, which provide suitable nest sites or foraging areas. Fire has the potential to change the availability of some or all of these features and thus affect the associated community of forest birds.

There are currently no long-term data on how burning affects forest-bird populations. We describe changes in habitat conditions and bird population levels in response to prescribed burning in closed-canopy, mixed-oak forests. We quantified changes in response to repeated burning (1–4 years of annual burning) and recovery (1 year after burning), with comparisons to preburn conditions and to areas that remained unburned. The effects of prescribed burning on nesting success and nest site selection are examined elsewhere (Artman 2000).

Methods

Study Areas

Our four study areas are located in the Wayne National Forest (Bluegrass and Youngs Branch) and the Vinton Furnace Experimental Forest (Arch Rock and Watch Rock) in southcentral Ohio (Fig. 1a). The study areas are highly dissected by ridges and drainages (Fig. 1b), with elevations ranging from 200 to 300 m. The forest cover is dominated by a mix of oaks and hickories (*Carya* spp.) ranging from 80 to 120 years old. Common midstory species include red maple, American beech, and flowering dogwood (*Cornus florida*). Common understory species include greenbrier (*Smilax rotundifolia*), spicebush (*Lindera benzoin*), and mapleleaf viburnum (*Viburnum acerifolium*).

Experimental Design

We used a randomized block design to test for effects of burning, with each of the four study areas representing a block. Each study area was divided into three treatment units: control, frequent burn, and infrequent burn (Fig. 1b). The size of the treatment units ranged from 20 to 30 ha. The original treatment-unit boundaries were es-

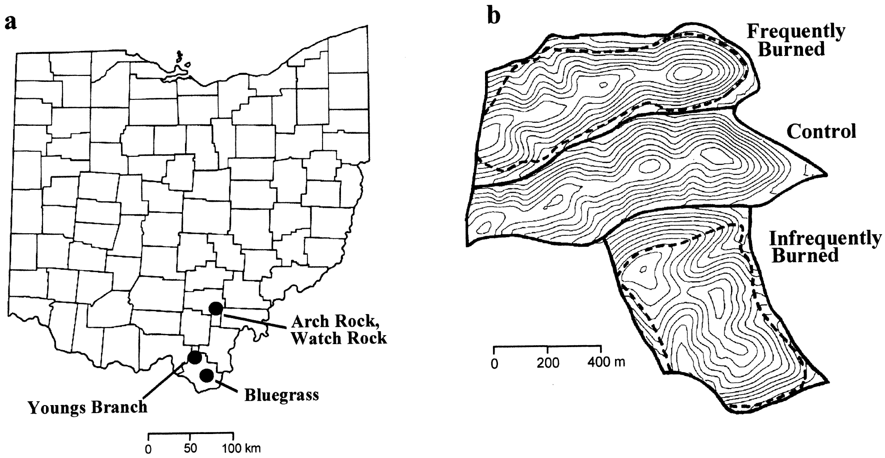

Figure 1. (a) Location of study areas in Ohio and (b) topographic map of Arch Rock study area. Topographic lines are 20-foot (6.1-m) elevation contours. Solid heavy lines are treatment-unit boundaries, and dashed lines are firelines.

tablished in 1994, pre-treatment data were collected in 1995, and firelines were established inside the original boundaries prior to burning in 1996 (Fig. 1b). Firelines were not established in the control units but were designated on maps of control units, following a similar approach for the designation of firelines in the burned units. The treatments consisted of multi-year regimes of burning and recovery: frequent units were burned 4 years in a row from 1996 to 1999, infrequent units were burned twice (in 1996 and 1999), and control units were not burned. All prescribed burning was conducted during early spring between late March and early April, before leaf-out and before arrival of most Neotropical migratory bird species.

Fire Temperatures and Fuel Consumption

Researchers with the U.S. Forest Service monitored fire behavior with temperature sensors during the fires and collected leaf litter before and after each fire. Monitoring was conducted at six points along the perimeter of nine rectangular (25 × 50 m) subplots within each treatment unit. Temperature sensors consisted of aluminum tags painted with temperature-sensitive Tempilac paints. Each tag contained seven paint strips that melt at different temperatures, ranging from 38° to 427° C. Several days before burning, tags were placed 25 cm above the ground at each sample point. After burning, the tags were collected and maximum temperatures were estimated from the melted paints. Before and after each fire at each sample point, leaf-litter samples were collected by means of a square metal frame (0.0225 m^2) with a cutting edge. The samples were dried at 70° C for 48 hours and weighed to the nearest 0.1 g.

Vegetation Sampling

Vegetation characteristics were measured in each treatment unit in 1997 and 1999 following the Breeding Biology Research and Monitoring Database (BBIRD) protocol (Martin et al. 1997). We took measurements at sampling points randomly selected from a 200 × 200 m grid. Four subplots were set up at each sampling point: the first subplot was centered on the point; the remaining three were located 30 m from the point, spaced 120° apart, starting in a random direction. Within a subplot with a 5-m radius (78.5 m^2), we estimated percent cover of understory vegetation ($<$50 cm tall) and counted the number of shrubs and saplings by size class (small shrubs and saplings were $>$50 cm tall and $<$2.5 cm diameter at breast height [dbh]; large shrubs and saplings were 2.5–8.0 cm dbh). Within a subplot with a 11.3-m radius (401 m^2), we counted the number of trees and snags that were $\geq$8.0 cm dbh.

We conducted vegetation measurements at two or three sampling points (8–12 subplots) per treatment unit per year. The locations of the sampling points varied between years, so we did not repeatedly sample the same habitat features in 1997 and 1999. Our data from different years are thus assumed to represent independent samples of the individual treatment units. We ran a one-way randomized-block ANOVA (Zar 1984) to compare vegetation characteristics among treatment-per-year combinations (i.e., control 1997 vs. control 1999). Each study area was treated as a block in the analysis. The primary sampling unit in our analysis was each individual treatment unit, such that one value (for each vegetation characteristic) was entered for each treatment-per-year combination, with the value representing a mean from the multiple subplots. We did not collect data on vegetation characteristics prior to burning (in 1995), but data collected independently by researchers with the U.S. Forest Service show that vegetation characteristics did not differ among the treatment units prior to burning (T. Hutchinson, unpublished data).

Bird Surveys

We used the spot-mapping method (Robbins 1970) to survey bird populations. Each treatment unit was surveyed six times per year between mid-May and late June in 1995, 1996, 1997, and 1999. Four different observers conducted the surveys each year. To reduce variability and ensure consistency, all observers were trained in bird identification by sight and sound, and initial surveys were conducted by groups of observers. Each treatment unit was surveyed one time per week over the sample period to avoid bias toward early or late-season birds. To further minimize bias, observers alternated among treatment units and rotated starting points of survey routes. During each survey, observers recorded the location of birds seen or heard on topographic maps. These observations were transferred to composite maps for each bird species, and territories were delineated based on clusters of observations from separate visits. Observers also searched for and monitored active nests in each treatment unit, and locations of nests were added to the composite maps to supplement survey results. We mapped all territories within the boundaries of the original treatment units but in our final analysis, only territories located inside the designated firelines were counted (Fig. 1b). To estimate population densities, we converted the number of territories per treatment unit to the number of bird pairs per 40 ha (100 acres). Densities were estimated only for bird species that occurred in discrete nonoverlapping territories smaller than the treatment units. To assess the responses of bird species using similar portions of habitat, we grouped bird species by nesting zone and foraging zone, following Ehrlich et al. (1988). Nesting zones included ground, low-shrub ($<$1 m), midstory (1–10 m), canopy ($>$10 m), cavity, and rock. Foraging zones included ground, low-shrub ($<$1 m), midstory (1–10 m), canopy ($>$10 m), bole, and aerial.

Because we repeatedly surveyed the entire breeding bird population within each treatment unit from year to year, we assumed that our data did not represent independent estimates of the bird populations. To account for the non-independence of multiple years, we ran a repeated-measures ANOVA (Hatcher & Stepanski 1994) on each bird species, nesting zone, and foraging zone to compare population levels among treatments and years. A significant treatment $\times$ year interaction was interpreted as a significant treatment effect because our experimental design incorporated pre- and post-treatment sampling (Brogan & Kutner 1980). A significant interaction effect thus indicated that the change from pre- to post-treatment conditions differed between burned and unburned (control) treatment units.

Integrated Moisture Index

Because variation in moisture levels and topographic features is likely to influence fire intensity and the resultant magnitude of change in habitat conditions and bird populations, the treatment units were stratified by means of an integrated moisture index (IMI) based on a geographic information system (GIS) and developed for this region by Iverson et al. (1997). The IMI has been used to describe variation in distributions of vegetation types and bird species in southcentral Ohio (Iverson et al. 1997; Dettmers & Bart 1999; Hutchinson et al. 1999). The IMI stratification is based on an integration of slope aspect, position on slope (e.g., ridge vs. bottom of slope), curvature of the landscape (e.g., flat, convex, or concave), and soil type (e.g., shallow upland soils vs. deep silty valley soils). Moisture availability is thus modeled by a combination of exposure to solar radiation (aspect), flow accumulation (position on slope and curvature), and water-holding capacity (soil type). Each of the components is weighted and standardized on a 0–100 scale, and three IMI categories are defined: xeric (IMI score, $<$35), intermediate (35–50), and mesic ($>$50). Xeric areas include ridgetops and south-facing slopes, where direct solar radiation is maximized and soils have low water-holding capacity. Mesic areas include depressions, drainages, and lower slopes, where solar radiation is minimized and soils are capable of storing large amounts of water.

An IMI category was assigned (based on GIS-derived maps provided by the U.S. Forest Service) to each fire-behavior monitoring point to describe differences in fire temperature along moisture gradients. The IMI categories were also assigned to individual bird territories to describe the association of different bird species with moisture gradients. Many territories covered more than one IMI category (i.e., they were half mesic, half intermediate); to account for this variation, we assigned IMI categories to the nearest half-territory. We ran chi-square tests (Zar 1984) to compare observed and expected distributions of bird territories among IMI categories in unburned and burned areas. Expected distributions were calculated according to the existing composition of the four study areas combined (31% xeric, 40% intermediate, and 29% mesic). We ran additional chi-square tests to compare territory distributions between unburned and burned areas and thus to determine whether territory placement shifted in response to prescribed burning. The primary sampling unit in our analysis was each individual territory (or half-territory). Because of small sample sizes for some bird species, counts of territories were pooled across years and treatments for each bird species. "Unburned areas" represented territories in all units in 1995 (preburn) and territories in control units in 1996, 1997, and 1999, whereas "burned areas" represented territories in frequent and infrequent burn units in 1996, 1997, and 1999 (postburn). Our counts of territories by IMI thus do not represent true independent samples for each bird species, because we assigned IMI categories to the nearest half-territory and because, by combining data across years, we probably counted territories of some unknown number of individuals more than once. This approach limits our scope of statistical inference (Hurlbert 1984; Bart et al. 1998); we intend for this analysis to provide a description of associations between bird species and moisture levels, with the results applicable only to interpreting observed changes in population levels of each bird species in response to the prescribed burning treatments.

Results

Fire Temperatures and Fuel Consumption

Fire intensity varied by year, burning regime, and IMI category. During the initial fires in 1996, fire temperatures were hotter in xeric areas and cooler in mesic areas (Table 1). Subsequent fire temperatures in the frequently burned units were more uniform among moisture categories in 1997 but were hotter in mesic areas in 1998 and 1999, showing opposite patterns from the initial fires. In contrast, subsequent fire temperatures in the infrequently burned units in 1999 were similar to the initial fires, with hotter fires in xeric areas and cooler fires in mesic areas. Overall, these second fires burned at hotter temperatures than the initial fires.

The initial fires in 1996 eliminated about 50% of the leaf litter in both the frequently and infrequently burned units (Table 2). Subsequent fires in the frequently burned units resulted in incremental reductions in leaf litter, with only 10% of the baseline leaf litter remaining after the fourth round of fires in 1999. Leaf litter recovered to preburn levels within 2 years after burning in the infrequently burned units, but was reduced to about 20% of initial levels by the second round of fires in 1999.

Table 1. Mean (SE) fire temperatures (°C) 25 cm above ground by year and integrated moisture index (IMI) category.

Year and moisture index	Treatment unit[a] frequently burned	infrequently burned[b]
1996		
xeric	104 (11.8)	129 (18.7)
intermediate	82 (13.5)	87 (11.5)
mesic	66 (10.7)	68 (9.2)
total	85 (7.5)	93 (8.5)
1997		
xeric	62 (8.3)	—
intermediate	79 (7.7)	—
mesic	76 (13.3)	—
total	72 (5.1)	—
1998		
xeric	98 (13.6)	—
intermediate	111 (13.8)	—
mesic	115 (9.3)	—
total	107 (7.5)	—
1999		
xeric	69 (9.0)	146 (19.2)
intermediate	94 (11.9)	143 (16.7)
mesic	106 (11.3)	115 (17.7)
total	88 (6.6)	133 (10.4)

[a]*Frequently burned units were burned in 1996, 1997, 1998, and 1999; infrequently burned units were burned in 1996 and 1999. There were 36 subplots per treatment per year (9 subplots within each of 4 treatment units) subdivided among IMI categories as follows: frequently burned units: xeric* n = 13*, intermediate* n = 14*, and mesic* n = 9*; infrequently burned units: xeric* n = 11*, intermediate* n = 11*, and mesic* n = 14*. Each sample represents a mean based on multiple subsamples (6) at each subplot.*
[b]*Dash indicates no fires.*

Changes in Vegetation Characteristics

Prescribed burning resulted in incremental reductions in the densities of small and large shrubs and saplings (Table 3). Densities of small shrubs and saplings increased 1 year after burning in the infrequent units due to extensive sprouting at the base of topkilled and otherwise damaged trees and saplings.

Other measured habitat characteristics were unaffected by prescribed burning. Densities of trees and snags did not change in response to the low-intensity surface fires. Percent cover of understory vegetation also did not change in response to the fires, because fire-killed stems were rapidly replaced by sprouting or newly germinating plants.

Responses of Individual Bird Species

Of 30 bird species monitored, 4 were negatively affected by prescribed burning (Table 4; Fig. 2). Densities of Hooded Warblers and Ovenbirds declined in response to the initial fires, continued to decline in response to repeated burning, and did not recover to preburn levels

Table 2. Mean (SE) leaf-litter mass (g/m²) before and after burning.

Year and time of sampling	Treatment unit[a] frequently burned	infrequently burned	control
1996			
preburn	419 (20.2)	466 (24.1)	425 (17.3)
postburn[b]	216 (18.5)	226 (27.8)	—
1997			
preburn	334 (18.1)	364 (19.5)	465 (18.3)
postburn	119 (12.5)	—	—
1998			
preburn	309 (12.0)	489 (30.2)	441 (18.2)
postburn	69 (9.2)	—	—
1999			
preburn	334 (19.3)	580 (28.0)	—
postburn	35 (6.9)	74 (13.0)	—

[a]*Frequently burned units were burned in 1996, 1997, 1998, and 1999; infrequently burned units were burned in 1996 and 1999; control units were not burned. There were 36 subplots per treatment per year (9 subplots within each of 4 treatment units). Each sample represents a mean based on multiple subsamples (6) at each subplot. Dash indicates no data collected.*
[b]*There were only 27 subplots for post-burn conditions in frequently burned and infrequently burned units in 1996 because post-burn sampling was not conducted in one study area (bluegrass).*

Table 3. Habitat characteristics (mean and SE) by treatment unit and year.

Treatment[a]	Understory vegetation (% cover)	Stem counts (number/0.01 ha) small shrubs and saplings	large shrubs and saplings	trees	snags
Unburned (control 1997)	37.3 (3.1)	103.5 (13.3)	6.3 (0.6)	4.6 (0.1)	0.5 (0.1)
Unburned (control 1999)	32.3 (2.9)	99.5 (10.7)	6.9 (1.2)	4.4 (0.3)	0.8 (0.1)
1 year post-burn (infrequent 1997)	32.0 (7.1)	70.6 (23.7)	4.1 (0.9)	4.2 (0.2)	0.6 (0.1)
Recently burned, 2 burns in 4 yrs (infrequent 1999)	36.1 (6.1)	17.8 (1.7)	2.1 (1.0)	4.5 (0.2)	0.4 (0.1)
Recently burned, 2 burns in 2 yrs (frequent 1997)	35.7 (3.7)	23.5 (10.0)	3.4 (0.7)	3.9 (0.2)	0.7 (0.2)
Recently burned, 4 burns in 4 yrs (frequent 1999)	33.3 (5.8)	9.4 (0.5)	1.6 (0.3)	4.3 (0.3)	0.8 (0.1)
F^b	0.4	10.0***	11.1***	2.3	2.6

[a]*Control units were not burned; infrequently burned units were burned in 1996 and 1999; frequently burned units were burned in 1996, 1997, 1998, and 1999. There were four replicates of each treatment and year combination. Each sample represents a mean based on multiple subsamples (8–12) within each treatment unit.*
[b]*Based on one-way randomized-block ANOVA for effect of treatment. Critical value:* $F_{0.05(1),5,15} = 2.90$ *(Zar 1984)* (***$p < 0.001$).

within 1 year after burning in the infrequent units (Fig. 2a & 2b). Densities of Worm-eating Warblers did not decline initially in response to the first fires but declined following subsequent fires in both the frequent and infrequent units (Fig. 2c). Densities of Northern Cardinals fluctuated among years in the control units, peaking in 1999 but remaining low in the frequent and infrequent units (Fig. 2d).

Two bird species responded positively to prescribed burning (Table 4). American Robins were rarely observed before burning or in the control units. Densities increased slightly after the initial fires in 1996, declined in 1997, and increased to higher levels in 1999 after 4 successive years of burning in the frequent units (Fig. 2e). Eastern Wood-Pewees were common before burning and in the control units. Densities increased from 1995 to 1997 in all units; they continued increasing in 1999 in burned units but declined slightly in control units (Fig. 2f).

Densities of 11 bird species varied significantly by year (Table 4): 6 species increased in abundance from 1995 to 1999 (Blue-gray Gnatcatcher, Carolina Wren, Indigo Bunting, Northern Flicker, Yellow-billed Cuckoo, and Yellow-throated Vireo); 2 declined in abundance from 1995 to 1999 (Acadian Flycatcher and Wood Thrush); and 3 varied from year to year with no consistent trends (Carolina Chickadee, Red-eyed Vireo, Scarlet Tanager). No species showed significant treatment effects independent of year effects. Thirteen bird species were unaffected by prescribed burning or by year (Table 4).

Distributions of Bird Species by IMI

Ten bird species showed nonrandom distribution of territories with respect to IMI, with no shifts in distribution in response to prescribed burning. Six bird species, including the Worm-eating Warbler, occurred more frequently in mesic areas than xeric areas (Table 5). Three bird species, including the Eastern Wood-Pewee, occurred more frequently in xeric and intermediate areas than mesic areas. The American Robin also occurred more frequently in xeric areas in burned units, but data were insufficient to assess preferences in unburned units.

One species, the Hooded Warbler, showed a significant shift in territory distribution in response to prescribed burning (Table 5). In unburned areas, Hooded Warbler territories were evenly distributed among IMI categories, whereas in burned areas territories were disproportionately placed in mesic areas.

Community-Level Changes

Densities of ground- and low-shrub-nesting birds declined significantly in response to burning (Table 6; Fig. 3a). No treatment-related changes were detected for other nest-site groups. Trends for groups of species categorized by foraging site were similar to trends for those categorized by nest site, with significant declines in densities of ground- and low-shrub-foraging birds (Table 6; Fig. 3b) and no treatment-related changes in other groups.

Total breeding bird densities did not change significantly in response to burning (Table 6). There were no changes in species richness as a result of prescribed burning. No bird species disappeared or invaded the areas in response to prescribed burning. Carolina Wrens invaded the study areas in 1999, but this change appeared to be unrelated to burning regime, as low population densities were observed in both unburned and burned areas.

Discussion

Prescribed burning is being introduced to ensure the continued dominance of the mixed-oak forest type in the central hardwood landscape. In our study, prescribed burning was applied as low-intensity surface fires during early spring in closed-canopy forests to promote the regeneration and growth of oaks in the understory. Our data show that, within the short term, burning reduced the amount of leaf litter and the density of

Table 4. Baseline densities (mean and SE; pairs per 40 ha) of breeding bird populations in unburned areas in 1995, and results of repeated-measures analysis of variance for effects of treatment (control, frequent burn, and infrequent burn) and year (1995, 1996, 1997, and 1999).

Bird species by nest site[a]	Foraging site[b]	Baseline density	F[c] treatment × year	F[c] treatment	F[c] year
Ground					
Black-and-white Warbler (*Mniotilta varia*)	B	3.3 (0.8)	2.3	0.8	1.1
Carolina Wren (*Thryothorus ludovicianus*)	G	0 (0)[d]	0.7	0.6	12.3***
Kentucky Warbler (*Oporornis formosus*)	LS	0.4 (0.2)	0.8	0.5	0.3
Louisiana Waterthrush (*Seiurus motacilla*)	G	0.4 (0.3)	2.1	2.9	0.1
Ovenbird (*Seiurus aurocapillus*)	G	38.9 (0.9)	12.0***	26.3***	43.6***
Worm-eating Warbler (*Helmitheros vermivorus*)	LS	8.0 (1.0)	7.2***	6.4*	7.2***
Low shrub					
Hooded Warbler (*Wilsonia citrina*)	LS	7.2 (0.8)	6.1***	13.7**	4.3**
Indigo Bunting (*Passerina cyanea*)	LS	0.1 (0.1)	0.8	0.4	5.3**
Midstory					
Acadian Flycatcher (*Empidonax virescens*)	A	12.4 (1.4)	0.7	1.0	4.9**
American Redstart (*Setophaga ruticilla*)	M	1.3 (0.7)	0.9	0.1	1.1
American Robin (*Turdus migratorius*)	G	0 (0)[e]	5.0***	5.7*	11.4***
Northern Cardinal (*Cardinalis cardinalis*)	M	0.9 (0.4)	4.4**	1.5	5.2**
Red-eyed Vireo (*Vireo olivaceus*)	M	39.1 (2.0)	1.0	2.1	4.1*
Wood Thrush (*Hylocichla mustelina*)	G	26.2 (2.0)	0.6	4.1	4.7**
Yellow-billed Cuckoo (*Coccyzus americanus*)	M	2.4 (0.3)	2.0	0.3	16.5***
Canopy					
Blue-gray Gnatcatcher (*Polioptila caerulea*)	C	3.6 (0.4)	0.4	0.2	4.9**
Cerulean Warbler (*Dendroica cerulea*)	C	8.9 (2.2)	1.0	0.2	0.8
Eastern Wood-Pewee (*Contopus virens*)	A	6.0 (0.8)	4.6**	0.0	30.0***
Scarlet Tanager (*Piranga olivacea*)	C	12.2 (0.6)	1.2	2.2	3.3*
Summer Tanager (*Piranga flava*)	C	0.5 (0.3)	0.6	0.5	0.6
Yellow-throated Vireo (*Vireo flavifrons*)	C	3.9 (0.6)	1.3	1.4	11.1***
Cavity					
Carolina Chickadee (*Poecile carolinensis*)	C	1.4 (0.4)	2.0	0.0	3.8*
Downy Woodpecker (*Picoides pubescens*)	B	1.8 (0.3)	0.4	0.3	1.8
Great Crested Flycatcher (*Myiarchus crinitus*)	A	1.0 (0.4)	1.4	0.2	2.7
Hairy Woodpecker (*Picoides villosus*)	B	1.1 (0.3)	1.0	1.0	2.4
Northern Flicker (*Colaptes auratus*)	B	0.5 (0.3)	0.5	0.8	7.8***
Red-bellied Woodpecker (*Melanerpes carolinus*)	B	4.4 (0.4)	0.7	0.0	1.4
Tufted Titmouse (*Baeolophus bicolor*)	C	5.9 (0.6)	1.9	0.4	1.2
White-breasted Nuthatch (*Sitta carolinensis*)	B	6.9 (0.7)	0.4	0.1	2.7
Rock					
Eastern Phoebe (*Sayornis phoebe*)	A	2.5 (0.7)	0.8	0.0	2.4

[a] *The following additional bird species were observed during surveys, but their densities were not estimated because species was rarely observed, did not defend discrete nonoverlapping territories, or territory size was larger than treatment unit: American Crow (*Corvus brachyrhynchos*), Blue Jay (*Cyanocitta cristata*), Brown-headed Cowbird (*Molothrus ater*), Pileated Woodpecker (*Dryocopus pileatus*), Ruby-throated Hummingbird (*Archilochus colubris*), and Wild Turkey (*Meleagris gallopavo*).*
[b] *Foraging sites: A, aerial; B, bole; C, canopy; G, ground; LS, low shrub; M, midstory.*
[c] *Based on repeated-measures ANOVA. Critical values of F: treatment × year effect, $F_{0.05(1),6,27} = 2.46$; treatment effect, $F_{0.05(1),3,27} = 2.96$; year effect, $F_{0.05(1),2,9} = 4.26$ (Zar 1984) (*$p \leq 0.05$; **$p \leq 0.01$; ***$p \leq 0.001$).*
[d] *Baseline density of Carolina Wrens was 0.00 (0.00).*
[e] *Baseline density of American Robins was 0.04 (0.03).*

shrubs and saplings. Repeated burning resulted in incremental reductions in leaf litter, shrubs, and saplings. Within 1 year after burning, recovery of leaf litter to preburn levels was underway; within 2 years after burning, the amount of leaf litter exceeded preburn levels.

Bird Species Negatively Affected by Prescribed Burning

Four bird species associated with these habitat features, the Ovenbird, Worm-eating Warbler, Hooded Warbler, and Northern Cardinal, were negatively affected by prescribed burning. Ovenbirds were one of the most abundant and widespread species prior to burning, with population levels close to one pair per hectare. Ovenbirds nest on the ground in areas where a thick layer of leaf litter, which they use for construction and concealment of nests, is available. Ovenbird populations declined incrementally in response to repeated burning, in parallel with the incremental reductions in leaf litter. Although the availability of leaf litter increased within 1 year after burning, Ovenbird densities did not recover to preburn levels, suggesting a lag time in the populations' responses to changes in habitat conditions. Ovenbirds were not eliminated from burned areas but continued to occur at

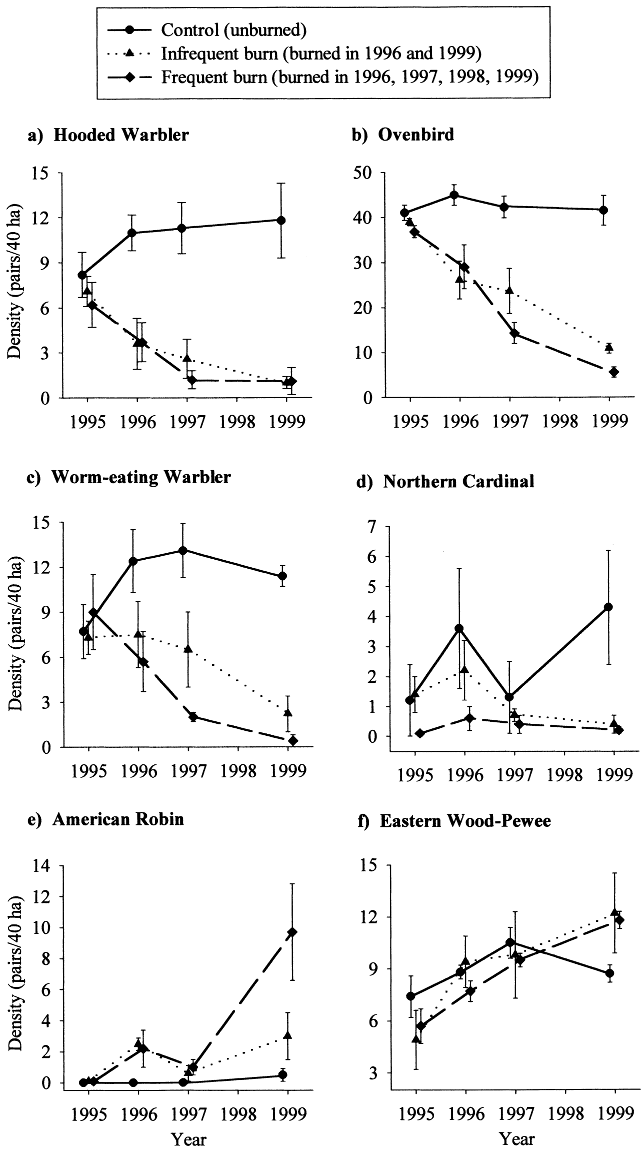

Figure 2. Mean (SE) densities of bird species negatively (a-d) and positively (e-f) affected by prescribed burning (1995 represents pre-treatment conditions).

low densities even after 4 successive years of burning. Many territories (11 of 20) remaining in 1999 in frequently burned units straddled the fireline. We found several nests located just outside the fireline in unburned habitat, and we observed adults and juveniles foraging in both burned and unburned portions of these territories (V.L.A., unpublished data). Ovenbirds thus persisted in recently burned areas as long as suitable nesting habitat remained available in adjacent unburned areas. Other territories (9 of 20) in frequently burned units in 1999 were located entirely in burned areas. We did not observe any shift in location among these territories with respect to IMI, indicating that Ovenbirds continuing to inhabit recently burned areas were not restricted to moist areas. Other characteristics of these territories were not assessed, but potential changes include shifts in territory size (Steele 1992), reductions in mating success (Burke & Nol 1998), or reductions in nesting success (Brooker & Rowley 1991). The potential for such changes suggests that prescribed burning may result in additional reductions in habitat quality for Ovenbirds beyond that indicated by the observed declines in population levels.

Worm-eating Warblers also nest on the ground but do not require leaf litter for construction or concealment of their nests, instead placing their nests in depressions along steep slopes (Gale 1995). They occur primarily along stream bottoms and other areas with high moisture levels (Dettmers & Bart 1999), and in our study were more common in mesic than xeric portions of our treatment units. These nest-site requirements and habitat preferences appeared to mitigate the effects of the initial fires. The initial fires did not burn some areas where Worm-eating Warblers had territories (V.L.A., personal observation), so population levels did not change because habitat conditions did not change. In addition, temperatures of the initial fires in mesic areas were cooler than those of subsequent fires, further minimizing initial effects on habitat of Worm-eating Warblers. Subsequent fires, however, including the second fires in the infrequent units, were hotter and more uniform in mesic areas, and significant declines in Worm-eating Warbler populations occurred as a result. In the aggregate, multiple fires probably shifted the microclimate of mesic areas, making them warmer and drier from the loss of leaf litter and blackening of the soil surface, and further reducing the suitability of the habitat for Worm-eating Warblers.

Hooded Warblers typically nest within 1 m of the ground in dense shrub thickets (Evans Ogden & Stutchbury 1994). Fires eliminated most of the dense shrub thickets, particularly those dominated by greenbrier on dry ridges and hillsides. Hooded Warbler populations declined in response to these changes and did not recover to preburn levels within 1 year after burning, despite some recovery (sprouting and germination) of shrubs and seedlings. Although Hooded Warblers are characteristically hilltop species associated with dry ridges (Dettmers & Bart 1999), we found that territories in unburned areas were distributed evenly among moisture categories. After burning, territories were located disproportionately in mesic areas, suggesting that higher moisture levels mitigated the effects of prescribed burning on the suitability of habitat for Hooded Warblers. Also, several Hooded Warbler territories were located in small patches of unburned habitat within the burned units; natural firebreaks such as downed logs or bare ground prevented spread of the fire into such patches (V.L.A., personal observation). Habitat availability was thus maintained on a limited basis by variation in local habitat characteristics and fire intensity.

Table 5. Distributions of breeding bird territories by integrated moisture index (IMI) in unburned and burned treatment units.

		Proportion of territories by IMI				
Species	*Treatment* [a]	*xeric*	*intermediate*	*mesic*	N [b]	χ^2 *values* [c]
Mesic areas						
Acadian Flycatcher	unburned	0.04	0.22	0.73	181	178.6***
	burned	0.02	0.23	0.75	145	155.6***
American Redstart	unburned	0	0.04	0.96	13	27.5***
	burned	0	0.18	0.82	8	11.8**
Eastern Phoebe	unburned	0.13	0.23	0.63	26	15.1***
	burned	0.08	0.25	0.67	24	17.1***
Kentucky Warbler	unburned	0.09	0.19	0.72	10	9.0*
	burned	0.05	0.13	0.82	4	5.4
Louisiana Waterthrush	unburned	0	0.05	0.95	10	20.9***
	burned	0	0	1.00	2	4.9
Worm-eating Warbler	unburned	0.12	0.35	0.53	148	46.8***
	burned	0.06	0.31	0.63	27	35.5***
Xeric areas						
American Robin [d]	burned	0.49	0.32	0.19	49	8.1*
Eastern Wood-Pewee	unburned	0.46	0.39	0.15	107	14.8***
	burned	0.51	0.38	0.10	151	38.4***
Summer Tanager	unburned	0.83	0.17	0	6	7.4*
	burned	0.78	0.17	0.06	10	12.4**
Yellow-throated Vireo	unburned	0.45	0.35	0.20	59	6.0*
	burned	0.47	0.39	0.14	56	8.6*
Shifts in territory distributions						
Hooded Warbler	unburned	0.35	0.42	0.22	128	13.4[e]***
	burned	0.17	0.33	0.50	30	

[a] *Unburned areas are all units in 1995 (preburn) and control units in 1996, 1997, and 1999. Burned areas are frequently and infrequently burned units in 1996, 1997, and 1999 (postburn).*
[b] *Total number of territories per treatment category for multiple years combined.*
[c] *Chi-square tests compared observed and expected numbers of territories among IMI categories. Expected values were calculated based on composition of four combined study areas: 31% xeric, 40% intermediate, 29% mesic. Critical value:* $\chi^2_{0.05,2} = 5.991$ *(Zar 1984) (*p < 0.05; **p < 0.01; ***p < 0.001).*
[d] *Data for American Robin were insufficient to assess preferences in unburned areas.*
[e] *Compared distributions of territories among IMI categories between unburned and burned areas.*

Northern Cardinals also nest in dense shrub thickets, although higher off the ground (1–2 m) than Hooded Warblers (Halkin & Linville 1999). Northern Cardinals were not directly affected by prescribed burning because they were uncommon in the treatment units prior to burning. Population levels fluctuated among years in the control units, peaking in 1999, but did not fluctuate in the burned units. Our results suggest that Northern Cardinals did not invade the burned units because habitat conditions in these areas were unsuitable.

Bird Species Positively Affected by Prescribed Burning

Population levels of two bird species, the American Robin and Eastern Wood-Pewee, increased as a result of the fires. Prescribed burning appeared to improve foraging habitat for both species. We frequently observed American Robins and other ground-foraging birds, including Ovenbirds, Brown-headed Cowbirds, and Wood Thrushes, feeding in recently burned areas. We expected that prescribed burning would reduce the availability of food resources on the ground, either by incinerating food or by shifting the microclimate toward hotter, drier conditions, leading to reductions in arthropod populations. Indeed, Burke and Nol (1998) found that forest fragmentation led to reductions in food availability for ground-foraging Ovenbirds because reduced litter depth and desiccation of leaf litter near edges apparently reduced the suitability of habitat for litter-dwelling arthropods. Other studies have shown, however, that arthropods vary disparately in response to fire: burning in oak savannas reduced the abundance of Homoptera and Lepidoptera, but did not affect the overall population levels or species richness of arthropod communities (Siemann et al. 1997); burning in pine forests resulted in increased activity levels of carabid beetles (Neumann 1997), suggesting that visibility and accessibility of arthropod prey increases after burning. Research on selected arthropod taxa from our study areas does not indicate that spring burns have a significant negative effect on arthropod abundance or diversity (Stanton 2000; D. Horn, unpublished data). Fires therefore may not reduce the availability of food resources for ground-foraging birds, but may increase food accessibility by removing leaf litter, brush, and dense vegetation, exposing both insects and seeds (Woinarski 1990).

Table 6. Baseline densities (mean and SE; pairs per 40 ha) of breeding bird populations in unburned areas in 1995, by nest site and foraging site, and results of repeated-measures analysis of variance for effects of year (1995, 1996, 1997, and 1999) and treatment (control, frequently burned, and infrequently burned).

		F[a]		
	Baseline density	*treatment × year*[b]	*treatment*[b]	*year*[b]
Nest site				
ground	51.1 (1.8)	12.8***	17.1***	25.7***
low shrub	7.3 (0.8)	4.3**	13.7**	1.8
midstory	82.3 (4.4)	0.8	3.8	4.8**
canopy	35.1 (2.7)	0.2	0.4	11.0***
cavity	23.2 (1.9)	1.7	0.0	13.1***
rock	2.5 (0.7)	0.8	0.0	2.4
Foraging site				
ground	65.6 (2.7)	3.0*	12.2**	17.0***
low shrub	15.7 (1.4)	9.0***	12.1**	5.9**
midstory	43.7 (2.2)	1.5	2.4	9.6***
canopy	36.4 (2.7)	0.2	0.6	6.6***
bark	18.1 (1.5)	0.3	0.1	5.5**
aerial	21.9 (2.4)	2.0	0.3	1.8
Total	201.4 (9.0)	2.4	4.1	3.7*

[a]*Critical values of* F*: treatment × year effect,* $F_{0.05(1),6,27} = 2.46$*; treatment effect,* $F_{0.05(1),3,27} = 2.96$*; year effect,* $F_{0.05(1),2,9} = 4.26$ *(Zar 1984).*
[b]$*p \le 0.05$*;* $**p \le 0.01$*;* $***p \le 0.001$*.*

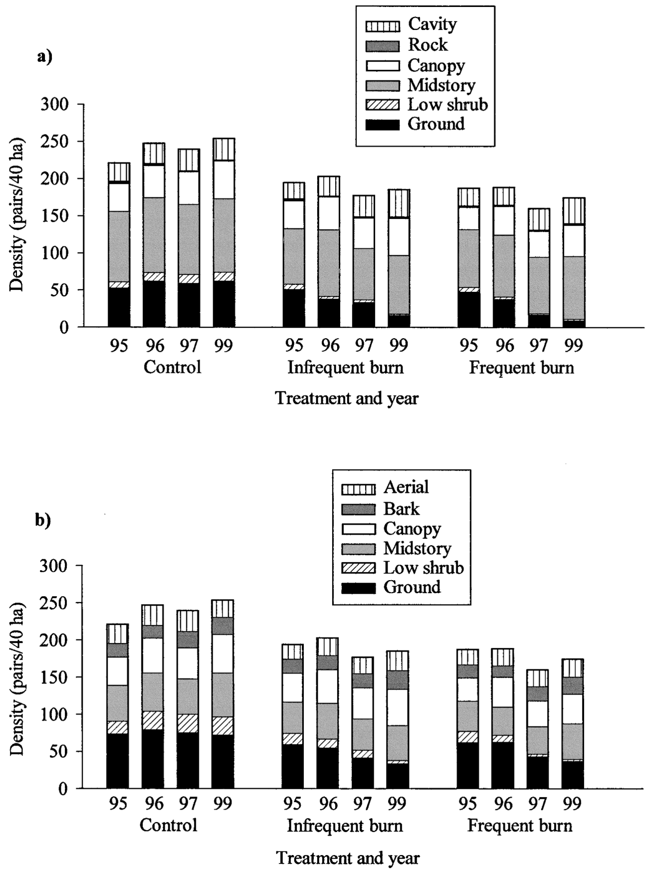

Figure 3. Mean densities of bird species grouped by (a) nesting zone and (b) foraging zone: 1995, pretreatment conditions; control units not burned; infrequent units burned in 1996 and 1999; frequent units burned in 1996, 1997, 1998, and 1999.

Eastern Wood-Pewees are aerial feeders, preferring to forage from exposed perches in the midstory (McCarty 1996). Prescribed burning reduced the density of shrubs and saplings, creating more open and park-like conditions, apparently enhancing the suitability of foraging habitat for pewees. But Eastern Wood-Pewees were common in unburned units (although densities declined slightly in 1999), suggesting that these areas also provided suitable habitat conditions.

Comparison with Other Studies

Our results are consistent with those of other studies documenting changes in bird populations in response to fire and similar forest-management practices. Rodewald and Smith (1998) studied the effects of removing understory and midstory vegetation on breeding birds in oak-hickory forests in Arkansas, although leaf litter was not directly affected by the treatments. They reported that in treated areas, Ovenbirds and Worm-eating Warblers were less abundant, Hooded Warblers were replaced by Indigo Buntings, and Eastern Wood-Pewees were more abundant. A combination of prescribed burning and thinning in oak-pine forests in Arkansas resulted in declines in populations of Ovenbirds and Black-and-white Warblers and increased populations of Eastern Wood-Pewees and Indigo Buntings (Wilson et al. 1995). Surface fires in coniferous forests in eastern California were detrimental to ground- and low-shrub nesters, including the Hermit Thrush (*Catharus guttatus*), but were beneficial to aerial foragers, including the Western Wood-Pewee (*Contopus sordidulus*) (Kilgore 1971; Granholm 1982). Attraction of ground-foraging birds to recently burned areas has been observed in response to both prescribed burning and wildfires (Kilgore 1971; Apfelbaum & Haney 1981; Bock & Bock 1983). Indeed, prescribed burning is recognized as an appropriate management strategy to improve habitat for ground-foraging gamebirds such as Wild Turkeys and Ruffed Grouse (*Bonasa umbellus*) (Rogers & Samuel 1984; Palmer et al. 1996).

Conservation Implications

Overall, we observed no changes in the composition of the breeding bird community: no species was eliminated or added as a result of prescribed burning. Total breeding bird population levels were also unaffected by burning. But the objective of the prescribed burning was to maintain the existing mixed-oak community, not to restore a previously existing community type. In other regions, prescribed burning is being implemented to convert closed-canopy forests into a mix of forest and open grasslands to create savanna-like conditions. These treatments result in more substantial changes in the associated bird communities, increasing population levels of bird species associated with more open areas (Wilson et al. 1995; J. Brawn, unpublished data).

Our results suggest, however, that if prescribed burning is applied on a frequent basis or across large spatial scales in mixed-oak forests of southern Ohio, shifts in the composition of the breeding bird community may result from declines in population levels of ground- and low-shrub-nesting bird species such as the Ovenbird, Hooded Warbler, and Worm-eating Warbler. Populations of these three bird species declined to less than 20% of baseline levels after 4 years of repeated burning, and similar declines were observed when burning treatments were applied less frequently (two fires in 4 years). In addition, we found that 1 year was insufficient for populations of these bird species to recover from the initial fires. Ovenbird and Black-and-white Warbler populations did not recover within 2 years after burning and thinning in Arkansas oak-pine forests (Wilson et al. 1995), but that combination of treatments created more gaps in the overstory, resulting in a shift in the composition of the bird community from forest-interior species to more edge and early-successional species. Other studies have not addressed the issue of recovery of bird populations after prescribed burning or other forest-management practices, so the amount of time necessary for populations to recover is unknown. Such recovery may depend on the interval between fires, the amount of time necessary for habitat conditions to recover, the lag time necessary for bird populations to respond, and the presence of surplus individuals in surrounding areas to repopulate recovering areas. Burning treatments may be reapplied at our study areas once every 5 years in the future, thus providing opportunities to address the extent of recovery of habitat conditions and bird populations when the interval between fires is increased.

Despite effects on ground- and low-shrub-nesting bird species, our results indicate that other breeding bird species would be relatively unaffected by long-term or large-scale burning regimes, as long as the closed-canopy forest structure is maintained. The goal of prescribed burning has been to restore and maintain various components of the mixed-oak forest community. It is impossible to speculate about whether prescribed burning would restore the composition of the breeding bird community to what it was when fires were more prevalent in the region, because no data are available on long-term changes in breeding bird populations.

Forest-bird populations in the region undoubtedly have undergone massive retractions and declines during the last 100 years in response to land-use practices such as timber harvesting, agricultural cultivation, and fire suppression (Franzreb & Rosenberg 1997). In general, prescribed burning is unlikely to be applied on a widespread basis in the region, given specific management objectives, economic costs, and limitations imposed by landowners. Most forest lands in Ohio are small, privately owned parcels (Ervin et al. 1994), so substantial coordination among landowners would be necessary to facilitate burning across larger spatial scales. In addition, prescribed burning treatments are unlikely to be applied on a frequent basis in the region, thus minimizing the potential for incremental reductions in breeding bird populations and providing opportunities for populations to recover. Nevertheless, our research is of substantial value for deciding whether or not to use prescribed burning as a restoration tool, particularly when considering the need to balance multiple resources and potentially conflicting uses within the context of long-term forest management.

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