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Associations of avian and herpetofauna communities with forest management at multiple spatial scales

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ASSOCIATIONS OF AVIAN AND HERPETOFAUNA COMMUNITIES WITH FOREST MANAGEMENT AT MULTIPLE SPATIAL SCALES

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ABSTRACT

Decline of amphibians, reptiles, and numerous Neotropical migrant birds has been attributed to habitat destruction and alteration, which warrants examination of these groups in managed forests and their association with habitat characteristics at multiple spatial scales. We surveyed avifauna and herpetofauna communities in 3 managed forests in Louisiana during 2003-2004. Study areas included Sherburne Wildlife Management Area (WMA), a bottomland hardwood forest under uneven-aged management, Ben's Creek WMA, an even-aged, short-rotation loblolly pine plantation, and Sandy Hollow WMA, a longleaf pine-savannah maintained with prescribed fire. Field techniques included surveys consisting of avian point counts, drift fence arrays (PFFT), cover boards, visual encounters, anuran calls (ACS), and microhabitat. We derived landscape variables with GIS landcover maps and ArcView GIS 3.3. General trends included the following: PFFT and ACS accounted for the greatest percentage of detections among herpetofauna surveys, and results primarily reflect these efforts. Anuran calling surveys made a substantial contribution to total number species of detected. Species of conservation concern were among detections of both early- and late-successional bird species. At Sherburne, abundance and richness of amphibians, and occurrence of late-successional birds were greater in uncut and individual-selection stands, whereas occurrence of early-successional birds was greater in recent selection cuttings with groups. Abundance of reptiles did not differ across stand type. At Ben's Creek, abundance and richness of anurans was greater in 1-year and 11-23-year stands, whereas abundance and richness of lizards was similar across stand age. Late-successional bird species occurred with greater frequency in 11-23-year stands at Ben's Creek, whereas frequency of occurrence of early-successional bird species was greater in 1-year and 4-5-year stands. At Sandy Hollow, abundance of reptiles was greater than amphibians, and occurrence of avifauna was similar to pine-savannah ecosystems elsewhere. Responses to

habitat factors at all scales were species specific. In general, canopy closure and shrub cover were the most frequent predictors of occurrence at the microhabitat scale. At the landscape scale, canopy closure and streamside management zones were important predictors of occurrence at Ben's Creek, whereas openings and shape complexity of longleaf pine and longleaf savannah were frequent predictors of occurrence at Sandy Hollow. Effects of selection cutting and stand age appear to benefit certain species, including species of conservation concern, but are potentially costly for other species. Efforts to combine management of timber with conservation of amphibians, reptiles, and songbirds must take into consideration both the complexity of habitat requirements of species within these groups and the landscape context in which these requirements occur.

CHAPTER 1. INTRODUCTION

INTRODUCTION

Wildlife communities are increasingly influenced by both changes in land use and forest management (Turner et al. 2002). Recently, the combined influence of multiple ecological factors, each operating at different spatial scales, has been recognized to affect the structure and dynamics of wildlife communities. Stand- and landscape-level factors, such as forest age, plant composition and structure, size and shape of tract, distance to water, topography, and spatial relationships with other habitats, often interact with components of wildlife habitat, including food, cover, and substrates for activities (Cromer et al 2002; DeMaynadier and Hunter 1998; Dupuis et al 1995; Hagan and Meehan 2002; Herrando and Brotons 2002; Lewis et al. 2000; Seoane et al 2004). The effects of some factors may vary by scale, whereas others may be scale invariant (Bohning-Gaese 1997). As a result, much recent research has focused on multiple scales, incorporating large areas and multiple species. Forest management activities, such as timber harvesting, also can influence each of these habitat components (Wigley and Roberts 1994). In fact, the influence of factors at different scales on species occurrence may be more relevant in landscapes that provide a continuum of habitat types, such as managed forests (Hagan and Meehan 2002). Aspects of amphibian ecology, including aquatic and terrestrial life cycles, small home ranges, and limited dispersal ability, suggest that impacts of forest management practices on microhabitat may affect amphibian communities before such impacts are apparent in other vertebrate classes (deMaynadier and Hunter 1995). However, in a recent review of scientific literature on issues related to forest management and wildlife, DeStefano (2002) found that studies of herpetofauna and forest management accounted for only 10% of all research papers, and therefore, predicting the effects of forest management on herpetofauna is difficult at best. Similarly, because bird communities are considered ecological indicators of forest

condition (Canterbury et al. 2000), they also may be affected by changes in habitat brought about by forest management. Neotropical migrant birds may be especially sensitive to landscape changes that compromise the spatial continuity and integrity of natural ecosystems (Maurer 1993). The current decline of numerous Neotropical migrant bird species (Robbins et al. 1993), amphibians (Wyman 1990), and reptiles (Pough et al. 1998) has been attributed to habitat destruction and alteration. Examining relationships between habitat characteristics at multiple spatial scales and avian and herpetofauna communities within managed forests is necessary.

Forest management techniques throughout the Southeast continue to evolve and develop. Because bottomland hardwood forests are vital to the ecology of numerous wildlife species, as well as a valuable source of timber, these forests are commonly managed for timber production and wildlife through the concept of multiple-use management (Wigley and Roberts 1994). This concept incorporates a variety of management strategies, including even-aged and uneven-aged management. Clearcutting is the most common even-aged reproduction method, and often is preferred because many merchantable tree species such as Nutall (*Quercus texana* Palmer), laurel (*Q. laurifolia* Michaux) and overcup (*Q. lyrata* Walt.) oak are shade intolerant, though clearcutting is not restricted to hardwoods. Examples of uneven-aged regeneration methods include group selection, in which similar treatments are applied to groups of trees similar in size and growth, and single-tree selection, in which individual trees are removed to create a specific stand structure and species composition (Kellison and Young 1997).

Such forest management practices in bottomland hardwood forests potentially influence temporal and spatial changes in wildlife diversity and abundance. Dahl (1990) reported that the southern wetland forest base declined 96% from 45 million hectares (ha) in 1780 to 23 million ha in 1980. Given that bottomland hardwood forests have been drastically altered and reduced in area over the last 40 years, primarily through timber harvesting (Stanturf 1994), examining the

consequences of managing such forests under the multiple-use management concept is timely and relevant.

Louisiana converted 960,279 ha of timberland to agriculture between 1936 and 1996, 96% of which was in the Mississippi Alluvial Valley (Rosson 1995). The Atchafalaya River Basin (ARB), part of the Lower Mississippi Alluvial Valley and located in the central and southern half of Louisiana, contains the largest contiguous bottomland hardwood forest in North America, and it contains the largest overflow alluvial hardwood swamp (323,749 ha) remaining in the United States. The ARB covers approximately 566,560 ha and the Atchafalaya River is 217 kilometers long from north to south (Reuss 1988). The ARB is known to have 53 species of reptile and 28 species of amphibian (Lockwood 1981). It also is home to over 200 species of resident and migratory birds, including species of Neotropical migrant songbirds that use Louisiana wetlands for resting and feeding habitat during migration. Approximately 60% of the migratory species in the North American flyway use the ARB annually (National Audubon Society 1999). Furthermore, forest management of bottomland hardwoods occurs within the basin, making it an important location to examine relationships between management strategies and avian and herpetofauna communities at multiple spatial scales.

Changes in forest management with respect to other forest types also are occurring. Timber production of pine, particularly loblolly pine (*Pinus taeda*), in many southern states is moving from the use of natural stands to pine plantations as the major wood producer (Yin and Sedjo 2001). These pine-dominated forests are managed with a variety of techniques, including clearcutting, use of herbicides to control competing vegetation, fertilization, and thinning. The acreage of intensively managed pine forests in the South has increased over the past decade and continues to rise. Pine plantations currently account for 15% (12.5 million ha) of timberland in the South, just short of the natural pine total (Sheffield and Dickson 1998). This is in response to

findings that economic performance is positively related to management intensity, in which the optimal rotation age is 36 years (Allen et al. 1996). Shorter rotations are typical and have become possible through genetic improvements, which produce faster growing trees. However, shorter rotations may reduce biodiversity and litter cover, and deplete nutrients in the soil after several rotations (Gresham 2002).

As most pine plantations often are surrounded by streamside management zones or other land uses, effects on wildlife may be difficult to determine (Harris et al. 2002). In addition to potential impacts on wildlife from a shorter rotation, factors associated with clearcutting, such as establishment practices, and size, shape and juxtaposition of regenerating stands, also have an influence (Thill 1990). Therefore, some wildlife species historically abundant in areas where plantations have replaced natural stands, and traditionally associated with forests of older age classes, may exist at reduced densities, or worse, be unable to utilize younger stands. Therefore, the ability of intermediate-aged stands to support species that are dependent on habitat features typical of mature forests needs to be examined.

In Louisiana, 47% (5.6 million ha) of the total land base is classified as commercial forest land, 20% of which is in plantations (Rosson 1995). Forests comprised of loblolly and shortleaf (*P. echinata*) pine are the prevailing forest types in timber plantations, and accounted for 85% of the total softwood output in 1999 (Bentley et al. 2002). Between 1975 and 1991, over 1 million ha of upland timberland in Louisiana were clearcut. Fifty-eight percent of Louisiana's plantations are less than 20 years old, and very few are over 40 years old (Rosson 1995). Each of Louisiana's 8 Florida parishes contains timberland, some in the form of plantations. In 6 of these parishes, timberland makes up 61-80% of the total area (Rosson 1995).

The once widespread longleaf pine (*P. palustris* Mill) savannahs of the southeastern coastal plain have been considerably reduced. Historically, longleaf pine forests, maintained by

thousands of years of fires initiated by lightning or other causes, that burned through every 2 to 4 years, covered 25 million ha across the southeast (Platt 1988). Burns associated with lightning strikes are believed to have occurred more frequently during the summer between May and August (Hermann et al. 1998). European settlement in the 1700s, widespread commercial timber harvesting, and the naval stores and turpentine market initially reduced longleaf pine forests. This ecosystem was further diminished in the 1900s due to commercial tree farming, urbanization, agriculture, and fire suppression (USFWS 1985). Less than 809,371 ha currently remain, representing a 97% decline in this ecosystem (USFWS 1998). Regeneration of longleaf pine across the southeast is increasingly common through efforts to reestablish the species on former longleaf sites (Sheffield and Dickson 1998). Management of longleaf pine systems and other pine forests includes prescribed burning regimes that vary by fire-return interval (McLeod and Gates 1998; Carter and Foster 2004; Schurbon and Fauth 2003) and season of application (Hiers et al. 2000; Haywood et al. 2001; Boyer 2000).

Louisiana (1 of 8 states with longleaf pine savannahs) is comprised of 343,619 ha of longleaf-slash (*P. elliotti*) pine forest types, 67% of which is planted, and 33% natural longleaf stands (Vissage et al. 1992). Longleaf and slash, either individually or in combination, dominate the system. Longleaf pine savannahs primarily occur in the southwestern portion of the state, where growing conditions are most favorable (Rosson 1995). Tracts of longleaf pine savannahs also are present in the Florida Parishes, which were part of West Florida, a British territory established in 1763. Some of these tracts are managed with a prescribed burning regime that includes both winter and growing season burns (LDWF 2002). At Sandy Hollow Wildlife Management Area, located in Tangipahoa Parish, prescribed burning was primarily conducted during winter (December-March) until 2002, when growing season (April-August) fire was increasingly implemented. Conducting prescribed burns during summer has been determined by

some to be more appropriate, since historic lightning-season fires typically occurred during this time of year (Hiers et al 2000, Schurbon and Fauth 2003). Plant and wildlife species associated with longleaf pine systems may be better adapted to fire disturbance in the summer, which closer mimics past conditions. Results are mixed, however (Haywood et al. 2001), and thus examination of wildlife and habitat responses to seasonal burns in longleaf pine savannahs are needed.

The Florida Parishes of Louisiana support a number of species of herpetofauna, whose distributions within the state are limited to this region. Examples include the long-tailed salamander (*Eurycea longicauda*), southern cricket frog (*Acris gryllus*), and gopher tortoise (*Gopherus polyphemus*), the latter recently listed by the state of Louisiana as critically imperiled (Dundee and Rossman 1989; LNHP 2004). Moreover, the resident, migratory, and breeding season distributions of numerous bird species incorporate the Florida Parish region, including species such as the Pine Warbler (*Dendroica pinus*) and Brown-headed Nuthatch (*Sitta pusilla*), which are closely associated with pine forests, and the Prairie Warbler (*Dendroica discolor*), which is associated with second growth habitat (Sibley 2000).

Amphibians and reptiles in forest ecosystems play a vital role among terrestrial vertebrates (deMaynadier and Hunter 1995). As ectotherms, herpetofauna obtain nearly all of the energy needed for thermoregulation from external sources, and can therefore devote a large segment of their ingested energy to producing new biomass, making them important components of energy pathways in food web dynamics (Pough 1980). In addition, both amphibians and reptiles are believed to be reliable indicators of changes in the environment. Aspects of amphibian natural history, including a biphasic lifestyle that exposes them to both aquatic and terrestrial habitat, small home ranges, and moist, permeable eggs, gills and skin, make them potentially vulnerable to local environmental stress (Welsh and Ollivier 1998). Likewise,

reptiles have been shown to be sensitive to local climatic variations in temperature (Janzen 1994).

In the South, amphibians and reptiles dominate the list of species of conservation concern. Fifty-four amphibians and 40 reptiles are currently classified as imperiled by southern State Natural Heritage agencies (Trani 2002). Habitat destruction and changes in land use have been considered the primary cause of recent herpetofauna population declines (Wyman 1990). Since many herpetofauna are obligate forest dwellers (Duguay and Wood 2002), forest management strategies may substantially affect herpetofauna assemblages. It is generally believed that herpetofauna abundance and richness decline after forests are harvested. For example, Harpole and Haas (1999) found that abundance of salamanders was lower after harvest in 3 out of 4 treatment types relative to control stands. Abundance and richness of amphibians also has been associated with forest age. Aubry (2000) reported that species richness and abundance was highest in the oldest age class (70-80 years old) among 4 distinct age classes across second-growth stands. Steele et al. (2003), however, found that captures of salamanders were greatest in 25- to 60-year-old forests, least in 0- to 24-year-old forests, and intermediate in forests $\geq$ 60 years. Despite acknowledgement that herpetofauna are a vital component of ecological communities and the most abundant vertebrates in many forests, amphibians and reptiles are typically not fully considered in forest management decisions (Russell et al. 2002).

Over the past 3 decades, numerous species of Neotropical migrant landbirds have experienced both regional and continental population declines (Hagan and Johnston 1992, Robbins et al. 1992). Hunter et al. (1993) identified 46 species of Neotropical migrants within the southeastern United States that need increased habitat and population conservation. Similarly, most bird species associated with disturbance-driven forests, such as pine savannahs, have exhibited sharp population declines (Hunter et al. 2001). Declines in breeding densities of

Neotropical migrants and concurrent increases in resident species are best explained by patterns of forest successional change affected by land-use history (Litwin and Smith 1992). The significant loss and degradation of longleaf pine forests in the southeastern United States have resulted in declines of several bird species associated with this ecosystem (Provencher et al. 2002). Bottomland hardwood forests provide essential breeding habitats for a number of Neotropical migrant species, many of which are in decline (Hunter et al. 1993). Forest management may therefore affect avian communities, and coupled with increasing concern over many bird species that utilize forested landscapes, it is critical that relationships between avian communities and the landscapes they inhabit be examined.

Similar to herpetofauna, birds are important components of forest systems. Clout and Hay (1989) reported that 70% of birds are frugivorous and, thus, important seed dispersers in forests of New Zealand. Birds also play a vital role in seed germination. Passage through the digestive tracts of birds is beneficial to the germination of many species of seeds, such that consumption assures both seed dispersal and an increased chance of seedling establishment (Krefting and Roe 1949). Some species of birds also function as primary pollinators for forest plants, which have adapted specifically for pollination by birds (Anderson 2003). Furthermore, birds may be more effective than other vertebrate groups in nutrient cycling in a forest ecosystem. Migratory birds may redistribute nutrients across ecosystem units by deposition of fecal matter at roosting and nesting sites or along flight pathways (Sturges et al. 1974). Additionally, habitat assemblages, in which birds with similar successional preferences are grouped together, can be useful instruments for environmental monitoring. Multiple habitat assemblages can be combined to create a bird-community index, which can detect major disturbances to natural systems, such as reproduction cuts or fragmentation (Canterbury et al. 2000). Finally, distributions of breeding bird species may be ideal estimators of biodiversity,

which would facilitate a way to prioritize areas of high biodiversity. Concentration of efforts could be targeted at places in greatest need of conservation (Garson et al. 2002).

OBJECTIVES

Because the relationship between forest management and herpetofauna and avian communities in Louisiana is poorly understood, the objectives of this study were to (1) examine relative abundance and frequency of occurrence of various avian and herpetofauna species associated with selection cutting and stand age; (2) quantify relationships between microhabitat conditions and the occurrence of selected species of avifauna and herpetofauna; and (3) assess relationships between landscape characteristics and abundance or occurrence of avian and herpetofauna assemblages within 3 forested ecosystems in Louisiana. These ecosystems included a bottomland hardwood forest, a short-rotation loblolly pine forest, and a longleaf pine savannah.

CHAPTER 2. STUDY AREAS AND METHODS

STUDY AREAS

Sherburne Wildlife Management Area

Research was conducted on a 17,652 ha tract of bottomland hardwood forest located in the Morganza floodway system of the Atchafalaya River Basin, which is in the southern Mississippi Valley alluvium region (Rudis 1988). The study area included Sherburne Wildlife Management Area (WMA, 4,767 ha) owned by the State and managed by the Louisiana Department of Wildlife and Fisheries (LDWF), Bayou des Ourses (6,317 ha) owned by the Army Corps of Engineers and the Atchafalaya National Wildlife Refuge (6,159 ha) owned by the U. S. Fish and Wildlife Service. Hereafter, the study area will be referred to as Sherburne or Sherburne WMA. Sherburne is situated in the lower and upper portions of Pointe Coupee, St. Martin and Iberville Parishes, respectively. It is bordered on the North by Hwy 190, on the South by I-10, on the West by the Atchafalaya River, and on the East by the East Protection Guide Levee.

Sherburne was approximately 87% forested, 11% openings and 2% riparian habitat. Stands can be categorized into 4 primary types: cottonwood-sycamore, oak-gum-sugarberry-ash, willow-cypress-ash, and overcup oak-bitter pecan (LDWF 2002). Logging practices of previous landowners limited the location of hard mast producing species primarily to streamside management zones or sites where hydrology made logging practices difficult. Although logged extensively in the 1950's, some areas of Sherburne have remained virtually untouched since. Individual overstory species most commonly found were eastern cottonwood (*Populus deltoids*), American sycamore (*Platanus occidentalis*), willow oak (*Quercus phellos*), water oak (*Q. nigra*), overcup oak (*Q. lyrata*), delta post oak (*Q. stellata* var. mississippiensis), Nuttall oak (*Q. texana*), live oak (*Q. virginiana*), diamondleaf oak (*Q. laurifolia*), American elm (*Ulmus*

americana), winged elm (*U. alata*), sweetgum (*Liquidambar styraciflua*), sugarberry (*Celtis laevigata*), green ash (*Fraxinus pennsylvanicus*), black willow (*Salix nigra*), baldcypress (*Taxodium distichum*), bitter pecan (*Carya aquatica*), water tupelo (*Nyssa aquatica*), and honey locust (*Gleditsia triacanthos*). Midstory was composed primarily of boxelder (*Acer negundo*), Drummond red maple (*A. rubra* var. drummondii), black cherry (*Prunus serotina*), red mulberry (*Morus rubra*), tallowtree (*Triadica sebifera*), and rough-leaf dogwood (*Cornus drummondii*), with regeneration of the canopy species also present. Understory was relatively sparse because of shading and annual persistent flooding. Common understory species included rattan vine (*Berchemia scandens*), greenbrier (*Smilax spp*.), blackberry (*Rubus spp*.), bedstraw (*Gallium spp*.), horsetail (*Equisetum hyemale*), trumpet creeper (*Campsis radicans*), Virginia creeper (*Parthenocissus quinquefolia*), wild carrot (*Daucus carota*), stinging nettle (*Urtica chamaedryoides*), poison ivy (*Toxicodendron radicans*) and southern shield fern (*Thelypteris kunthii*). Wildlife food plots dominated forest openings and were comprised primarily of brown top millet (*Panicum ramosum*), wheat (*Triticum* spp.) or sunflowers (*Helianthus* spp.). The remaining openings consisted of right-of-ways, levees or natural regeneration from forest cuts. Dominant species in these openings were Johnson grass (*Sorghum halepense*), ragweed (*Ambrosia* spp.), black-eyed susan (*Rudbeckia* spp.), ryegrass (*Lolium multiflorum*), goldenrod (*Solidago* spp.), beefsteak (*Perilla frutescens*), teaweed (*Sida rhombifolia*), and blackberry.

Harvest prescriptions were applied in contiguous sections, referred to as compartments. Study sites included 14 stands, representing 3 compartments: 5 individual-selection harvest stands (compartment size: 38.429 ha), 4 stands in which individual and group-selection harvest strategies were combined (compartment size: 50.402 ha) and 5 uncut mature stands (compartment size: 72.409 ha). Individual-selection stands were harvested in 1986**,** in which individual trees of American sycamore, black willow, boxelder, Drummond red maple and poor

quality green ash were selected for pulp and sawtimber. A combination of individual and group-selection cuts occurred between November 2000 and April 2001. The prescription for the group-selection cuts prescribed gaps of 0.202 to 2.023 ha in size, but in actuality most gaps fell between 0.101 and 1.214 ha after the marking and sequential harvest was completed. Trees selected for harvest were not specified; the gaps were created in areas that had good stocking levels (300+ stems per 0.40 ha) of regeneration present under the overstory canopy. The regeneration size for that determination was seedling sapling size (>0.914 m height and < 5.08 cm diameter; (Kenny Ribbeck, LDWF, personal communication). Stand replicates in which the combination of individual and group-selection cuts occurred hereafter will be referred to as group-selection stands.

Ben's Creek Wildlife Management Area

Research was conducted at Ben's Creek WMA, a 5,607 ha tract of intensively managed loblolly pine (*Pinus taeda)* forest, owned by Weyerhaeuser Company and managed by LDWF. It is located west of Bogalusa, LA, in Washington Parish, and is accessible by LA Highway 10. The terrain was rolling hills managed primarily for pine timber. Loblolly pine was the dominant overstory species. Longleaf pine, red maple (*Acer rubrum*), black cherry, persimmon (*Diospyros virginiana*) and southern red oak (*Q. falcata*) also were found in the overstory, but to a much lesser extent. The vegetation succession and composition of the area were influenced by frequent timber management activities. Yaupon (*Ilex vomitoria*), broom sedge, French mulberry (*Callicarpa americana*), blackberry (*Rubus* spp.) and wax myrtle (*Myrica cerifera*) were found in the understory. There were several small creeks located on the area, which were characterized by several dominant overstory species, including blackgum (*Nyssa sylvatica*), yellow poplar (*Liriodendron tulipifera*), and sweetbay magnolia (*Magnolia virginiana*). Wax myrtle, black titi (*Cliftonia monophylla*), greenbriar (*Smilax* spp.), gallberry (*Ilex glabra*) and switchcane

(*Arundinaria gigantea*) were commonly found in the understory. An extensive system of wildlife food plots was established to benefit deer, turkey, quail, and rabbits, as well as other non-game species.

Commercial timber production was the primary management objective of this forest. Establishment of intermediate practices included clearcutting, chemical or prescribed-burn site preparation, chemical release, bedding, repeated fertilization, and thinning. These strategies created groups of stands that differed in age class. Four replicate stands in each of 4 forest age classes were selected for this study. Replicates included 4 stands harvested in 2002 (referred to as 1 year stands), 4 stands harvested in 1998-1999 (referred to as 4-5 year stands), 4 stands harvested in 1994 (2 stands) and 1996 (2 stands), referred to as 7-9 year stands, and 4 stands, 1 harvested in each of 1980 (thinned in 2000), 1981 (thinned in 2003), 1988 (thinned in 2002) and 1990 (thinned in 2003). Stand replicates are referred to as 13-23 year stands. Site references reflect age of stands at the time data collection began and hereafter will be used to differentiate stand age classes. Average size of selected stands was 6.4 ha (range 2.2 – 11.6 ha). Site preparation of selected stands included prescribed burning; regeneration of stands was carried out by machine planting of seedlings, with the exception of 3 stands, which were hand planted.

Sandy Hollow Wildlife Management Area

Research was conducted at Sandy Hollow WMA, a 1422.44 ha tract of longleaf pine (*P. palustris)* savannah, owned and managed by LDWF. Sandy Hollow is located in the northern portion of Tangipahoa Parish along LA Highway 10, near Arcola, LA. Most of the rolling-hill terrain of Sandy Hollow was characterized by savannahs, including grasses such as broom sedge (*Andropogon virginicus*), and longleaf pine, with a few scattered hardwoods, such as blackjack oak (*Quercus marilandica*), bluejack oak (*Q. incana*), and black cherry (*Prunus serotina*). Only a small portion of the area was comprised of mature trees. Sandy Hollow was managed for

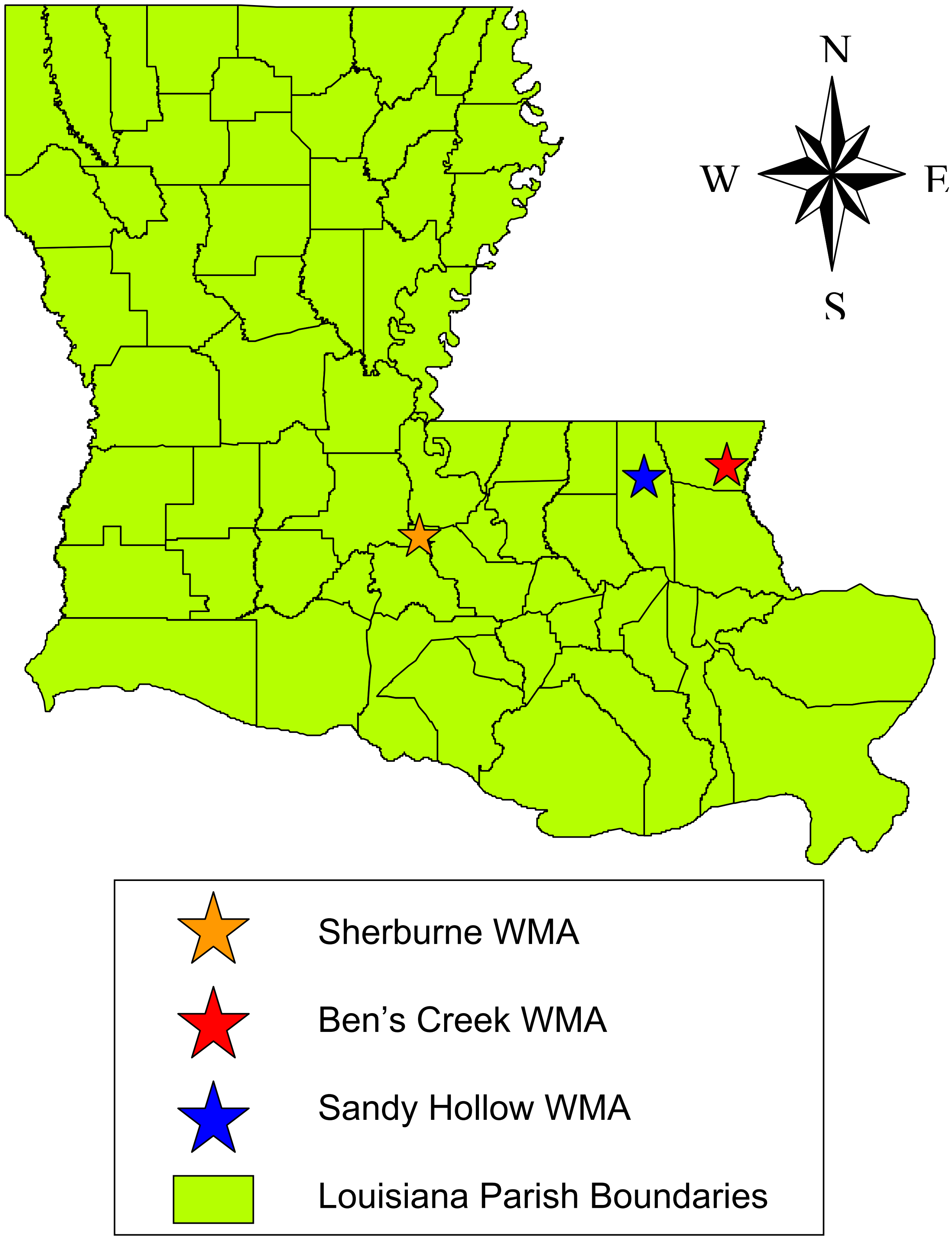


Figure 2.1. Location of Sherburne, Sandy Hollow, and Ben's Creek Wildlife Management Areas (WMA) in Louisiana.

upland game birds, including the northern bobwhite (*Colinus virginianus*), and is subjected to seasonal prescribed burns, use of herbicides, and an intensive food-plot program (LDWF 2002). Prescribed burning commenced in 1986 and was initially conducted annually in the winter season (December–March). Growing season burns (April–August) were increasingly implemented in 2002, primarily on an alternate year cycle. The primary fuel source was bluestem (*Andropogon* spp.); pine needles also contributed. Fire temperatures varied depending on fuel densities, humidity, wind, temperature, and the type of fire employed, including back, flank and head fires (Jimmy Stafford, LDWF, pers. comm.).

Twelve replicate stands were selected for this study. Each stand was burned at least once during the 2-year study period and possibly up to 3 times. Average size of selected stands was 3.5 ha (range 1.4 – 11.1 ha).

DATA COLLECTION

Drift Fence Arrays

We established drift fence arrays within each stand replicate representing management types selected on each study area. The exact number of drift fence arrays differed by area, depending primarily on the variety of management strategies present. Three drift fence arrays were located along established 500 m transects within each stand at 150, 300, and 450 m. Each array consisted of 3 pitfalls (PVC pipe, 0.305 m x 0.152 m), separated by 2, 4.572 m strips of aluminum flashing, arranged in a straight line. To reduce risk of mortality due to desiccation, pitfalls were at least partially covered or shaded by wooden covers. When not sampling, pitfalls remained either fully covered or filled with sticks to allow escape. We determined array orientation on a per array basis, but arrays generally altered between being positioned either perpendicular or parallel to the sampling transect. We positioned double-ended funnel traps (61 cm x 18 cm) on either side of each strip of aluminum flashing, totaling 4 funnel traps per drift

fence array (Enge 2001, Greenberg et al. 1994). These traps were anchored by substrate and a wooden stake, and shaded with wood covers. Drift fences were monitored for herpetofauna once monthly for 3 consecutive nights during April-June and September-November 2003 and 2004 on each study area.

Cover Boards

We distributed artificial cover objects (cover boards, 77.42 cm x 77.42 cm x 6.45 cm) made of green Eastern cottonwood throughout the selected sites at each WMA beginning during February 2003. Three linear arrays, each consisting of 10 cover boards, spaced 6 m apart, were established at 100, 250, and 400 m along each 500 m transect. Orientation of each cover board array generally was perpendicular to the primary transect. As cover boards are not suitable for permanently flooded areas or standing water, any point along the array where a board would end up in the water was skipped until all 10 boards were in place. Each cover board was uniquely labeled and placed directly on the ground surface after the surface was leveled out and cleared of leaf litter. Cover board sampling involved checking the underside surface for herpetofauna. We monitored cover board arrays once during the day in the months of January, April, July, and October. Cover board sampling began during April 2003 at all WMAs and ended in October 2004.

Visual Encounter Surveys

We used area constrained searches (visual encounter surveys) to estimate relative abundance of amphibians and reptiles that do not vocalize and are not typically captured with drift fences. Each 100 m x 2 m band transect bisecting avian point counts and used for avian vegetation sampling (described later) was used for searches. In addition to sightings on the ground, all ground cover objects located within the band transect were turned over to check for hidden individuals. We recorded the substrate on which each species was found. Because the

chance of recapturing individuals along a transect was fairly slim, and because capturing and marking of animals will reduce the time available for additional survey work, mark-recapture methods were not used in time constrained searches (Crump and Scott 1994). We surveyed each transect once during the day during April-June 2003 at all WMAs and during June 2004 at Sherburne. After conducting these surveys during 2003, the method was deemed inappropriate for Sandy Hollow (few observations, perhaps due to few cover objects) and Ben's Creek WMAs (very few observations, likely due to low visibility). Thus in 2004, surveys were conducted at Sherburne only. All surveys were conducted within 72 hours of a rain event.

Anuran Calling Surveys

Anuran calling surveys consisted of permanent listening stations stratified by habitat type and management scenarios specific to each study area. We established 5 listening stations 100 m apart along each primary sampling transect within each stand replicate, beginning at either 0 m (Sherburne and Sandy Hollow) or 100 m (Ben's Creek) and ending at either 400 m (Sherburne and Sandy Hollow) or 500 m (Ben's Creek). We performed calling surveys during nocturnal periods, beginning 30 minutes after sunset and continuing for several hours, ending no later than 12 am (Zimmerman 1994). Humidity, temperature, wind speed (Beaufort scale), sky condition, and moon phase were recorded for each site (USGS 2002). We conducted surveys at each listening station for 5 minutes. We recorded anuran species detected based on a calling index similar to that used by the Louisiana Amphibian Monitoring Program (LAMP 1996) and other monitoring programs currently underway (USGS 2002). Specifically, the index included the following values: 0 (no individuals calling), 1 (individuals can be counted, there is space between calls), 2 (calls of individuals can be distinguished, but there is overlapping of calls), 3 (full chorus, calls are constant, continuous and overlapping). An absolute count of species detected was attempted but was not analyzed due to both insufficient data and the difficulty in

obtaining a reliable absolute count of calling anurans. We performed surveys once during winter (December-January), spring (March-April), and summer (May-June) seasons between December 2002 and June 2004. Winter 2002 and spring 2003 calling surveys were not conducted at Ben's Creek WMA due to delayed approval to conduct research at this site.

Avian Point Counts

During the breeding seasons of 2003 and 2004, we used a 50 m circular fixed-radius point count method to survey avifauna at permanently established point count stations along WMA roads and within stands. Road point count stations were established at 0.4 (643.72 m) mile intervals adjacent to selected sites in each study area (Ralph 1993). These point count locations were determined prior to initial sampling based on odometer readings and were relocated during subsequent sampling periods using either flagging or the odometer. We also established similar point count stations along survey transects within each stand replicate at each study area. Transects generally began at the edge of a selected site, with some exceptions, due either to limited accessibility or shape or size of the site. In general, we established 6 point count stations, 150 m apart, and at least 100 m from the habitat edge (Hamel et al. 1996), within each stand replicate.

We conducted point counts for 10 minutes, and recorded all individual birds detected either by sight or sound at each point count station (Ralph 1993, Hamel et al. 1996). We recorded detections on bull's eye data sheets and then transcribed data onto bird count data forms similar to those suggested by Hamel et al. (1996). Bird distance from the observer was listed in 4 categories, as indicated on the bull's eye data sheet: 0-25 m, 25-50 m, beyond 50 m, or as flyovers. The time of first detection was listed in 3 categories: within the first 3 minutes, during the next 2 minutes, or during the last 5 minutes (Hamel et al. 1996). Point counts were conducted from April-June, from 15 minutes before sunrise to 3.25 hours following sunrise

(MacFaden and Capen 2002). We conducted point counts at each location a minimum of 3 times (approximately monthly) each year, with the exception of 2 stands at Sherburne during April of 2004, due to inaccessibility as a result of flooding. Surveys were not conducted during very windy or rainy weather, as this would have impaired detection (Hamel et al. 1996). Total number of road point counts (Sherburne: 30; Ben's Creek: 31; Sandy Hollow: 20) and stand point counts (Sherburne: 84; Ben's Creek: 96; Sandy Hollow: 72) differed by study area, depending on area of available habitat types, overall number of habitat types, and forest management objectives unique to each area. One observer conducted point counts in 2003 and two observers conducted point counts in 2004.

Microhabitat

We examined relationships between microhabitat and herpetofauna captured at drift fence arrays based on select habitat variables measured at each array. Specifically, we quantified trees, shrubs, ground cover, mid- and upper story characteristics within a 10 m circular plot centered on each drift fence array. We identified, measured and tallied trees ($\geq$ 8 cm in diameter), and quantified other woody shrubs at least 1 m in height (<8 cm in diameter). We took an absolute count of fine woody debris (FWD, 3-10 cm x 30 cm), coarse woody debris (CWD, 10+ cm x 30 cm), and number of stumps within the 10 m circular plot. We measured ground cover classes at 4 equidistant points along the perimeter of the 10 m circle using a 1 m^2 Daubenmire frame (Daubenmire 1959) and included percentage leaf litter, grass, forb cover, woody cover, fern cover, bare ground, rock, water, moss, CWD, and palmetto (Sherburne only). We measured litter depth and herbaceous vegetation height (cm) at 16 points spaced 1 m apart, along 2 10 m line transects that intersected and were centered within the 10 m circular plot. Measurements along the line began 2 m from the point of intersection and continued outward towards the perimeter of the circle. We estimated canopy closure using a forest densiometer (Lemmon 1956)

at the center of each 10 m circular plot. We surveyed trees, shrubs, CWD, FWD and stumps once each year at drift fence arrays in May or June. We measured canopy closure, litter height, litter depth and ground cover class at each drift fence array in May or June and October of both years.

We examined potential relationships between microhabitat and herpetofauna detected during visual encounter surveys using selected habitat data sampled along 100 m x 4 m band transects (discussed below) bisecting avian point count stations. Since visual encounter surveys were conducted on 3 of the 6 band transects in each stand replicate, only habitat data associated with these 3 transects were included in analyses. Due to low numbers of detections during visual encounter surveys at Sandy Hollow and Ben's Creek WMAs, these analyses were conducted with data collected at Sherburne only.

Although initially proposed and begun, vegetation sampling associated with anuran calling surveys was eventually deemed inappropriate. Location of calling anurans in relation to listening stations was often distant and difficult to judge. Thus microhabitat sampling associated with listening stations may not have reflected a true relationship between herpetofauna and microhabitat.

We surveyed habitat characteristics associated with each stand-level avian point count station during the summer field season of each year. A 100 m band transect, measuring 4 m across, was established to completely bisect each point count station. We identified, measured and enumerated all trees (> 8 cm in diameter [breast height]), and tallied all snags located within the band transect. Additionally, woody species (<8 cm in diameter) located within the inner 2 m of the transect were counted and identified. We measured canopy closure every 10 m along the transect using a forest densiometer, and ground cover classes as described previously were estimated at the same point using a 1 m^2 Daubenmire frame.

Landscape Characteristics

We derived landscape composition and configuration variables for each drift fence array, anuran listening station, and avian point count station at Ben's Creek and Sandy Hollow WMAs by using 1998 Digital Orthophoto Quarter Quadrangle (DOQQ) images archived by the Louisiana State University CADGIS Research Laboratory (http://atlas.lsu.edu), ArcView GIS 3.3 (Environmental Systems Research Institute, Inc. 1992-2002), and a geographic information system (GIS) representative of each WMA. A GIS based on forest inventory stand data for Ben's Creek was provided by Weyerhaeuser Company, and included year of regeneration and tree type (pine or hardwood) for each stand (polygon). From this information, we generated 6 cover classes, including 5 pine cover classes based on period of regeneration and 1 streamside management zone (SMZ) cover class, which were used to develop landscape variables for Ben's Creek. We developed a spatial layer based on 5 cover types for Sandy Hollow. Polygons were identified as longleaf pine, longleaf savannah, mixed pine-hardwood forest, pine forest, or openings (agriculture, food plots, residential areas, and ponds). Although a polygon depicted as one cover type may have been heterogeneous at a finer resolution, the selected cover type represented the dominant cover type. The 1998 DOQQs predate group-selection cuts at Sherburne, and time did not permit using 2004 DOQQs. Thus landscape attributes for Sherburne WMA are not reported.

We calculated class-specific composition and configuration metrics within a 500 m radius circular buffer zone (7.26 ha) around each drift fence array, avian point count station, and central anuran listening station within each stand replicate. The buffer radius was selected based on an average home range size (i.e. dispersal from breeding pond of amphibians for terrestrial foraging) of herpetofauna (deMaynadier and Hunter 1995), and previous research of landscape attributes associated with both herpetofauna (Nuzzo and Mierzwa 2000; Guerry and Hunter

2002; Knutson et al. 2004) and avifauna (Berry and Bock 1998; Drapeau et al. 2000; Hennings and Edge 2003). We created buffers and then intersected buffers with landcover. Class-specific metrics were generated with the Patch Analyst (Elkie et al. 1999) extension in ArcView. We calculated class-specific configuration metrics for the 6 cover classes at Ben's Creek (0-2-, 4-6-, 7-9-, 11-23-, and 24-63-year old loblolly pine stands, and SMZs) and 5 cover classes at Sandy Hollow (longleaf pine, longleaf savannah, mixed pine-hardwood forest, pine forest, openings) and included median patch size (MEDPS), edge density (ED), and area-weighted mean shape index (AWMSI). We calculated percentage composition of each class within each buffer by dividing class area (ha) by total landscape area (ha) within each buffer. We calculated mean values and standard errors for all metrics for all sampling points within each drift fence array, anuran listening station, and avian point count station data set.

We calculated 2 additional landscape variables, distance of each drift fence and survey station to nearest body of water (stream or pond) and nearest road. These were included because many amphibians require both aquatic and terrestrial habitat during their life cycle, and roads may influence dispersal and habitat use of birds (Hennings and Edge 2003) and amphibians and reptiles (Petranka et al. 2003; Stevens et al. 2002).

STATISTICAL ANALYSIS

We subjected 4 types of herpetofauna data sets to statistical analyses. Anuran calling survey data and cover board data comprised 2 of the data sets. Anuran calling survey data was subjected to statistical analysis of effects of selection cutting and stand age, and effects of landscape characteristics. Cover board data was tested for effects of selection cutting and stand age. We combined visual encounter survey data and vegetation data associated with these surveys, which formed another data set. Drift fence captures and the vegetation sampled within the 10 m circular plots at each drift fence array were combined into one data set. Avian point

count surveys and the vegetation data associated with these surveys were combined into one data set. We performed all analyses using the Statistical Analysis System (SAS Institute 1999-2001).

Associations of Herpetofauna with Selection Cutting and Stand Age

We used a nested analysis of variance (ANOVA), with month and season nested in year (where appropriate), stands nested within treatments and arrays nested within stands, to compare mean species richness and relative abundance of herpetofauna collected per array and sampling period. Treatment (n=3 for Sherburne data; n=4 for Ben's Creek data) was the explanatory variable; dependent variables included herpetofauna groups and species with sufficient captures or encounters for analysis.

We used the GLIMMIX procedure, a type of generalized linear mixed models (Blouin and Saxton 1990) procedure, to perform the ANOVA tests. GLIMMIX fits statistical models to data with correlations of nonconstant variability and where the response is not necessarily normally distributed. Since count data is presumed to follow a Poisson distribution (Littell et al. 2002), we specified a Poisson distribution with a log link to examine species richness and relative abundance. Therefore, herpetofauna data were log transformed prior to tests. Recent research has found that count data also are typically overdispersed; the GLIMMIX procedure automatically corrects for over- or underdispersion in the F statistics. The Kenward-Rogers adjustment for degrees of freedom also was included to reduce the potential for Type 1 error, which is possible with nested and unbalanced data. This sometimes, but not unexpectedly, resulted in some noninteger estimated degrees of freedom. When ANOVA indicated a difference among treatments, we used the Tukey-Kramer adjustment for pair wise comparison procedures to find which treatments were different. Year, season, and month (drift fence data only) were included in the random statement as sources of variation.

Species richness was defined as the number of species observed per array (or listening station) and survey period. Definition of relative abundance varied by survey type. Estimates of relative abundance at drift fences and cover boards equaled the total number of captures per array and sampling period, divided by sampling effort (discussed below). Values are reported per 20 trap days (multiplied by 20) as an index of herpetofauna relative abundance (Grialou et al. 2000). Estimates of relative abundance using visual encounter surveys equaled total number of encounters per array and survey period. Relative abundance estimates based on anuran calling surveys equaled the average calling index level per stand and survey period. Prior to analysis, winter surveys conducted during 2003 at Ben's Creek were removed from the data set due to 0 detections. We calculated herpetofauna sampling effort for each array and sampling period for drift fence and cover board data. One pitfall, funnel trap, or cover board active for one day was equivalent to one trap day. Effort for drift fence data was calculated by multiplying the total number of traps per array (3 pitfalls + 4 funnel traps) times the total number of trap days and then subtracting from this the number of "bad" traps times the total number of trap days (i.e. [7 traps x 3 trap days]-[1 bad trap x 3 trap days] = 18) which equals the sampling effort for one array for one period. We designated traps as bad during a particular sampling period if they were flooded, inhabited by paper wasps (*Polistes* spp.) or covered by imported red fire ant (RIFA, *Solenopsis invicta* Buren) mounds, ripped apart or burned beyond use (funnel traps), or otherwise unfit for effectively sampling herpetofauna.

Effort was similarly calculated for cover board data, except that there was only 1 trap day per sampling period, and thus effort equaled the total number of boards per array (10) minus the number of bad boards. We designated cover boards as "bad" during a sampling period if they were under water, completely consumed by termites or RIFA, destroyed or missing. Effort was not calculated for visual encounter surveys or anuran calling surveys because effort was assumed

to be 100% during each sampling period for these survey types. Seasonal effects were not tested within the anuran calling survey data set, as frogs have a strong seasonality to their breeding, and hence, detection.

Associations of Avifauna with Selection Cutting and Stand Age

Prior to analysis, detections of birds > 50 m from the center of the point count, or recorded as flyovers, were removed from the data set given that estimates of abundance or density depend on having a measure of the sample area (Farnsworth et al 2002). We also removed wading birds, nocturnal birds, and birds with < 3 total detections from the data set. Wading birds were not the focus of the project, diurnal surveys did not appropriately represent habitat use of nocturnal species, and species with < 3 detections may have been incidental. We classified bird species into 17 guilds (2 migratory, 4 habitat, 5 nesting, and 6 foraging). Species with ≥ 10% frequency of occurrence across all surveys were included in individual analyses.

As with herpetofauna data sets, we used GLIMMIX to perform nested ANOVA tests on effects of treatment on species richness, relative abundance of each guild, and occurrence of each individual with ≥ 10% frequency of occurrence. Species richness was defined as the number of species observed per point count and survey period. We defined relative abundance of guilds as number of individuals detected per point count and survey.

Associations of Microhabitat with Selection Cutting and Stand Age

With the exception of moss along visual encounter survey routes at Sherburne, palmetto, rock, moss, and water were detected in insufficient quantities for analysis at all study areas. In Sherburne data sets, percentage leaf litter and other litter were combined to form one variable, percentage litter cover. In Sherburne and Sandy Hollow data sets, percentage grass and forb cover also were combined to form one variable, percentage herbaceous cover. Two composite variables, total number of non-pine saplings and total number of non-pine trees, were included in

microhabitat analyses of Ben's Creek avian point count data. Three composite variables, percentage hardwood, and longleaf and loblolly pine trees, were included in microhabitat analyses of Sandy Hollow avian point count data.

Microhabitat variables were tested for normality (Shapiro-Wilk's *W*; SAS 1999-2001) and, where appropriate, log, arcsine square root, or rank transformed to help normalize distributions when required (Ott and Longnecker 2002; Conroy 1999). We used a nested ANOVA to test for differences among treatments for all habitat variables measured at drift fences and along band transects associated with avian point count stations. We tested variables measured during fall and spring with season nested within years, stands nested within treatment, season and year, and arrays nested within stand, treatment, and year. We tested variables measured only in spring with stand nested within treatment and year, and arrays nested within treatment and stand. Treatment was the explanatory variable; dependent variables included all habitat variables specified for each WMA. Only means and standard errors of the untransformed microhabitat variables are reported.

We used the mixed model procedure (PROD MIXED) in SAS to perform the ANOVA tests on each microhabitat variable. Selection of microhabitat variables for analysis varied slightly among the WMAs, due to absence of certain variables at each site (i.e., 0% water at Ben's Creek WMA). Absolute count of CWD, rather than the percentage CWD, was used in analyses of drift fence data. We surveyed vine and woody cover as one variable in 2003, and this was later deemed inappropriate. Thus, values for woody and vine cover are based on 2004 data only.

Associations of Herpetofauna and Avifauna with Microhabitat

We examined occurrence and relative abundance of select herpetofauna and avifauna related to microhabitat conditions using a nested ANOVA and principal components,

representative of microhabitat variables, as the independent variables (Stevens et al. 2002). These analyses were performed using drift fence and visual encounter survey data sets. Microhabitat variables were first tested for multicollinearity using Pearson's correlation coefficients between all variables. Correlations of ≥ 0.8 resulted in one variable from a pair being dropped. When possible, the harder variable to interpret biologically was dropped.

We performed a centered and standardized principal components analysis (PCA; Proc Factor; SAS 1999-2001; Cody and Smith 1997; Conroy 1999) using a correlation matrix (Jongman et al 1995) to summarize vegetation patterns based on the microhabitat variables examined. This procedure standardized data collected on different scales. An orthogonal rotation (Varimax) was used to maximize dispersion of the ordination to facilitate interpretation by minimizing the number of variables with high loadings on one component, causing the loadings of each variable to be more clearly differentiated. We rotated the axes to reduce the number of variables that were highly loaded on (i.e., correlated with) more than one component. Principal components with eigenvalues ≥ 1.0 and that accounted for $\geq 5\%$ of the variance were retained for further analyses because these component variables accounted for more variation in the data set than one variable potentially could by itself (Jackson 1993). Correlations between variables and significant principal components were interpreted as strong if the value was > 0.40. If a variable appeared on more than one component, the component with which the variable had the strongest correlation was interpreted to be a better representation of the variance explained by that variable.

We performed ANOVA using GLIMMIX, with the retained principal components as explanatory variables, and the occurrence of select herpetofauna species or groups with ≥ 10 captures or detections, and select avifauna species or guilds with a minimum frequency of occurrence of 0.10 as dependent variables. We specified a binomial distribution and a logistic

regression in the procedure to examine species occurrence related to principal components. All possible models among the retained principal components were examined for associations with occurrence of herpetofauna and avifauna. Species occurrence within the visual encounter survey data set was not examined due to models with poor goodness of fit (Hosmer and Lemeshow 2000).

Associations between species richness and relative abundance of herpetofauna and microhabitat were similarly examined for drift fence data (species richness only) and visual encounter survey data using a Poisson error term, also using retained principal components. Dependent variables included species or groups for which data was sufficient. Likewise, associations of relative abundance of bird guilds, and select bird species, with microhabitat were examined.

A subset of models was selected using Akaike Information Criterion (AIC), which is an estimate of the expected, relative distance between fitted models and the unknown true mechanism that actually generated the observed data. AIC adjusts for the number of variables in each model, and thus selects the most parsimonious model among all possible model combinations. Models with a Δ AIC score between 0-2 were retained in the subset of models. This was based on the idea that models with $\Delta_i < 2$ have a substantial level of empirical support in explaining variation in the data (Burnham and Anderson 2002). We calculated model averaged parameter estimates, unconditional standard errors, and AIC weights, which represented the relative importance of each component among a set of models.

Associations of Herpetofauna and Avifauna with Landscape Characteristics

We tested landscape variables for multicollinearity using Pearson's correlation coefficients between all variables. However, strong correlations between landscape variables, especially configuration variables, are to be expected because all spatial metrics are based on a

limited number of basic parameters, primarily patch area, edge length, and inter-patch distance, and thus are interrelated (Hargis et al. 1998). Therefore the multicollinearity tests were conducted for informational purposes only, and did not result in removal of any of the landscape variables.

As with microhabitat data, we performed a centered and standardized principal components analysis using a correlation matrix, and specified an orthogonal rotation in the output. This was done to reduce the number of variables and to summarize landscape attribute patterns based on the landscape variables examined (see Avian and Herpetofauna Associations with Microhabitat). Principal components with eigenvalues $\geq$ 1.0 and that accounted for > 5% of the variance were retained for further analyses. Correlations between variables and significant principal components were interpreted as strong if the value was > 0.40. If a variable appeared on more than one component, the component with which the variable had the strongest correlation was interpreted to be a better representation of the variance explained by that variable.

We performed ANOVA using GLIMMIX, with the retained principal components as explanatory variables, and the occurrence of select herpetofauna and avifauna as dependent variables. Selection of herpetofauna and avifauna species, groups, and guilds corresponded with those selected for analysis of associations with microhabitat. Selection of models and calculation of parameter estimates, standard errors, and AIC weights were similar to above.

CHAPTER 3. ASSOCIATIONS OF HERPETOFAUNA WITH FOREST MANAGEMENT, MICROHABITAT, AND LANDSCAPE CHARACTERISTICS: RESULTS AND DISCUSSION

RESULTS: SHERBURNE WMA

Associations of Herpetofauna with Selection Cutting

Between December 2002 and November 2004, we detected 32 species of herpetofauna, including 15 amphibians and 17 reptiles, based on the combination of all standardized survey methods and chance encounters. Mean species richness in group-selection (18.5 $\pm$ 1.04), individual-selection(18.4 $\pm$ 0.81), and uncut sites (18.8 $\pm$ 1.66) was similar ($F_{2,11}$=0.04, P=0.964). We recorded 1,165 captures from drift fences, cover boards and visual encounter surveys combined (detections during anuran calling surveys were indexed and thus not included in this number). The gulf coast toad (*Bufo valiceps*), eastern narrowmouth toad (*Gastrophryne carolinensis*), spring peeper (*Pseudacris crepitans*), green treefrog (*Hyla cinera*), bronze frog (*Rana clamatans*), and southern leopard frog (*R. utricularia*) were detected by all survey methods. The eastern newt (*Notophthalmus viridescens*), green anole (*Anolis carolinensis*), five-lined skink (*Eumeces fasciatus*), ground skink (*Scinella lateralis*) and southern ringneck snake (*Diadophus punctatus*) were detected by 3 surveys methods, excluding anuran calling surveys (Table 3.1).

Drift-fence Arrays

We captured 591 individuals (11 amphibian and 9 reptile species) between April 2003 and November 2004 (9,978 trap days, Table 3.1). Eastern narrowmouth toad, bronze frog and southern leopard frog were the most frequently captured amphibians, constituting 80% of amphibian captures and 61% of total captures. Ground skink was the most frequently captured reptile, accounting for 61% of reptile captures and 11% of total captures. Sixteen of 20 species (80%) were captured at least once in all 3 treatment types. Four species were unique to one

treatment type within drift fence surveys. Two species, the rough earth snake (*Virginia striatula*) and eastern mud turtle (*Kinosternon subrubrum*), were unique to drift fence surveys.

Species richness and relative abundance of amphibians did not differ by year ($F_{1,13.6}$= 0.46, P=0.511 and $F_{1,15.3}$=0.87, P=0.365, respectively) or season ($F_{1,13.5}$=0.07, P=0.789 and $F_{1,14.9}$=0.05, P=0.835, respectively). Species richness and relative abundance of reptiles did not differ by year ($F_{1,20}$=1.89, P=0.184 and $F_{1,19.1}$=1.56, P=0.226, respectively) or season ($F_{1,19.3}$= 0.99, P=0.331 and $F_{1,18.6}$=0.71, P=0.409, respectively). Species richness varied across stand type for amphibians ($F_{2,135}$=16.72, P<0.001) but not for reptiles ($F_{2,132}$=0.25, P=0.781). Richness of amphibians was greater in individual-selection and uncut sites than group-selection sites. Relative abundance of amphibians ($F_{2,200}$=11.30, P<0.001) and anurans ($F_{2,201}$= 11.01, P<0.001) also was greater in individual-selection and uncut sites compared to group-selection sites. Relative abundance of reptiles ($F_{2,141}$=0.45, P=0.640) and lizards ($F_{2,210}$=0.35, P=0.707) did not vary across stand type.

We captured 3 species of amphibian, 1 species of reptile, and 1 genus of snake (*Agkistrodon* spp.) in sufficient numbers for individual analysis. Relative abundance of eastern narrowmouth toad ($F_{2,223}$=6.31, P=0.002) and bronze frog ($F_{2,256}$=6.75, P=0.001) was greater in both uncut and individual-selection than group-selection sites, whereas relative abundance of southern leopard frog ($F_{2,250}$=1.52, P=0.221), ground skink ($F_{2,267}$=0.17, P=0.842), and *Agkistrodon* spp. ($F_{2,61.3}$=0.49, P=0.616) did not vary across stand type.

Cover Boards

We observed 67 individuals (8 amphibian and 5 reptile species) either on or under cover boards between April 2003 and October 2004 (2,584 trap days, Table 3.1). Gulf coast toad, eastern narrowmouth toad and bronze frog were the most frequently captured amphibians, constituting 90% of amphibian captures and 64% of total captures. Ground skink was the most

frequently captured reptile, accounting for 38% of reptile captures and 7% of total captures. Gulf coast and eastern narrowmouth toads were captured in all treatment types. Green anole and skinks (*Eumeces* spp.) were captured in group-selection and individual-selection stands only. Ranids (*Rana* spp.), ground skink, and southern ringneck snake (*Diadophis puntatus*) were captured in individual-selection and uncut stands only. A single mole salamander (*Ambystoma talpoideum),* and eastern newt (*Notopthalmus viridescens)* were unique to individual-selection stands. The mole salamander was unique to cover board surveys.

Species richness and relative abundance of herpetofauna did not differ by year ($F_{1,9.12}$=0.04, P=0.844 and $F_{1,9.41}$=0.04, P=0.839, respectively). However, species richness and relative abundance were greater in fall, winter, and spring than summer ($F_{3,62.3}$=5.74, P=0.002 and $F_{3,65.1}$=5.71, P=0.002, respectively). Species richness ($F_{2,91.5}$=2.91, P=0.059) and relative abundance ($F_{2,99.1}$=2.77, P=0.068) of herpetofauna also varied across stand type. Both species richness and relative abundance were greater in individual-selection than group-selection sites, and similar in uncut sites. Species richness ($F_{2,114}$=2.19, P=0.116) and relative abundance ($F_{2,114}$=2.18, P=0.117) of amphibians, and relative abundance of anurans ($F_{2,120}$=0.94, P=0.394) were similar across stand type. There were insufficient captures of reptiles for analysis. We captured the gulf coast toad in sufficient numbers for individual analysis. Relative abundance of gulf coast toad did not vary across stand type ($F_{2,124}$=2.10, P=0.127).

Visual Encounter Surveys

We encountered 507 individuals (9 amphibian and 10 reptile species) during spring visual encounter surveys in 2003 and 2004 (84 surveys, Table 3.1). Spring peeper was the most frequently encountered amphibian, constituting 53% of amphibian captures and 48% of total captures. Ground skink was the most frequently encountered reptile, constituting 49% of reptile captures and 5% of total captures.

Species richness and abundance varied across years for reptiles ($F_{1,19.9}$=5.41, P=0.031 and $F_{1,19.4}$=5.39, P=0.031, respectively) but not amphibians ($F_{1,17.9}$=2.13, P=0.162 and $F_{1,23.5}$=2.24, P=0.148, respectively). Richness and abundance of reptiles were greater in 2003 than 2004. Species richness varied across stand type for herpetofauna ($F_{2,26}$=2.61, P=0.093) and amphibians ($F_{2,24.2}$=2.68, P=0.092). Species richness for both groups was greater in individual than group-selection sites and similar in uncut sites. Relative abundance of herpetofauna ($F_{2,29.1}$=2.54, P=0.096), amphibians ($F_{2,27.4}$=3.02, P=0.065) and anurans ($F_{2,27.5}$=3.13, P=0.059) was greater in uncut than group-selection sites, and similar in individual-selection sites to the other 2 stand types. Richness and relative abundance of reptiles ($F_{2,27.1}$=0.89, P=0.423 and $F_{2,27.6}$=0.61, P=0.548, respectively) and relative abundance of lizards ($F_{2,28.4}$=0.37, P=0.696) did not vary across stand type.

We encountered 4 amphibian and 1 reptile species during visual encounter surveys in sufficient numbers for individual analysis. Relative abundance of spring peeper was greater in individual-selection and uncut ($F_{2,31.1}$=5.27, P=0.017) than group-selection sites. Relative abundance of gulf coast toad ($F_{2,51.6}$=1.02, P=0.368), bronze frog ($F_{2,48.4}$=0.56, P=0.572), southern leopard frog ($F_{2,28}$=0.46, P=0.637), and ground skink ($F_{2,37}$=1.13, P=0.335) did not vary across stand type.

Anuran Calling Surveys

We detected 13 species during anuran calling surveys between December 2002 and June 2004 (420 listening station surveys, Table 3.1). Green treefrog and spring peeper were the most frequently detected anurans. Prior to analysis of effects of stand type on anuran species within the anuran calling survey data set, December 2003 (winter) surveys were removed due to 0 calling anurans detected. Neither species richness nor relative abundance of anurans differed across stand type ($F_{2,24.9}$=1.10, P=0.345 and $F_{2,21.6}$=0.31, P=0.739, respectively).

We detected 7 anuran species and 1 genus of anurans (*Bufo* spp.) at a frequency sufficient to conduct individual analyses. Relative abundance of squirrel treefrog (*Hyla squirella*) was greater in group-selection ($F_{2,12.4}$=4.37, P=0.037) than individual-selection sites, and similar in uncut sites to both. Relative abundance of gray treefrog (*H. versicolor*) was greater in both individual-selection and uncut sites ($F_{2,19.4}$=2.62, P=0.099) compared to group-selection sites. Relative abundance of toads (*Bufo* spp., $F_{2,21.8}$=2.29, P=0.125), northern cricket frog (*Acris creptians,* $F_{2,20.1}$=1.45, P=0.259), spring peeper ($F_{2,57.9}$=0.91, P=0.408), Cope's gray treefrog (*Hyla chrysoscelis,* $F_{2,18.3}$=2.57, P=0.104), green treefrog ($F_{2,55.9}$=2.14, P=0.923) and bronze frog ($F_{2,26.6}$=2.04, P=0.150) did not differ across stand type.

Associations of Microhabitat with Selection Cutting

Drift Fence Arrays

Eight variables differed across stand type (Table 3.2). Canopy closure ($F_{2,50}$=24.78, P<0.001) and percentage bare ground ($F_{2,50}$=9.86, P=0.002) were greater in individual-selection and uncut than group-selection sites. Percentage fern cover was greater in uncut ($F_{2,50}$=7.59, P=0.001) than individual and group-selection sites, which were similar. Percentage litter was greater in individual ($F_{2,50}$=6.97, P=0.002) than group-selection sites, and similar in uncut sites. Group-selection sites had greater percentage vine cover ($F_{2,50}$=4.36, P=0.018) compared to uncut sites, with values in individual-selection sites similar to both. Percentage woody cover ($F_{2,50}$= 10.35, P=0.002), vegetation height ($F_{2,50}$=9.92, P=0.002) and shrub richness ($F_{2,24}$=4.96, P=0.016) were greater in group than individual-selection and uncut sites.

Visual Encounter Surveys

Nine variables varied across stand type (Table 3.3). Percentage fern cover was greater in uncut ($F_{2,24}$=4.48, P=0.022) than group-selection sites, and similar in individual-selection sites to both. Canopy closure was greater in uncut and group-selection sites ($F_{2,24}$=5.81, P=0.009) than

Table 3.1. Herpetofauna captures/detections using drift fence arrays, cover boards, visual encounters, anuran calling surveys and chance encounters (denoted by X) between December 2002 – November 2004 in group-selection (GS), individual-selection (IS) and uncut (US) stands at Sherburne WMA, LA.

	Herpetofauna Survey Method											
	Drift-fence arrays[a]			Cover boards[a]			Visual encounters[b]			Anuran calling surveys[bc]		
Stand Type	GS	IS	US	GS	IS	US	GS	IS	US	GS	IS	US
Effort (# trap days)	2906	3324	3748	802	760	1022	24	30	30	130	105	150
Amphibians												
Bufo valiceps	9	17	7	7	6	1	4	7	1	1	3	2
B. woodhousii	4	7	5	0	0	0	0	0	0	0	1	0
Unknown *Bufo* spp.	1	0	2	0	0	0	0	0	0	0	0	0
Gastrophryne carolinensis	19	61	35	7	7	2	2	0	2	1	2	1
Acris creptians	1	9	7	0	0	0	3	5	3	3	3	3
Hyla chrysoscelis	0	0	0	0	0	0	0	0	0	2	3	3
H. cinera	0	0	2	0	0	1	1	3	0	3	3	3
H. squirella	0	0	0	0	0	0	0	0	0	2	1	2
H. versicolor	0	0	0	0	0	0	0	0	0	1	3	3
Pseudacris creptians	2	0	4	0	0	1	1	60	180	3	3	3
P. triseriata	1	3	1	0	0	0	0	5	1	2	3	2
Unknown *Pseudacris* spp.	0	0	0	0	0	0	1	0	2	0	0	0
Rana catesbeiana	0	1	0	0	0	0	0	X	0	1	1	2
R. clamatans	5	76	76	0	5	8	3	20	20	3	2	2
R. utricularia	23	28	39	0	2	1	30	22	36	2	2	2
Unknown Rana spp.	1	0	0	0	0	0	2	18	24	0	0	0
Total anurans	66	202	178	14	20	14	47	140	269	3	3	3
Ambystoma talpoideum	0	0	0	0	1	0	0	0	0	NA	NA	NA
Notopthalmus viridescens	1	2	6	0	5	0	1	1	2	NA	NA	NA
Total salamanders	1	2	6	0	6	0	1	1	2	NA	NA	NA
Total amphibians	67	204	184	14	26	14	48	141	271	NA	NA	NA

[a] Effort = [# captures/# functional traps] x 20 per array per survey period per year; [b] Effort = total # of surveys during study period.
[c] Highest index value (0-3) assigned during study period: 1= individuals can be counted, 2= individuals distinguished but calls overlapping, 3= full chorus.
[d] Does not include chance encounters.

Table 3.1 continued.

	Herpetofauna Survey Method											
	Drift fence arrays			Cover boards			Visual encounters			Anuran calling surveys		
Stand Type	GS	IS	US	GS	IS	US	GS	IS	US	GS	IS	US
<u>Reptiles</u>												
Anolis carolinensis	5	15	7	1	1	0	3	1	3	NA	NA	NA
Eumeces fasciatus	6	3	0	1	1	0	1	0	1	NA	NA	NA
E. laticeps	1	2	1	1	1	0	1	X	1	NA	NA	NA
Unknown *Eumeces* spp.	0	0	2	0	0	0	0	1	0	NA	NA	NA
Scinella lateralis	17	27	23	0	1	4	2	8	13	NA	NA	NA
Lizards	29	47	33	3	4	4	7	10	18	NA	NA	NA
Kinosternon subrubrum s.	0	0	1	0	0	0	0	0	X	NA	NA	NA
Macroclemys temmincki	0	0	0	0	0	0	X	0	X	NA	NA	NA
Terrapene Carolina t.	0	0	0	0	0	0	0	2	X	NA	NA	NA
Agkistrodon contortrix c.	3	4	5	0	0	0	1	1	X	NA	NA	NA
Agkistrodon piscivorus l.	0	3	4	0	0	0	0	3	1	NA	NA	NA
Coluber constrictor	0	0	0	0	0	0	X	X	X	NA	NA	NA
Diadophis puntatus p.	3	2	1	0	1	1	X	1	1	NA	NA	NA
Elaphe obsolete l.	0	0	0	0	0	0	X	1	0	NA	NA	NA
Farancia abacura	0	0	0	0	0	0	X	0	0	NA	NA	NA
Opheodrys aestivus	0	0	0	0	0	0	0	0	X	NA	NA	NA
Thamnophis proximus	0	0	0	0	0	0	0	X	0	NA	NA	NA
T. sirtalis	0	0	0	0	0	0	0	0	X	NA	NA	NA
Thamnophis spp.	0	0	0	0	0	0	1	0	0	NA	NA	NA
Virginia striatula	1	0	0	0	0	0	0	0	0	NA	NA	NA
Snakes	7	9	10	0	1	1	2	3	2	NA	NA	NA
Total reptiles	36	56	44	3	5	5	9	18	20	NA	NA	NA
Total captures [d]	103	260	228	17	31	19	57	159	291	NA	NA	NA
Species Richness [d]	16	16	17	5	11	8	14	15	14	13	14	13

individual-selection sites. Percentage litter ($F_{2,24}$=7.73, P=0.003) and bare ground ($F_{2,24}$=7.37, P=0.003), number of trees ($F_{2,24}$=14.31, P<0.001) and tree richness ($F_{2,24}$=6.98, P=0.004) were greater in uncut and individual-selection than group-selection sites. Number of shrubs was greater in individual-selection ($F_{2,24}$=6.06, P=0.007) than group-selection sites, with uncut sites similar to both. Shrub richness was greater in group-selection ($F_{2,24}$=4.44, P=0.023) than uncut sites, and similar in individual-selection sites to both. Percentage vine cover was greater in group-selection ($F_{2,24}$=16.37, P<0.001) than individual-selection or uncut sites.

Associations of Herpetofauna with Microhabitat

Drift Fence Arrays

None of the microhabitat variables measured at drift fence arrays were highly correlated ($\geq$ 0.8). So principal components analysis (PCA) was therefore performed with all variables included. Based on eigenvalues $\geq$ 1.0, 5 principal components were retained for the drift fence-pitfall data set, accounting for 62% of the variance (Table 3.4). Component 1 was interpreted to represent a gradient from areas with a greater percentage of vine cover (primarily *Rubus* spp.) and shrub species richness (positive loadings), to areas with a greater percentage of closed canopy and bare ground (negative loadings; Table 3.5). Tree richness and number of trees were positive loadings on component 2. Percentage woody ground cover, number of shrubs, and litter depth were positively loaded on component 3, whereas percentage herbaceous cover was negatively loaded. On component 4, percentage litter scored positively, whereas percentage fern cover, and vegetation height scored negatively. Finally, positive loadings on component 5 included fine and coarse woody debris, and stumps (downed woody debris, Table 3.5). Associations between microhabitat and occurrence of select herpetofauna captured at drift fence arrays were examined for amphibians, reptiles, eastern narrowmouth toad, bronze frog, southern

Table 3.2. Mean values ($\bar{X}$) and standard errors (SE) of microhabitat variables measured at drift fence arrays in group-selection stands (GS), individual-selection stands (IS) and uncut stands (US) in 2003 and 2004 at Sherburne WMA, LA.

	Stand Type					
	Group-selection (n[a]=12)		Individual-selection (n=15)		Uncut (n=15)	
Microhabitat Variable	$\bar{X}$ [b]	SE	$\bar{X}$	SE	$\bar{X}$	SE
% Canopy closure	77.68B[c]	1.72	89.73A	0.57	90.33A	0.77
% Litter	31.34B	2.85	46.82A	2.53	39.01AB	1.99
% Herbaceous cover	11.32A	2.07	7.28A	0.93	6.83A	0.94
% Woody[d]	10.50A	1.51	5.33B	0.77	3.67B	0.52
% Vine[d]	17.33A	2.23	13.50AB	1.12	8.86B	0.82
% Fern	10.72B	1.83	10.72B	2.19	22.98A	2.34
% Palmetto[e]	0.00	0.00	0.00	0.00	0.18	0.11
% Rock[e]	0.00	0.00	0.00	0.00	0.00	0.00
% Moss[e]	0.05	0.04	0.10	0.06	0.07	0.05
% Water[e]	3.46	1.60	3.38	1.52	0.25	0.25
% Bare ground	2.65B	0.99	4.18A	0.69	7.23A	1.09
Vegetation height (cm)	29.85A	1.98	17.6B	1.70	17.43B	1.31
Litter depth (cm)	1.69A	0.12	1.73A	0.10	1.63A	0.08
# Fine woody debris	21.17A	3.81	24.10A	2.91	26.17A	2.00
# Coarse woody debris	4.25A	0.87	3.60A	0.59	4.40A	0.68
# Stumps	0.88A	0.31	2.50A	0.83	1.40A	0.86
# Shrubs (<8.0 cm dbh)	48.58A	8.33	61.20A	8.63	34.20A	3.87
# Trees (≥8.0 cm dbh)	4.13A	0.79	5.47A	0.59	4.83A	0.37
Shrub species richness	8.42A	0.80	5.80B	0.31	5.27B	0.29
Tree species richness	2.17A	0.39	3.00A	0.29	2.60A	0.20

[a] n = number of arrays per type of harvest strategy.
[b] $\bar{X}$ = average of 4 measurements per array, per season (2), per year (% Canopy closure - % Bare ground);
$\bar{X}$ = average of 16 measurements per array, per season (2), per year (Vegetation height and Litter depth);
$\bar{X}$ = value per array, per year (# Fine woody debris – Tree species richness).
[c] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons (alpha=0.10).
[d] Values based on 2004 measurements only; [e] Insufficient data for analysis.

Table 3.3. Mean values (X) and standard errors (SE) of microhabitat variables associated with visual encounter surveys by stand type, measured in 2003 and 2004 at Sherburne WMA, LA.

	Stand Type					
	Group-selection (n[a] =12)		Individual-selection (n=15)		Uncut (n=15)	
Microhabitat Variable	$\bar{X}$ [b]	SE	$\bar{X}$	SE	$\bar{X}$	SE
% Canopy cover	89.12A [c]	2.49	84.98B	2.04	93.60A	0.23
% Litter	17.50B	1.96	33.89A	2.08	32.37A	2.54
% Herbaceous cover	0.12A	0.03	0.11A	0.01	0.09A	0.01
% Woody cover[d]	30.59A	5.47	12.26A	1.47	12.14A	1.33
% Vine cover[d]	33.99A	4.88	9.66B	1.71	10.21B	1.25
% Fern cover	9.67B	2.48	17.65AB	2.92	24.78A	2.64
% Bare ground	2.71B	0.76	7.82A	1.11	7.94A	1.25
% Water	2.50A	1.30	3.17A	1.78	1.79A	1.05
% Moss	0.05A	0.03	0.14A	0.09	0.06A	0.04
% Coarse woody debris	8.03A	0.97	9.14A	0.71	7.15A	0.71
# Shrubs	111.58AB	10.53	224.97A	27.84	96.18B	13.72
# Trees	15.29B	1.76	26.77A	1.65	30.73A	1.54
Shrub species richness	11.67B	0.70	10.07A	0.35	8.87A	0.32
Tree species richness	6.67B	0.44	8.67A	0.26	8.50A	0.34

[a] n = number of arrays per type of harvest strategy.

[b] $\bar{X}$ = average of 11 measurements per array per year (% Canopy closure - % Coarse woody debris);
$\bar{X}$ = value per array per year (shrub and tree species richness).

[c] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparison (alpha=0.10).

[d] Values based on 2004 measurements only.

leopard frog, and ground skink. Each of these was the dependent variable in analyses of model sets. The full model did not converge for data representing occurrence of ground skink and thus could not be analyzed. Results for models ΔAICc values ≤ 2 are listed in Table 3.6. Model averaged parameter estimates, unconditional standard errors, and relative importance ($\sum w_i$) of each component variable are presented in Table 3.7.

Table 3.4. Principal components (PC) analysis results for microhabitat variables surveyed at drift fence arrays at Sherburne WMA, LA.

	PC 1	PC 2	PC 3	PC 4	PC 5
Eigenvalue	2.89	2.35	1.86	1.50	1.33
Variance explained	0.18	0.15	0.12	0.09	0.08
Variables:					
% Canopy	**-74** [a,b]	27	-6	3	-10
% Litter	-11	4	26	**87**	0
% Herbaceous cover	19	-26	**-67**	-22	13
% Woody cover	39	-18	**51**	9	6
% Vine cover	**68**	-8	-19	-9	7
% Fern cover	-40	22	24	**-55**	-39
% Bare ground	**-46**	-13	-19	-34	34
Vegetation height (cm)	26	-22	27	**-73**	3
Litter depth (cm)	-8	0	**76**	-9	-3
# Fine woody debris	-1	20	-24	8	**72**
# Coarse woody debris	2	-24	9	-18	**75**
# Stumps	2	9	-3	10	**64**
# Shrubs	19	-36	**58**	-18	-16
# Trees	-7	**87**	-3	7	0
Shrub species richness	**67**	13	7	-16	-8
Tree species richness	-4	**89**	0	5	6

[a] Correlation coefficients are multiplied by 100 and rounded to the nearest integer.
[b] Values > |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.

Table 3.5. Principal components (PC, eigenvalues $\geq$ 1) derived from microhabitat variables measured at drift fence arrays at Sherburne WMA, LA, 2003 and 2004. Associated variables are those with a correlation coefficient of $\geq |0.40|$ with each respective PC.

PC	Associated Variables[a]	Interpretation
PC1	% Vine cover, Shrub species richness, (-) % Canopy closure, (-) % Bare ground	Vine cover and shrub richness
PC2	# Trees and Tree species richness	Tree richness and density
PC3	% Woody cover, Litter depth, # Shrubs, (-) % Herbaceous cover	Litter depth, woody ground cover, and number of shrubs
PC4	% Litter cover, (-) % Fern cover, (-) Vegetation height	Litter cover
PC5	# Fine woody debris, # Coarse woody debris, # Stumps	Downed woody debris

[a] Variables are positively related to the principal component unless otherwise noted.

Table 3.6. Model selection results for select herpetofauna detected at drift fence arrays at Sherburne WMA, LA, 2003-2004. Model subsets include models with substantial empirical support ($\Delta AICc \leq 2$) in explaining associations of herpetofauna with microhabitat. K is number of model parameters, AICc equals Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and the best model, and w_i is Akaike weight.

Species	Model[a]	K	Deviance	AIC_c	ΔAIC_c	w_i
Amphibian occurrence	PC3	2	506.44	510.46	0	0.307
	PC2	2	508.13	512.16	1.70	0.132
Reptile occurrence	PC2	2	411.48	415.50	0	0.172
	PC1	2	411.88	415.91	0.41	0.140
	PC4	2	412.17	416.20	0.69	0.121
	PC1 PC2	3	411.07	417.13	1.62	0.076
	PC2 PC4	3	411.43	417.48	1.97	0.064
Eastern narrowmouth toad occurrence	PC2	2	275.45	279.48	0	0.178
	PC1 PC2 PC5	4	271.98	280.06	0.58	0.133
	PC1 PC2	3	274.44	280.49	1.01	0.108
	PC2 PC5	3	274.92	280.97	1.49	0.084
	PC2 PC4	3	275.23	281.28	1.80	0.072
	PC1 PC2 PC4 PC5	5	271.32	281.45	1.97	0.066
Bronze frog occurrence	PC3	2	196.53	200.55	0	0.248
	PC2 PC3	3	194.94	200.99	0.44	0.199
	PC3 PC5	3	196.20	202.25	1.70	0.106
	PC2 PC3 PC5	4	194.42	202.55	1.99	0.092
	PC1 PC3	3	196.50	202.55	1.99	0.091
Southern leopard frog occurrence	PC1 PC3	3	228.58	234.63	0	0.135
	PC1	2	231.14	235.17	0.54	0.103
	PC1 PC2	3	229.47	235.54	0.91	0.085
	PC1 PC2 PC3	4	227.47	235.55	0.93	0.085
	PC1 PC3 PC4	4	227.47	235.56	0.93	0.085
	PC1 PC4	3	229.65	235.70	1.08	0.079
	PC1 PC2 PC4	4	228.19	236.27	1.65	0.059
	PC1 PC2 PC3 PC4	5	226.49	236.62	1.99	0.50

[a]PC1= vine cover and shrub richness, PC2= tree richness and density, PC3= litter depth, woody ground cover, number of shrubs, PC4= litter, PC5= downed woody debris.

Table 3.7. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of microhabitat, for select herpetofauna detected at drift fence arrays at Sherburne WMA, LA, 2003-2004.

Species	Model variable[a]	Estimate[b]	Standard error	$\sum w_i$
Amphibian occurrence	PC1	-0.4948	0.1211	0.174
	PC2	0.0435	0.1009	0.379
	PC3	-0.0440	0.1056	0.596
	PC4	0.1863	0.1114	0.092
	PC5	0.1046	0.1110	0.226
Reptile occurrence	PC1	0.0571	0.1148	0.408
	PC2	0.0406	0.1107	0.466
	PC3	0.0354	0.1151	0.215
	PC4	-0.0541	0.1206	0.363
	PC5	0.0696	0.0867	0.147
Eastern narrowmouth toad occurrence	PC1	-0.4160	0.1614	0.524
	PC2	0.2190	0.1241	0.979
	PC3	0.2028	0.1357	0.261
	PC4	0.0315	0.1302	0.315
	PC5	-0.1487	0.1366	0.455
Bronze frog occurrence	PC1	-0.6271	0.2171	0.221
	PC2	-0.0489	0.1428	0.371
	PC3	-0.8621	0.1704	0.999
	PC4	0.4489	0.1640	0.195
	PC5	0.1505	0.1515	0.299
Southern leopard frog occurrence	PC1	-0.4019	0.1715	0.838
	PC2	-0.0745	0.1310	0.428
	PC3	-0.3618	0.1465	0.571
	PC4	0.2784	0.1472	0.412
	PC5	0.0487	0.1389	0.195

[a] PC1= vine cover and shrub richness, PC2= tree richness and density, PC3= litter depth, woody ground cover, number of shrubs, PC4= litter, PC5= downed woody debris.
[b] Abundance data analyzed with logistic ANOVA. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003)

The best approximating model for occurrence of amphibians retained one component variable, which represented litter depth, woody ground cover, and number of shrubs (component 3). As litter depth, woody ground cover and number of shrubs increased, occurrence of

amphibians decreased. Five models gained substantial empirical support in explaining occurrence of reptiles. Among these models, vine cover and shrub richness (component 1), tree richness and number of trees (component 2), and litter (component 4), were retained both as sole model components and in combination with each other. Vine cover, shrub richness, tree richness, and number of trees positively influenced occurrence of reptiles, whereas litter had a negative influence.

The best approximating model retained one variable, tree richness and number of trees (component 2), which positively influenced occurrence of eastern narrowmouth toad, and had the highest estimate of importance relative to other component variables (Table 3.7). In contrast, occurrence of bronze frog was best explained by the model which retained litter depth, woody ground cover, and number of shrubs (component 3), which negatively occurrence of bronze frog and had the greatest relative Akaike weight among component variables. Litter depth, woody ground cover, and number of shrubs also negatively influenced occurrence of southern leopard frog, but was not the only variable retained in the best approximating model for occurrence of southern leopard frog. Vine cover and shrub richness (component 1) also negatively influenced occurrence of southern leopard frog.

Visual Encounter Surveys

None of the microhabitat variables sufficiently measured along visual encounter survey routes was highly correlated; thus, all were included in the PCA. Six principal components representing microhabitat were retained, accounting for 70.7% of the variance in the data set (Table 3.8). Component 1 was interpreted as a gradient from areas with a greater percentage of litter cover and number of trees (positive loadings), to areas with a greater percentage of vine cover (*Rubus* spp.) and shrub species richness (negative loadings) (Table 3.9). Coarse woody debris scored positively on component 2, whereas percentage fern cover had a negative score.

Component 3 was interpreted to represent areas with greater amounts of percentage herbaceous cover and water, which were both positively loaded. Percentage woody ground cover was positively loaded on component 4, whereas percentage bare ground was negatively loaded. On component 6, percentage moss cover scored positively, whereas tree richness scored negatively (Table 3.9). Associations between microhabitat and relative abundance of amphibians, reptiles, spring peeper and Ranids were examined using the first 5 principal components as explanatory variables in tests of all possible models (n=31), with relative abundance of each group or species as the dependent variable in analysis of each model set. Principal component 6 was not included in model testing because moss was not considered influential in occurrence of herpetofauna. The best approximating model for abundance of amphibians included percentage woody ground cover (component 4) and number of shrubs (component 5), which both negatively influenced abundance of amphibians, and had greater relative Akaike weights than other components. Relative abundance of reptiles was best approximated by the model which retained percentage coarse woody debris (component 2), percentage woody ground cover (component 4), and number of shrubs (component 5). This was the only model which received substantial empirical support. Both percentage woody ground cover and number of shrubs negatively influenced abundance of reptiles, whereas percentage coarse woody debris had a positive influence.

Number of trees and litter cover (component 1), herbaceous cover and water (component 3), and woody ground cover (component 4), were the 3 component variables within the subset of models which gained support in explaining relative abundance of spring peeper, among which percentage herbaceous cover and water was more important. Abundance of spring peeper increased with increasing percentage of herbaceous cover and water (Table 3.11).

Finally, the best approximating model for relative abundance of ranids included 3 component variables, percentage coarse woody debris (component 2), herbaceous cover and

water (component 3), and number of shrubs (component 5). In this model, relative abundance of ranids increased with increasing percentages of coarse woody debris, herbaceous cover and water, and with decreasing number of shrubs. Estimates of relative importance 3 component variables were similar and greater than other component variables.

Table 3.8. Principal components (PC) analysis results for microhabitat variables surveyed along visual encounter survey routes at Sherburne WMA, LA.

	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6
Eigenvalue	2.99	2.03	1.38	1.26	1.15	1.09
Variance explained	0.21	0.14	0.10	0.09	0.08	0.08
Variables:						
% Canopy closure	21[a]	-38	19	17	**-58** [b]	-3
% Litter	**61**	48	-2	-41	-10	-20
% Herbaceous cover	-5	8	**62**	-4	-23	43
% Woody cover	-3	-10	-19	**72**	6	-14
% Vine cover	**-73**	29	-7	39	-14	-11
% Fern cover	27	**-67**	-42	16	31	8
% Bare ground	14	-28	-46	**-62**	-3	4
% Water	5	-3	**83**	-7	8	-9
% Moss	9	2	5	-17	3	**78**
% Coarse woody debris	1	**79**	-4	15	12	15
# Shrubs	3	-12	7	15	**87**	-3
# Trees	**84**	4	8	9	-5	-13
Shrub species richness	**-77**	9	16	3	7	-14
Tree species richness	40	-23	22	-42	3	**-51**

[a] Correlation coefficients are multiplied by 100 and rounded to the nearest integer.
[b] Values > |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.

Table 3.9. Principal components (PC, eigenvalues $\geq$ 1) derived from microhabitat variables measured along visual encounter survey routes at Sherburne WMA, LA. Associated variables are those with a correlation coefficient of $\geq |0.40|$ to each respective PC.

PC	Loaded Variables[a]	Interpretation
PC1	% Litter, # Trees, (-) % Vine cover, (-) Shrub richness	Number of trees and litter cover
PC2	# Coarse woody debris, (-) % Fern cover	Coarse woody debris
PC3	% Herbaceous cover, % Water	Herbaceous cover and water
PC4	% Woody ground cover (-) % Bare ground	Woody ground cover
PC5	# Shrubs, (-) % Canopy closure	Number of shrubs
PC6	% Moss, (-) Tree richness	Moss

[a] Variables are positively related to the principal component unless otherwise noted.

Table 3.10. Model selection results for select herpetofauna detected in visual encounter surveys at Sherburne WMA, LA, 2003-2004. Model subsets include models with substantial empirical support (ΔAICc $\leq$ 2) in explaining associations of herpetofauna with microhabitat. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and the best model, and w_i is Akaike weight.

Species	Model[a]	K	Deviance	AIC_c	ΔAIC_c	w_i
Amphibian abundance	PC4 PC5	3	86.36	92.66	0	0.302
	PC5	2	89.30	93.45	0.79	0.203
	PC3 PC4 PC5	4	85.22	93.72	1.06	0.177
Reptile abundance	PC2 PC4 PC5	4	41.80	50.30	0	0.308
Spring peeper abundance	PC3	2	39.11	43.26	0	0.248
	PC3 PC4	3	38.49	44.79	1.53	0.115
	PC1 PC3	3	38.78	45.08	1.82	0.100
Rana spp. abundance	PC2 PC3 PC5	4	19.50	28.01	0	0.145
	PC2	2	25.22	29.37	1.36	0.074
	PC4	2	25.32	29.47	1.46	0.070
	PC1 PC3 PC5	4	21.14	29.65	1.64	0.064
	PC2 PC4	3	23.65	29.95	1.94	0.055

[a]PC1= number of trees and litter cover, PC2= coarse woody debris, PC3= herbaceous cover and water, PC4= woody ground cover, PC5= number of shrubs.

Table 3.11. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of microhabitat, for select herpetofauna detected during visual encounter surveys at Sherburne WMA, LA, 2003-2004.

Species	Model variable[a]	Estimate[b]	Standard error	$\sum w_i$
Amphibian relative abundance	PC1	0.2008	0.1737	0.044
	PC2	0.0807	0.1335	0.101
	PC3	0.0755	0.1277	0.309
	PC4	-0.2417	0.1395	0.620
	PC5	-0.3016	0.1841	0.850
Reptile relative abundance	PC1	0.0869	0.1970	0.293
	PC2	0.1675	0.1818	0.688
	PC3	-0.1012	0.2079	0.256
	PC4	-0.2376	0.2037	0.806
	PC5	-0..3725	0.2293	0.870
Spring peeper relative abundance	PC1	0.4798	0.3048	0.241
	PC2	0.0759	0.2251	0.178
	PC3	0.4142	0.2366	0.756
	PC4	-0.3679	0.2126	0.260
	PC5	-0.0168	0.2957	0.309
Rana spp. relative abundance	PC1	0.6083	0.2657	0.314
	PC2	0.2645	0.1837	0.507
	PC3	0.2890	0.1786	0.569
	PC4	0.0797	0.2327	0.362
	PC5	-0.4246	0.2753	0.504

[a] PC1= number of trees and litter cover, PC2= coarse woody debris, PC3= herbaceous cover and water, PC4= woody ground cover, PC5= number of shrubs.

[b] Abundance data analyzed with logistic ANOVA. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003).

RESULTS: BEN'S CREEK WMA

Associations of Herpetofauna with Stand Age

Between April 2003 and November 2004, we detected 32 species of herpetofauna (21 amphibians and 11 reptiles) based on the combination of all standardized survey methods and chance encounters. Mean species richness in 1-year (15.25 $\pm$ 2.18), 4-5-year (11.5 $\pm$ 2.63), 7-9-year (12.75 $\pm$ 1.65), and 13-23-year stands (18.00 $\pm$ 2.04) was similar ($F_{3,12}$=2.38, P=0.1208). We recorded 179 captures from drift fences, cover boards and visual encounter surveys combined (detections during anuran calling surveys were indexed and thus not included in this figure). Bullfrog (*Rana catesbeiana*) detections were unique to 1-year stands. Eastern hognose snake (*Heterodon platyrhinos)*, long-tailed salamander (*Eurycea longicauda*) and broadhead skink (*Eumeces laticeps*) detections were unique to 13-23-year stands. Gulf coast toad (*Bufo valliceps*) and eastern narrowmouth toad (*Gastrophryne carolinensis*) were detected by all survey methods. Ground skink (*Scinella lateralis*), was detected by all 3 survey methods capable of detecting reptiles (excludes anuran calling surveys) (Table 3.12).

Drift Fence Arrays

We captured 141 individuals (10 amphibian and 6 reptile species) at drift fences between April 2003 and November 2004 (12,079 trap days). Eastern narrowmouth toad was the most common amphibian, constituting 61% of amphibian captures and 36% of total captures. Green anole (*Anolis carolinensis*) and ground skink were the most common reptile, constituting 33% and 35% of reptile captures, and 13% and 14% of total captures, respectively. Captures of southern leopard frog (*Rana utricularia*) were unique to 1-year stands. Captures of Woodhouse's toad (*B. woodhousii*), eastern spadefoot toad (*Scaphiopus holbrookii),* long-tailed salamander (*Eurycea longicauda*), broadhead skink (*E. laticpes*) and three-toed box turtle (*Terrapene Carolina triunguis*) were unique to 13-23-year stands. Eastern spadefoot toad, long-

tailed salamander, five-lined skink (*E. fasciatus*), broadhead skink, and three-toed box turtle, were detected only with drift fences.

Species richness and relative abundance of amphibians did not differ across years ($F_{1,10.8}$=0.02, P=0.902 and $F_{1,10.9}$=0.00, P=0.982, respectively) or seasons ($F_{1,11.1}$=1.73, P=0.215 and $F_{1,11.1}$=2.00, P=0.185, respectively). Similarly, species richness and relative abundance of reptiles did not differ across years ($F_{1,3.89}$=0.65, P=0.467 and $F_{1,3.88}$=0.59, P=0.486, respectively), but were greater in fall than summer ($F_{1,13.4}$=4.00, P=0.066 and $F_{1,13.4}$=4.13, P=0.063, respectively). Species richness of amphibians was greater ($F_{3,86.1}$= 4.31, P = 0.007) in 1- and 13-23-year stands than 7-9-year stands, with 4-5-year stands similar to other stand age classes. Species richness of reptiles was greater ($F_{3,77.2}$= 3.12, P = 0.031) in 13-23-year than 1- and 4-5-year stands, with 7-9-year stands similar to other stand age classes. Relative abundance of amphibians also differed by stand age ($F_{3,85.2}$= 4.26, P = 0.008). Relative abundance was greater in 1- and 13-23-year stands than 7-9-year stands, and similar in 4-5-year stands to other stand age classes. Lizards constituted all but one capture of the total number of reptiles captured (*n*=57). Therefore, tests for effects of stand age on lizard abundance, rather than reptile abundance, were deemed more appropriate. Relative abundance of lizards varied across stand age ($F_{3,72.8}$= 2.19, P = 0.0961), with more lizard captures in 13-23-year stands. However, after Tukey-Kramer adjustments, abundance was similar across stand age.

Anurans accounted for 86% of amphibian captures (*n*=84). Relative abundance of anurans was greatest ($F_{3,84.7}$= 4.09, P = 0.009) in 1- and 13-23-year stands. Although we captured 4 species and 1 genus (*Bufo* spp.) $\geq$ 10 times during the study period, the eastern narrowmouth toad was the only species with a sufficient number of captures across the study period for data analysis. Abundance of eastern narrowmouth toad was greater in 1-year than 7-9-year stands ($F_{3,103}$= 2.51, P = 0.063), with 4-5-year and 13-23-year stands similar to both.

Cover Boards

We observed 20 individuals (4 amphibian and 2 reptile species) at cover boards between April 2003 and October 2004 (3,312 trap days). Eastern narrowmouth toad was the most common (40% of detections) amphibian detection and green anole was the most common (25% of captures) reptile capture. Species richness and relative abundance of herpetofauna did not differ across years ($F_{1,5.78}$=2.84, P=0.145 and $F_{1,5.5}$=2.42, P=0.176, respectively). Low capture rates did not allow for tests of differences across season. Further, low capture rates only allowed for tests of effects of stand age on species richness and relative abundance of all captures combined; neither ($F_{3,46.2}$= 0.86, P = 0.467; $F_{3,46.3}$= 0.92, P = 0.439) varied across stand age.

Visual Encounter Surveys

We detected 19 individuals (4 amphibian and 2 reptile species) during spring 2003 visual encounter surveys (n = 48). Woodhouse's toad (*Bufo woodhousii*) was the most common (25%) amphibian detected and the ground skink was the most common (20%) reptile detected. Data was insufficient for statistical tests of effects of stand age on species richness or relative abundance. However, comparison of means across stand age indicated that both anurans and lizards were captured more frequently in 13-23-year stands. The gulf coast toad and eastern narrowmouth toad were exceptions to this trend; abundance of gulf coast toad was similar in 4-5- and 13-23-year stands, and abundance of eastern narrowmouth toad was greatest in 1-year stands.

Anuran Calling Surveys

We detected 18 species from June 2003-June 2004 (320 surveys). Spring peeper (*Pseudacris crucifer*), green treefrog (*Hyla cinera*), and gray treefrog (*H. versicolor*) were the most frequently detected anurans. Species richness and relative abundance of anurans did not differ across years ($F_{1,3.14}$=0.14, P=0.735 and $F_{1,3.53}$=0.00, P=0.948, respectively). There was

sufficient data to test effects of stand age on species richness, relative abundance of anurans, and relative abundance of spring peeper. Species richness was greater in 1-year than 7-9-year stands ($F_{3,28.6}$=2.43, P=0.086), with 4-5- and 13-23-year stands similar to both. Relative abundance of anurans ($F_{3,27.8}$=1.51, P=0.234) and spring peeper ($F_{3,44.8}$=0.73, P=0.539) did not vary across stand age.

Associations of Microhabitat with Stand Age

Eighteen variables differed across forest age (Table 3.13). Values of 9 variables generally followed a gradient that positively increased from 1- to 7-9-year stands. Number of pine shrubs was lower in 13-23-year stands ($F_{3,27}$ = 24.51, P <0.001) than other stand age classes. Percentage grass was greater ($F_{3,57}$ = 57.12, P <0.001) in 1- and 4-5-year stands than older age classes. Percentage vine cover ($F_{3,57}$ = 4.92, P = 0.004) and density of coarse woody debris ($F_{3,27}$ = 2.95, P = 0.051) were greater in 1-year than 7-9-year stands, and similar in 4-5-and 13-23-year stands to both. Fine woody debris was greater ($F_{3,27}$ = 15.42, P <0.001) in 1- and 13-23-year stands than other stand age classes. Percentage bare ground ($F_{3,57}$ = 21.24, P <0.001) and number of stumps ($F_{3,27}$ = 11.97, P <0.001) was greater in 1-year stands than other stand age classes. Percentage forb cover was greatest in 1-year stands, decreased in 13-23-year stands, and was lowest in 7-9-year stands ($F_{3,57}$ = 11.58, P <0.001. Vegetation height was greatest in 1-year stands, followed by 13-23-year stands, and least in 7-9-year stands ($F_{3,57}$ = 19.94, P <0.001).

Values of the other 9 variables that differed across stand age class followed an inverse gradient to those above. Percentage leaf litter ($F_{3,57}$ = 15.02, P <0.001) and pine litter ($F_{3,57}$ = 39.87, P <0.001), and density of trees ($F_{3,27}$ = 19.44, P <0.001) and pine trees ($F_{3,27}$ = 26.27, P <0.001) were greatest in 7-9-year stands, similar in 4-5- and 13-23-year stands, and least in 1-year stands. Canopy closure was greatest ($F_{3,57}$ = 88.61, P <0.001) in 7-9-year stands, followed by 13-23-, 4-5-, and 1-year stands. Shrub richness was lower in 1-year stands than 7-9- and 13-

Table 3.12. Herpetofauna captures/detections using drift fence arrays, cover boards, visual encounters, anuran calling surveys and chance encounters (denoted by X) between April 2003 – November 2004 in 1 year (A), 4-5 year (B), 7-9 year (C), and 13-23 year (D) stands at Ben's Creek WMA, LA.

	Herpetofauna Survey Method															
	Drift fence arrays [a]				Cover boards [a]				Visual encounters [b]				Anuran calling surveys [bc]			
Stand Age Class	A	B	C	D	A	B	C	D	A	B	C	D	A	B	C	D
Effort (# trap days)	3020	3017	3022	3020	827	835	839	841	12	12	12	12	16	16	16	16
<u>Amphibians</u>																
Bufo americanus	1	0	0	2	0	0	0	0	0	X	X	0	2	2	1	2
B. terrestris	2	0	1	4	1	0	0	0	0	0	0	0	1	2	1	1
B. valiceps	0	0	1	1	0	1	0	0	0	1	0	1	2	2	1	1
B. woodhousii	0	0	0	3	0	0	0	0	0	0	0	5	1	1	0	1
Gastrophryne carolinensis	28	10	1	12	4	3	1	0	1	0	0	0	2	1	0	1
Scaphiopus holbrookii	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Acris creptians	0	0	0	0	0	0	0	0	X	0	0	0	2	1	0	1
A. gryllus	0	0	0	0	0	0	0	0	X	0	0	1	2	1	2	2
Unknown *Acris* spp.	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0
Hyla avivoca	0	0	0	0	0	0	0	0	0	0	0	0	2	1	1	2
H. chrysoscelis	0	0	0	0	0	0	0	0	0	0	0	0	2	2	3	3
H. cinera	0	0	0	0	0	0	0	0	X	0	0	0	3	3	3	3
H. femoralis	0	0	0	0	0	0	0	0	0	0	0	2	3	2	1	1
H. gratiosa	0	0	0	0	0	0	0	0	0	0	0	0	3	3	2	3
H.squirella	0	0	0	0	0	0	0	0	0	0	0	0	3	3	2	0
H. versicolor	0	0	0	0	0	0	0	0	0	0	0	0	3	2	3	3
Pseudacris creptians	0	0	0	0	0	0	0	0	0	0	0	0	3	3	3	3
Rana catesbeiana	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
R. clamatans	0	1	0	1	0	0	0	0	0	0	X	0	2	3	3	3
R. utricularia	3	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Anurans	34	11	3	24	5	4	1	0	2	1	0	10	NA	NA	NA	NA
Eurycea longicauda	0	0	0	6	0	0	0	0	0	0	0	0	NA	NA	NA	NA
Plethodon glutinosus	0	0	1	5	0	0	1	2	0	0	0	0	NA	NA	NA	NA
Total amphibians	34	11	4	35	5	4	2	2	2	1	0	10	NA	NA	NA	NA

Table 3.12 continued

	Herpetofauna Survey Method															
	Drift fence arrays				Cover boards				Visual encounters				Anuran calling surveys			
Stand Age Class	A	B	C	D	A	B	C	D	A	B	C	D	A	B	C	D
<u>Reptiles</u>																
Anolis carolinensis	3	2	5	9	1	0	0	4	0	0	0	0	NA	NA	NA	NA
Eumeces fasciatus	0	0	0	2	0	0	0	0	0	0	0	X	NA	NA	NA	NA
E. laticeps	0	0	1	1	0	0	0	0	X	0	0	X	NA	NA	NA	NA
Scinella lateralis	0	0	2	18	0	0	0	2	0	0	X	4	NA	NA	NA	NA
Sceloporus undulates	6	2	0	5	0	0	0	0	0	0	X	2	NA	NA	NA	NA
Lizards	9	4	8	35	1	0	0	6	0	0	0	6	NA	NA	NA	NA
Gopherus ployphemus	0	0	0	0	0	0	0	0	0	X	X	0	NA	NA	NA	NA
Terrapene Carolina t.	0	0	0	1	0	0	0	0	0	0	X	X	NA	NA	NA	NA
Coluber constrictor	0	0	0	0	0	0	0	0	X	X	0	X	NA	NA	NA	NA
Elaphe obsolete l.	0	0	0	0	0	0	0	0	X	0	0	0	NA	NA	NA	NA
Heterodon platyrhinos	0	0	0	0	0	0	0	0	0	0	0	X	NA	NA	NA	NA
Lampropeltis getulus	0	0	0	0	0	0	0	0	0	0	X	X	NA	NA	NA	NA
Total captures [d]	43	15	12	71	6	4	2	8	2	1	0	16	NA	NA	NA	NA
Species richness [d]	6	4	7	15	3	2	2	3	2	1	0	6	18	16	14	16

[a] Effort = [# captures/# functional traps] x 20 per array per survey period per year

[b] Effort = total # of surveys during study period.

[c] Highest index value (0-3) assigned during study period: 1= individuals can be counted, 2= individuals distinguished but calls overlapping, 3= full chorus.

[d] Does not include chance encounters.

23-year stands ($F_{3,27} = 4.79$, $P = 0.008$), with 4-5-year stands similar to other stand age classes. Litter depth was greatest in 7-9- and 13-23-year stands and least in 1-year stands ($F_{3,57} = 34.57$, $P < 0.001$). Density of non-pine trees was greater ($F_{3,27} = 3.13$, $P = 0.042$) in 13-23-year stands than 1-year stands, and similar in 4-5- and 7-9-year stands to other stand age classes. Finally, tree species richness was greatest in 13-23-year stands and was lowest in 1-year stands ($F_{3,27} = 30.49$, $P < 0.001$; Table 3.13).

Associations of Herpetofauna with Microhabitat

Pearson's correlation coefficient tests between the 22 microhabitat variables retained for analysis resulted in 2 variables being dropped, including number of non-pine shrubs (< 8.0 cm) and number of pine trees ($\geq$ 8.0 cm). Hence 20 variables were included in the PCA. Based on eigenvalues $\geq$ 1.0, 6 principal components representing microhabitat were retained for the drift fence array data set, and accounted for 67.28% of the variance (Table 3.14).

Component 1 was interpreted to represent a gradient from sites with greater percentages of ground litter, number of trees and canopy closure (positive loadings) to sites with greater percentages of grass and forb cover and with high herbaceous vegetation (negative loadings) (Table 3.15). Component 2 was interpreted as a gradient from areas of greater litter depth and tree richness (positive loadings), to areas with more percentage bare ground and number of pine shrubs (negative loadings). Amount of fine woody debris, coarse woody debris and number of stumps loaded positively on component 3, collectively referred to as downed woody debris. Positive loadings on component 4 included shrub richness and number of shrubs. Percentage fern cover had a positive score on component 5, whereas percentage vine cover had a negative score. Component 5 represented percentage woody ground cover (Table 3.15). Associations between microhabitat and occurrence of select herpetofauna captured at drift fence arrays were examined using the 6 principal components as explanatory variables in tests of all possible

Table 3.13. Mean values ($\bar{X}$) and standard errors (SE) of microhabitat variables measured at drift fence arrays in 1 year, 4-5 year, 7-9 year and 13-23 year stands (*n*=12 arrays per age class) in 2003 and 2004 at Ben's Creek WMA, LA.

Stand Age Class:	1 Year		4-5 Year		7-9 Year		13-23 Year	
Microhabitat	$\bar{X}$ [a]	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
% Leaf litter	5.64C [b]	0.85	13.00B	1.73	24.68A	1.99	17.79B	2.48
% Pine litter	3.47C	0.66	16.58B	1.62	37.57A	2.67	22.95B	1.68
% Grass	30.73A	3.32	24.40A	2.78	1.62B	0.81	3.46B	0.99
% Forb cover	7.45A	1.22	5.15AB	0.79	1.67C	0.52	3.84B	0.77
% Fern cover	0.52A	0.24	0.39A	0.17	0.52A	0.31	0.76A	0.38
% Vine cover	14.82A	2.49	9.91AB	1.55	5.26B	0.95	12.50AB	2.86
% Woody cover	23.78A	2.73	25.19A	1.64	23.33A	3.29	29.77A	3.46
% Bare ground	8.53A	1.25	2.19B	0.45	1.28B	0.35	1.12B	0.43
% Rock [e]	0.23	0.12	0.00	0.00	0.00	0.00	0.00	0.00
% Water [e]	0.00	0.00	0.00	0.00	0.00	0.00	0.59	0.59
% Moss [e]	0.63	0.27	0.14	0.07	0.33	0.15	0.57	0.31
% Canopy closure	8.22D	1.20	53.68C	3.28	82.14A	1.16	75.84B	1.75
Vegetation height (cm)	38.19A	2.53	24.26B	2.13	10.95C	1.64	18.48BC	2.37
Litter depth (cm)	0.66C	0.07	1.58B	0.12	2.15A	0.14	2.31A	0.16
# Fine woody debris	14.5A	2.05	4.13B	1.23	3.21B	0.54	19.42A	2.73
# Coarse woody debris	4.17A	1.09	2.08AB	0.95	0.75B	0.18	2.54AB	0.60
# Stumps	4.04A	0.46	0.96B	0.25	1.08B	0.28	2.04B	0.33
# Shrubs (<8.0cm dbh)	80.21A	5.86	110.67A	8.70	83.00A	6.71	86.54A	6.12
# *Pinus* shrubs	9.83A	0.57	8.08A	0.87	8.25A	1.41	0.21B	0.09
# Nonpine shrubs	70.38	5.89	102.58	8.35	74.75	6.32	86.33	6.11
Shrub species richness	9.08B	0.68	11.83AB	0.53	13.67A	0.84	12.63A	0.70
# Trees ($\geq$ 8.0 cm dbh)	0.00C	0.00	5.54B	0.98	12.00A	0.96	6.25B	0.74
# *Pinus* trees	0.00C	0.00	5.29B	0.99	11.67A	0.99	4.5B	0.57
Nonpine trees	0.00B	0.00	0.25AB	0.21	0.33AB	0.13	1.75A	0.86
Tree species richness	0.00C	0.00	0.96B	0.15	1.25AB	0.09	1.71A	0.23

[a] $\bar{X}$ = avg of 4 measurements per array, season and year (% Canopy closure - % Bare ground); X= avg of 16 measurements per array, season and year (Vegetation height and Litter depth); X= value per array, per year (# Fine woody debris – Tree species richness).
[b] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons.
[c] Values based on 2004 measurements only; [e] Insufficient data for analysis.

Table 3.14. Principal components (PC) analysis results for microhabitat variables measured at drift fence arrays (n=48) at Ben's Creek WMA, LA.

	PC1	PC2	PC3	PC4	PC5	PC6
Eigenvalue:	5.52	2.20	1.88	1.60	1.16	1.10
Variance explained:	0.28	0.11	0.09	0.08	0.06	0.06
Variables:						
% Leaf litter	**70**[a,b]	-1	-7	-3	39	-23
% Other litter	**71**	22	-28	10	-13	-19
% Grass	**-79**	-29	-19	2	4	-26
% Forb cover	**-54**	25	16	-27	19	6
% Fern cover	-2	2	10	12	**79**	4
% Bare ground	-17	**-40**	39	-17	-22	-23
% Canopy cover	**78**	36	-27	10	12	15
Vegetation height (cm)	**-81**	-7	5	7	-11	-1
Litter depth (cm)	28	**56**	-23	23	-14	20
% Vine cover (2004 only)	-31	38	42	-5	**-47**	7
% Woody cover (2004 only)	-5	-5	2	7	1	**94**
# Fine woody debris	2	4	**76**	-13	30	4
# Coarse woody debris	-1	3	**70**	1	-10	14
# Stumps	-29	-16	**70**	8	1	-20
# Shrubs (< 8.0cm)	-17	-12	-1	**82**	10	1
# Pine shrubs (< 8.0cm)	-30	**-44**	-19	21	-25	3
Shrub species richness	32	26	0	**76**	5	10
# Trees ($\geq$ 8.0cm)	**54**	40	-50	-14	-9	11
# Non-pine trees ($\geq$ 8.0cm)	-9	**82**	4	-10	-8	-15
Tree species richness	34	**82**	-9	12	3	0

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.

models (n = 63). Occurrence of amphibians, reptiles, and eastern narrowmouth toad were dependent variables in analyses of model sets.

Five models received substantial empirical support in explaining occurrence of amphibians detected at drift fence arrays. These 5 models were comprised of varying combinations of 4 component variables: percentage canopy closure and percentage litter (component 1); shrub richness and density (component 4); percentage fern cover (component 5); and percentage woody ground cover (component 6). Among these 5 models, the best approximating model (ΔAIC_c=0) retained 2 component variables, shrub richness and density, and

percentage fern cover, which both negatively influenced occurrence of amphibians (Tables 3.16 and 3.17). Occurrence of reptiles was best explained by the model which retained percentage canopy closure and percentage litter (component 1), which was present in each of the 6 models with substantial empirical support (Table 3.16). As percentage canopy cover and litter increased, occurrence of reptiles also increased. Additionally, percentage fern cover (component 5) and woody ground cover (component 6) negatively influenced occurrence of reptiles (Table 3.17).

All of the 6 component variables were present in varying combinations across the 14 candidate models that received substantial empirical support in explaining occurrence of eastern narrowmouth toad. However, the best approximating model retained only 2 component variables, shrub richness and density (component 4) and percentage woody ground cover (component 6). Occurrence of eastern narrowmouth toad increased with increasing percentage woody ground cover, and decreasing shrub richness and density. However, the large number of candidate models and associated small model weights (w_i, Table 3.16) suggests that these models did not account for much variation in occurrence of eastern narowmouth toad.

Table 3.15. Principal components (PC, eigenvalues $\geq$ 1.0) derived from microhabitat variables measured at drift fence arrays at Ben's Creek WMA, LA. Variables associated with each PC have a correlation coefficient $\geq |0.40|$.

PC	Loaded Variables[a]	Interpretation
PC1	% Leaf litter, % Pine straw, % Canopy cover, #Trees, (-) %Grass, (-) % Forb, (-) Vegetation height (cm)	Canopy closure, litter cover
PC2	Litter depth (cm), Non-pine trees, Tree richness, (-) % Bare ground, (-) % Pine shrubs	Litter depth, tree richness and density
PC3	# Fine woody debris, # Coarse woody debris, # Stumps	Downed woody debris
PC4	# Shrubs, Shrub species richness	Shrub richness and density
PC5	% Fern cover, (-) % Vine cover	% Fern cover
PC6	% Woody ground cover	% Woody ground cover

[a] Variables are positively related to the principal component unless otherwise noted.

Table 3.16. Model selection results for select herpetofauna captured at drift fence arrays at Ben's Creek WMA, LA, 2003-2004. Model subsets include models with substantial empirical support (ΔAICc $\leq$ 2) in explaining associations of herpetofauna with microhabitat. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and the best model, and w_i is Akaike weight.

Species	Model[a]	K	Deviance	AIC_c	ΔAIC_c	w_i
Amphibian occurrence	PC4 PC5	3	145.51	151.55	0	0.135
	PC4 PC5 PC6	4	144.30	152.37	0.82	0.090
	PC1 PC4 PC5 PC6	5	142.38	152.48	0.93	0.085
	PC1 PC4 PC5	4	144.56	152.63	1.08	0.079
	PC4	2	148.98	153.00	1.45	0.065
Reptile occurrence	PC1	2	149.70	153.73	0	0.107
	PC1 PC5	3	147.93	153.98	0.25	0.095
	PC1 PC6	3	148.09	154.13	0.40	0.088
	PC1 PC5 PC6	4	146.26	154.33	0.60	0.079
	PC1 PC2 PC5 PC6	5	145.03	155.13	1.41	0.053
	PC1 PC2 PC5	4	147.56	155.63	1.90	0.041
Eastern narrowmouth toad occurrence	PC4 PC6	3	97.53	103.57	0	0.060
	PC6	2	100.12	104.14	0.57	0.045
	PC4	2	100.30	104.32	0.75	0.041
	PC1 PC4 PC6	4	96.27	104.34	0.77	0.041
	PC2 PC4 PC6	4	96.36	104.43	0.86	0.039
	PC2 PC3 PC4 PC6	5	94.86	104.97	1.40	0.030
	PC3 PC6	3	98.93	104.97	1.40	0.030
	PC3 PC4 PC6	4	96.95	105.02	1.45	0.029
	PC2 PC4	3	99.04	105.09	1.51	0.028
	PC1 PC3 PC4	5	95.00	105.11	1.54	0.028
	PC2 PC4 PC5	5	95.00	105.11	1.54	0.028
	PC1 PC4 PC5	5	95.01	105.12	1.55	0.028
	PC3	2	101.39	105.41	1.84	0.024
	PC4 PC5 PC6	4	97.39	105.46	1.89	0.023

[a]PC1= canopy closure, litter; PC2= litter depth, tree richness and density; PC3= downed woody debris, PC4= shrub richness and density; PC5= fern cover; PC6= woody ground cover.

Table 3.17. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of microhabitat, for select herpetofauna detected at drift fence arrays at Ben's Creek WMA, LA, 2003-2004.

Species	Model variable[a]	Estimate[b]	Standard error	$\sum w_i$
Amphibian occurrence	PC1	0.1115	0.1777	0.389
	PC2	0.1289	0.1201	0.233
	PC3	-0.0198	0.1519	0.284
	PC4	-0.6335	0.1159	0.954
	PC5	-0.2843	0.1359	0.702
	PC6	0.1895	0.1185	0.436
Reptile occurrence	PC1	0.8142	0.1857	0.891
	PC2	-0.1460	0.1704	0.381
	PC3	0.1838	0.1564	0.180
	PC4	-0.0100	0.1183	0.263
	PC5	-0.1936	0.1163	0.514
	PC6	-0.1896	0.1312	0.508
Eastern narrowmouth toad occurrence	PC1	0.1220	0.2091	0.353
	PC2	-0.2483	0.1303	0.396
	PC3	-0.2301	0.1801	0.396
	PC4	-0.4658	0.1188	0.608
	PC5	-0.1229	0.1497	0.344
	PC6	0.3986	0.1284	0.625

[a]PC1= canopy closure, litter cover; PC2= litter depth, tree richness and density; PC3= downed woody debris, PC4= shrub richness and density, PC5= fern cover, PC6: woody ground cover.
[b] Abundance data analyzed with logistic ANOVA. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003).

Associations of Herpetofauna with Landscape Characteristics

Drift Fence Arrays

A total of 26 landscape variables (6 composition variables representing each cover class, 3 class-specific configuration variables for each of the 6 classes, distance to nearest road, and distance to nearest stream) were included in analyses of associations between occurrence of herpetofauna and landscape characteristics (Table 3.18). Based on eigenvalues >1 and a proportion of variance > 0.05, PCA resulted in 5 principal components, which accounted for 72.5% of the variance (Table 3.19).

Component 1 was interpreted to represent canopy closure (positive loading) and distance to water (negative loading; Table 3.20). Alternatively, component 2 was interpreted to represent edge density (ED) and complexity of streamside management zones (SMZ), and edge complexity of 4-6-year stands, all positive loadings. Stands which had received at least one thinning, leading to canopy release, were represented as positive loadings on component 3. On component 4, stands in which early successional communities were dominant, but without complete canopy closure, were positively loaded. Finally, median patch size (MEDPS) of SMZs, and distance to nearest road, scored positively on component 5, whereas area-weighted mean shape index (AWMSI, a measure of the complexity of stand types) of pine stands regenerated between 2001 and 2003 scored negatively. Associations between landscape characteristics and species richness of amphibians and reptiles, and occurrence of amphibians, lizards, and eastern narrowmouth toad were examined using the 5 principal components as explanatory variables in tests of all possible models (n = 31). Selection of these response variables corresponded with those selected for microhabitat models.

Table 3.18. Mean values ($\bar{X}$) and standard errors (SE) of landscape variables generated within 500 m radius circular buffer zones (*n*=48) centered on each drift fence array at Ben's Creek WMA, LA.

Variables	$\bar{X}$	SE
Landscape composition		
% Pine regenerated 2001-2003 (0-2 yr)	17.07	2.58
% Pine regenerated 1997-1999 (4-6 yr)	23.03	3.59
% Pine regenerated 1994-1996 (7-9 yr)	18.73	3.60
% Pine regenerated 1980-1992 (11-23 yr)	16.74	4.30
% Pine regenerated 1940-1979 (24-63 yr)	12.33	1.53
% Streamside management zone (hardwood)	8.90	1.21
Landscape configuration		
Median patch size, 0-2 yr (ha)	6.48	1.19
Edge density, 0-2 yr (m/ha)	0.003	0.0004
Area weighted mean shape index[a], 0-2 yr	1.35	0.14
Median patch size, 4-6 yr (ha)	7.88	2.27
Edge density, 4-6 yr (m/ha)	0.003	0.0003
Area weighted mean shape index, 4-6 yr	1.31	0.10
Median patch size, 7-9 yr (ha)	14.63	2.81
Edge density, 7-9 yr (m/ha)	0.002	0.0003
Area weighted mean shape index, 7-9 yr	0.85	0.12
Median patch size, 11-23 yr (ha)	3.49	1.02
Edge density, 11-23 yr (m/ha)	0.002	0.001
Area weighted mean shape index, 11-23 yr	0.81	0.12
Median patch size, 24-63 yr (ha)	4.05	0.77
Edge density, 24-63 yr (m/ha)	0.002	0.0003
Area weighted mean shape index, 24-63 yr	1.22	0.11
Median patch size, hardwood (ha)	3.03	0.58
Edge density, hardwood (m/ha)	0.004	0.0004
Area weighted mean shape index, hardwood	2.60	0.16
Other landscape aspects		
Distance to stream (m)	1432.69	109.82
Distance to nearest road (m)	173.65	13.28

[a] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 3.19. Principal components (PC) analysis results for landscape variables generated within 500 m radius circular buffers centered on drift fence arrays (*n*=48) at Ben's Creek WMA, LA.

	PC1	PC2	PC3	PC4	PC5
Eigenvalue:	6.30	5.90	3.77	3.47	2.29
Variance explained:	0.21	0.20	0.13	0.12	0.08
Variables:					
% Pine stands regenerated 2001-2003 (0-2 yr)	-42[a,b]	36	**-60**	-37	-22
% Pine stands regenerated 1997-1999 (4-6 yr)	-19	-17	-14	**92**	-11
% Pine stands regenerated 1994-1996 (7-9 yr)	**91**	-15	-17	-12	18
% Pine stands regenerated 1980-1992 (11-23 yr)	-15	1	**89**	-23	10
% Pine stands regenerated 1940-1979 (24-63 yr)	**-42**	-19	-34	-20	-4
% Streamside management zone (hardwood)	-21	**64**	47	-30	15
Median patch size, 0-2 yr (ha)	-27	-19	-26	11	-1
Edge density, 0-2 yr (m/ha)	-39	48	**-58**	-31	-29
Area weighted mean shape index[c], 0-2 yr	7	39	-37	15	**-58**
Median patch size, 4-6 yr (ha)	5	4	5	**64**	-9
Edge density, 4-6 yr (m/ha)	-17	8	-28	**84**	7
Area weighted mean shape index, 4-6 yr	11	**57**	-12	9	10
Median patch size, 7-9 yr (ha)	**91**	-15	-17	-12	18
Edge density, 7-9 yr (m/ha)	**96**	-3	-8	-10	8
Area weighted mean shape index, 7-9 yr	**76**	9	2	1	-6
Median patch size, 11-23 yr (ha)	-1	14	**76**	-12	-5
Edge density, 11-23 yr (m/ha)	-9	4	**91**	-20	7
Area weighted mean shape index, 11-23 yr	5	26	**68**	17	-14
Median patch size, 24-63 yr (ha)	-18	**-46**	-24	-34	27
Edge density, 24-63 yr (m/ha)	**-57**	13	-44	-15	-5
Area weighted mean shape index, 24-63 yr	**-59**	-9	-41	-18	8
Median patch size, hardwood (ha)	0	27	3	11	**80**
Edge density, hardwood (m/ha)	-15	**88**	24	-23	-1
Area weighted mean shape index, hardwood	7	**83**	-2	27	22
Distance to stream (m)	**-58**	-51	-30	39	12
Distance to nearest road (m)	11	35	9	3	**69**

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.
[c] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 3.20. Principal components (PC, eigenvalues $\geq$ 1 and variance explained >0.05) derived from landscape variables associated with drift fence arrays at Ben's Creek WMA, LA, 2003-2004. Associated variables are those with a correlation coefficient of $\geq$ |0.40| with each PC.

PC	Associated Variables[a,b]	Interpretation
PC1	7-9 yr [% composition, MEDPS, ED, AWMSI]; (-) 24-63 yr [% composition, ED, AWMSI]; (-) Distance to nearest stream	canopy closure, distance to water
PC2	SMZ [% composition, ED, AWMSI]; 4-6 yr [AWMSI]; (-) 24-64 yr [MEDPS]	edge density and shape of SMZs; shape of early successional stands
PC3	11-23 yr [% composition, MEDPS, ED, AWMSI]; (-) 0-2 yr [% composition, ED]	canopy release (thinned stands)
PC4	4-6 yr [% composition, MEDPS, ED]	early successional plant communities with open canopy
PC5	SMZ [MEDPS]; Distance to nearest road; (-) 0-2 yr [AWMSI]	SMZ size; distance to road

[a] Variables are positively related to the principal component unless otherwise noted.
[b] SMZ: streamside management zone; MEDPS=median patch size; ED=edge density; AWMSI=area weighted mean shape index.

The best approximating model in explaining species richness of amphibians relative to landscape attributes retained 2 component variables, canopy closure and distance to water (component 1), and edge density and shape of SMZs and shape of early successional stands (component 2; Table 3.21). Component 1 negatively influenced species richness of amphibians, whereas component 2 had a positive influence. Relative importance of component 1 and presence of component 1 in each of the models with substantial empirical support indicated that canopy closure and distance to water may have been the more important aspect of landscape associated with species richness of amphibians.

Among the 6 landscape models which gained substantial empirical support in explaining species richness of reptiles, the model with canopy closure and distance to water (component 1) and size of SMZ and distance to nearest road (component 5) was the best approximating model.

Component 1 was retained in 3 of the 6 candidate models, and negatively influenced species richness of reptiles. Component 5 also appeared in 3 candidate models and positively influenced species richness of reptiles. Canopy closure and distance to water, and SMZ size and distance to nearest road, were more influential that other component variables on species richness of reptiles.

The best approximating model for occurrence of amphibians included stands with a released canopy (component 3) and size of SMZ and distance to nearest road (component 5), both of which negatively influenced amphibian occurrence. The influence of canopy closure and distance to water (component 1) also was important, as it was present in 6 of the 10 candidate models (Table 3.21).

As with species richness of reptiles, occurrence of lizards at drift fence arrays was best explained by the model which retained canopy closure and distance to water (component 1) and size of SMZ and distance to nearest road (component 5). Canopy closure negatively influenced occurrence of lizards, whereas size of SMZ and distance to nearest road had a positive influence. The best approximating model in explaining occurrence of eastern narrowmouth toad included 1 component, which represented stands with a released canopy (component 3). This component variable was present in all 3 models with substantial empirical support, and negatively influenced occurrence of eastern narrowmouth toads. The importance of stands with a released canopy was greater than all other component variables (Table 3.22).

Anuran Calling Surveys

A total of 26 landscape variables (6 composition variables representing each cover class, 3 class-specific configuration variables for each of the 6 classes) were included in analyses of associations of detections of anurans during calling surveys with landscape characteristics (Table 3.23). Based on eigenvalues >1 and a minimum proportion of variance of 0.05, PCA resulted in

Table 3.21. Model selection results for select herpetofauna detected at drift fence arrays at Ben's Creek, WMA, LA, 2003-2004. Model subsets include models with substantial empirical support (ΔAICc $\leq$ 2) in explaining associations of herpetofauna detections with landscape attributes. K is number of model parameters, AICc equals Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and the best model, and w_i is Akaike weight.

Species	Model[a]	K	Deviance	AIC_c	ΔAIC_c	w_i
Amphibian species richness	PC1 PC4	3	89.54	95.59	0	0.207
	PC1	2	91.67	95.69	0.10	0.197
	PC1 PC5	3	91.50	97.55	1.95	0.078
Reptile species richness	PC1 PC5	3	94.94	100.99	0	0.139
	PC5	2	97.43	101.45	0.46	0.111
	PC2	2	97.50	101.52	0.53	0.107
	PC1	2	97.60	101.63	0.63	0.101
	PC4	2	98.67	102.70	1.70	0.059
	PC1 PC2 PC5	4	94.70	102.78	1.79	0.057
Amphibian occurrence	PC3 PC5	3	150.54	156.59	0	0.097
	PC5	2	152.86	156.88	0.29	0.084
	PC1	2	152.93	156.95	0.36	0.081
	PC1 PC5	3	151.22	157.27	0.68	0.069
	PC1 PC3	3	151.38	157.43	0.84	0.064
	PC1 PC3 PC5	4	149.70	157.78	1.19	0.054
	PC3	2	153.82	157.84	1.25	0.052
	PC3 PC4 PC5	4	150.11	158.19	1.60	0.044
	PC1 PC3 PC4	4	150.11	158.20	1.60	0.044
	PC1 PC4	3	152.28	158.33	1.74	0.041
Lizard occurrence	PC1 PC5	3	155.44	161.49	0	0.155
	PC5	2	158.02	162.05	0.56	0.114
	PC1	2	158.78	162.81	1.32	0.080
	PC2	2	159.07	163.10	1.61	0.069
	PC1 PC4 PC5	4	155.17	163.25	1.77	0.064
	PC1 PC2 PC5	4	155.24	163.32	1.83	0.062
Eastern narrowmouth toad	PC3	2	96.85	100.87	0	0.195
	PC3 PC5	3	96.44	102.49	1.61	0.087
	PC2 PC3	3	96.48	102.53	1.65	0.085

[a]PC1= canopy closure, distance to water; PC2= edge density and shape of SMZs, shape of early successional stands; PC3= stands with a released canopy; PC4=early successional plant communities with an open canopy; PC5= SMZ size; distance to road.

Table 3.22. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of landscape attributes, for select herpetofauna detected at drift fence arrays at Ben's Creek WMA, LA, 2003-2004.

Species	Model variable[a]	Estimate[b]	Standard error	$\sum w_i$
Amphibian species richness	PC1	-0.6828	0.2092	0.833
	PC2	0.1023	0.1600	0.249
	PC3	-0.5710	0.2066	0.174
	PC4	-0.4809	0.1672	0.453
	PC5	-0.3129	0.0933	0.340
Reptile species richness	PC1	-0.5236	0.1959	0.503
	PC2	0.0783	0.1508	0.371
	PC3	-0.2800	0.1691	0.190
	PC4	-0.2408	0.1668	0.248
	PC5	0.3894	0.1082	0.544
Amphibian occurrence	PC1	-0.2571	0.2480	0.526
	PC2	0.1606	0.1932	0.246
	PC3	-1.0719	0.3299	0.549
	PC4	-0.4019	0.1887	0.338
	PC5	-0.3231	0.1364	0.562
Lizard occurrence	PC1	-0.5305	0.2099	0.543
	PC2	0.0260	0.1820	0.325
	PC3	-0.3940	0.1958	0.218
	PC4	-0.3631	0.1917	0.290
	PC5	0.5433	0.1354	0.644
Eastern narrowmouth toad occurrence	PC1	-0.0983	0.2833	0.310
	PC2	-0.3613	0.2497	0.394
	PC3	-1.7431	0.3747	0.771
	PC4	0.1150	0.2337	0.294
	PC5	0.1983	0.1419	0.351

[a]PC1= canopy closure, distance to water; PC2= edge density and shape of SMZs, shape of early successional stands; PC3= stands with a released canopy; PC4=early successional plant communities with an open canopy; PC5= SMZ size; distance to nearest road.
[b] Abundance data analyzed with logistic ANOVA. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003).

5 principal components, which accounted for 78.5% of the variance (Table 3.24).

All landscape attributes of 24-63-year stands, and composition and increasing amounts of clearcut edge were positively loaded on component 1, whereas distance to nearest road and stands with a released canopy were negatively loaded. Therefore, component 1 was interpreted to represent stands with a pine-dominated overstory, and dense, woody understory, and edge density of clearcuts. Component 2 represented stands with full canopy closure (positive loadings). Edge density and shape of SMZs, and shape of clearcuts, scored positively on component 3, whereas size of 24-63-year stands and distance to nearest stream scored negatively. Variables on component 4 included composition, size and edge density of early successional stands, and size of clearcuts, all of which were positively loaded, and composition of SMZs, which was negatively loaded. Finally, component 5 represented size of SMZs and size of stands with a released canopy (positive loadings), and shape of early successional stands (negative loadings; Table 3.25).

Five models, each of which represented 1 of the 5 component variables, gained substantial empirical support in explaining variation in species richness of anurans (Table 3.26). However, model-averaged parameter estimates were close to 0 for each of these component variables and thus may not be very important. The best approximating model in explaining abundance of spring peeper retained 1 component, edge density and shape of SMZs, shape of clearcuts; and distance to water (component 3). This component was present in 3 of the 7 models which gained substantial empirical support. However, relative Akaike weight of component 3 was only slightly greater than weights of other component (Table 3.26).

Table 3.23. Mean values ($\bar{X}$) and standard errors (SE) of landscape variables generated within 500 m radius circular buffer zones (*n*=16) centered on anuran calling survey stations at Ben's Creek WMA, LA.

Variables	$\bar{X}$	SE
Landscape composition		
% Pine regenerated 2001-2003 (0-2 yr)	17.27	4.42
% Pine regenerated 1997-1999 (4-6 yr)	22.59	6.34
% Pine regenerated 1994-1996 (7-9 yr)	19.43	6.59
% Pine regenerated 1980-1992 (11-23 yr)	16.82	7.73
% Pine regenerated 1940-1979 (24-63 yr)	12.41	2.52
% Streamside management zone (hardwood)	8.42	2.00
Landscape configuration		
Median patch size, 0-2 yr (ha)	6.44	1.99
Edge density, 0-2 yr (m/ha)	0.003	0.001
Area weighted mean shape index[a], 0-2 yr	1.44	0.24
Median patch size, 4-6 yr (ha)	9.22	4.37
Edge density, 4-6 yr (m/ha)	0.003	0.001
Area weighted mean shape index, 4-6 yr	1.38	0.17
Median patch size, 7-9 yr (ha)	15.19	5.15
Edge density, 7-9 yr (m/ha)	0.002	0.001
Area weighted mean shape index, 7-9 yr	0.71	0.19
Median patch size, 11-23 yr (ha)	3.39	1.84
Edge density, 11-23 yr (m/ha)	0.002	0.001
Area weighted mean shape index, 11-23 yr	0.82	0.22
Median patch size, 24-63 yr (ha)	4.73	1.50
Edge density, 24-63 yr (m/ha)	0.002	0.000
Area weighted mean shape index, 24-63 yr	1.13	0.19
Median patch size, hardwood (ha)	3.39	1.17
Edge density, hardwood (m/ha)	0.004	0.001
Area weighted mean shape index, hardwood	2.67	0.28
Other landscape aspects		
Distance to stream (m)	1431.95	192.11
Distance to nearest road (m)	180.34	20.44

[a] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 3.24. Principal components (PC) analysis results for landscape variables generated within 500 m radius circular buffers (*n*=16) centered on anuran calling survey stations in each stand replicate at Ben's Creek WMA, LA.

	PC1	PC2	PC3	PC4	PC5
Eigenvalue:	6.73	6.38	4.07	3.28	2.95
Variance explained:	0.22	0.21	0.14	0.11	0.10
Variables:					
% Pine stands regenerated 2001-2003 (0-2 yr)	**68**[a,b]	-19	41	1	-30
% Pine stands regenerated 1997-1999 (4-6 yr)	0	-10	-9	**90**	-20
% Pine stands regenerated 1994-1996 (7-9 yr)	-5	**97**	-15	-3	1
% Pine stands regenerated 1980-1992 (11-23 yr)	**-60**	-45	-7	-50	29
% Pine stands regenerated 1940-1979 (24-63 yr)	**87**	-16	-19	-14	14
% Streamside management zone (hardwood)	-29	-38	48	**-53**	20
Median patch size, 0-2 yr (ha)	12	-22	-12	**48**	16
Edge density, 0-2 yr (m/ha)	**69**	-15	54	-4	-26
Area weighted mean shape index[c], 0-2 yr	24	26	**65**	27	8
Median patch size, 4-6 yr (ha)	-20	3	5	**80**	2
Edge density, 4-6 yr (m/ha)	8	3	12	**81**	-31
Area weighted mean shape index, 4-6 yr	18	30	-6	5	**-67**
Median patch size, 7-9 yr (ha)	-5	**97**	-15	-3	1
Edge density, 7-9 yr (m/ha)	-19	**96**	-3	1	-3
Area weighted mean shape index, 7-9 yr	-38	**81**	0	-6	-18
Median patch size, 11-23 yr (ha)	-8	-25	14	-29	**84**
Edge density, 11-23 yr (m/ha)	**-65**	-42	-3	-49	28
Area weighted mean shape index, 11-23 yr	**-58**	-36	8	14	**41**
Median patch size, 24-63 yr (ha)	43	3	**-64**	-8	-7
Edge density, 24-63 yr (m/ha)	**87**	-29	11	-6	-9
Area weighted mean shape index, 24-63 yr	**79**	-20	-12	1	-15
Median patch size, hardwood (ha)	-10	-5	22	-7	**88**
Edge density, hardwood (m/ha)	-8	-31	**79**	-39	-4
Area weighted mean shape index, hardwood	-4	2	**80**	22	20
Distance to stream (m)	25	-41	**-55**	44	-24
Distance to nearest road (m)	**-59**	5	8	-20	-18

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.
[c] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon)

Table 3.25. Principal components (PC, eigenvalues $\geq$ 1 and variance explained >0.05) derived from landscape variables associated with anuran calling survey stations at Ben's Creek, LA, 2003-2004. Associated variables have a correlation coefficient of $\geq |0.40|$ with each PC.

PC	Associated Variables[a,b,c]	Interpretation
PC1	24-63 yr [% composition, ED, AWMSI]; 0-2 yr [% composition, ED]; (-) distance to road; (-) 11-23 yr [% composition, ED, AWMSI]	mature stands, composition and edge density of clearcuts, distance to nearest road
PC2	7-9 yr [% composition, MEDPS, ED, AWMSI]	canopy closure
PC3	SMZ [ED, AWMSI];0-2 yr [AWMSI]; (-) 24-63 yr [MEDPS]; (-) distance to nearest stream	edge and shape of SMZs; shape of clearcuts, distance to water
PC4	4-6 yr [% composition, MEDPS, ED]; 0-2 yr [MEDPS]; (-) SMZ [% composition]	composition, size, and amount of edge of early successioanl stands; size of clearcuts
PC5	SMZ [MEDPS]; 11-23 yr [MEDPS]; (-) 4-6 yr [AWMSI]	size of SMZ; size of stands with canopy release

[a] Variables are positively related to the principal component unless otherwise noted.
[b] SMZ: streamside management zone; MEDPS=median patch size; ED=edge density; AWMSI=area weighted mean patch shape.

Table 3.26. Model selection results for select anurans detected during anuran calling surveys at Ben's Creek, WMA, LA, 2003-2004. Subsets include models with substantial empirical support (ΔAICc $\leq$ 2) in explaining anuran detections associated with landscape attributes. K is number of model parameters, AICc equals Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and the best model, w_i is Akaike weight.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Anuran species richness	PC2 [a]	2	16.54	20.56	0	0.155
	PC1	2	16.69	20.72	0.15	0.143
	PC4	2	16.84	20.87	0.30	0.133
	PC5	2	17.81	21.83	1.27	0.082
	PC3	2	18.21	22.24	1.67	0.067
Spring peeper abundance	PC3	2	29.70	33.73	0	0.116
	PC5	2	30.32	34.34	0.62	0.085
	PC2	2	30.40	34.43	0.70	0.081
	PC4	2	30.41	34.44	0.71	0.081
	PC1	2	30.57	34.60	0.87	0.075
	PC3 PC5	3	29.48	35.53	1.80	0.047
	PC3 PC4	3	29.52	35.57	1.85	0.046
	PC2 PC3	3	29.53	35.58	1.86	0.046

[a]PC1= pine-dominated overstory and woody understory; composition and edge of clearcuts, distance to road; PC2= canopy closure; PC3= edge and shape of SMZs, shape of clearcuts, distance to water; PC4= composition, size and edge of early successional stands, size of clearcuts; PC5= size of SMZs, size of stands with released canopy.

Table 3.27. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of landscape attributes, for select anurans detected during anuran calling surveys at Ben's Creek WMA, LA, 2003-2004.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Anuran species richness	PC1[a]	-0.0214	0.0985	0.360
	PC2	-0.0281	0.1027	0.388
	PC3	0.2901	0.0874	0.196
	PC4	0.0481	0.0985	0.347
	PC5	-0.1606	0.0998	0.241
Spring peeper abundance	PC1	-0.0127	0.1017	0.322
	PC2	0.0590	0.1010	0.342
	PC3	0.1265	0.0987	0.433
	PC4	0.0514	0.0997	0.341
	PC5	-0.0571	0.1164	0.352

[a]PC1= stands with pine-dominated overstory, and woody understory, composition and edge density of clearcuts, distance to road; PC2= canopy closure; PC3= edge density and shape of SMZs, shape of clearcuts, distance to water; PC4= composition, size and edge density of early successional stands, size of clearcuts; PC5= size of SMZs, size of stands with a released canopy.

RESULTS: SANDY HOLLOW WMA

Summary of Herpetofauna at Sandy Hollow

Between December 2002 and November 2004, we detected 32 species of herpetofauna (20 amphibians and 12 reptiles), based on the combination of all standardized survey methods and chance encounters (Table 3.28). We recorded 277 captures (83 amphibians and 194 reptiles), from drift fence arrays, cover boards and visual encounter surveys combined (detections during anuran calling surveys were indexed and thus not included in this number). Eastern narrowmouth toad (*Gastrophryne carolinensis*) was detected by every standardized survey technique. Green anole (*Anolis carolinensis*) and ground skink (*Scinella lateralis*) were detected by all survey methods designed to detect reptiles (excludes anuran calling surveys).

Drift Fence Arrays

We captured 198 individuals, 49 amphibians (3 species) and 149 reptiles (8 species), at drift fences (8,970 trap days) between April 2003 and November 2004. Eastern narrowmouth toad was the most frequently caught amphibian, constituting 73% of amphibian captures and 18% of total captures. Eastern fence lizard (*Sceloporus undulates*) and ground skink were the most common reptile captures, constituting 49% and 32% of reptile captures, and 37% and 24%, of total captures, respectively. Mean species richness and relative abundance of herpetofauna did not differ across years ($F_{1,11.2}$=0.08, P=0.786 and $F_{1,11}$=0.18, P=0.676, respectively), or seasons ($F_{1,11.2}$=0.02, P=0.878 and $F_{1,10.9}$=0.17, P=0.692, respectively).

Cover Boards

We observed 69 individuals, 32 amphibians (4 species) and 37 reptiles (8 species), at cover boards (2,476 trap days) between April 2003 and October 2004. Captures of eastern narrowmouth toad were more common than of any other amphibian, accounting for 94% of amphibian captures and 43% of total captures. Individuals of the genus *Eumeces* and ground

skink comprised most of the reptile captures, constituting 43% and 41% of reptile captures, and 23% and 22% of total captures, respectively. Mean species richness and relative abundance of herpetofauna did not differ across years ($F_{1,6.28}$=0.16, P=0.704 and $F_{1,6.46}$=0.05, P=0.837, respectively), or seasons ($F_{3,6.16}$=0.03, P=0.993 and $F_{3,6.33}$=0.01, P=0.999, respectively).

Visual Encounter Surveys

A total of 10 individuals, 2 amphibians (1 species) and 8 reptiles (3 species), were detected in the spring of 2003. Specifically, 2 eastern narrowmouth toads, 5 green anoles, 1 ground skink, and 2 racer snakes [*Coluber constrictor*]) comprised the encounters. None of the species detected was unique to this survey method. Encounters were not sufficient for statistical analysis, and were so infrequent that visual encounter surveys were not repeated at Sandy Hollow in 2004.

Anuran Calling Surveys

We detected 17 species of anurans between December 2002, and May 2004 (360 listening station surveys). Spring peeper (*Pseudacris crepitans*) and upland chorus frog (*P. triseriata*) were the most frequently detected species. Prior to analysis, winter listening surveys conducted in 2003 were removed from the data set due to 0 detections. Abundance of calling anurans was greater in 2002-2003 ($F_{1,8.7}$=12.89, P=0.006) than 2003-2004, but did not differ by season ($F_{2,49.4}$=0.71, P=0.495). Species richness of calling anurans did not differ by year ($F_{1,4.26}$=3.35, P=0.137) or season ($F_{2,2.16}$=2.33, P=0.289).

Associations of Herpetofauna with Microhabitat

None of the microhabitat variables (Table 3.29) measured at drift fence arrays that were retained for analysis were highly correlated ($\geq$ 0.8). Thus, principal components analysis (PCA) was performed with all variables included. Based on eigenvalues $\geq$ 1.0, 6 principal components were retained, accounting for 73% of the variance (Table 3.30). Component 1 was interpreted

Table 3.28. Herpetofauna captures/detections using drift fence arrays (PFFT), cover boards (CB), visual encounters (VES), anuran calling surveys (ACS) and chance encounters (denoted by X) between December 2002 – November 2004 at Sandy Hollow WMA, LA.

	Herpetofauna Survey Method			
Species	PFFT[a]	CB[a]	VES[b]	ACS[c]
Effort (# trap days)	8970	2476	36	360
Southern toad (*Bufo terrestris*)	5	0	X	1
Gulf coast toad (*B. valiceps*)	8	1	X	3
Woodhouse's toad (*B. woodhousii*)	0	0	X	0
Eastern narrowmouth toad (*Gastrophryne carolinensis*)	36	30	2	1
Spadefoot toad (*Scaphiopus holbrookii*)	0	0	0	1
Northern cricket frog (*Acris creptians*)	0	0	0	3
Southern cricket frog (*A. gryllus*)	0	0	0	3
Bird-voiced treefrog (*Hyla avivoca*)	0	0	0	2
Cope's gray treefrog (*H. chrysoscelis*)	0	0	0	2
Green treefrog (*H. cinerea*)	0	0	0	3
Barking treefrog (*H. gratiosa*)	0	0	0	3
Squirrel treefrog (*H. squirella*)	0	0	X	1
Common gray treefrog (*H. versicolor*)	0	0	0	1
Spring treefrog (*Pseudacris creptians*)	0	1	X	3
Upland chorus frog (*P. triseriata*)	0	0	X	2
Dusky gopher frog (*Rana capito sevosa*)	0	0	X	0
Bullfrog (*R. catesbeiana*)	0	0	0	1
Bronze frog (*R. clamatans*)	0	0	X	1
Southern leopard frog (*R. utricularia*)	0	0	X	1
Slimy salamander (*Plethodon glutinosus*)	0	1	X	0
Total amphibians	49	33	2	0
Green anole (*Anolis carolinensis*)	8	1	5	NA
Five-lined skink (*Eumeces fasciatus*)	1	5	0	NA
Southeastern five-lined skink (*E. inexpectatus*)	9	5	0	NA
Broadhead skink (*E. laticeps*)	2	4	0	NA
Unknown *Eumeces* spp.	6	0	0	NA
Ground skink (*Scinella lateralis*)	47	15	1	NA
Eastern fence lizard (*Sceloporus undulates*)	73	3	X	NA
Racer snake (*Coluber constrictor*)	0	1	2	NA
Eastern hognose snake (*Heterodon platirhinos*)	1	0	X	NA
Speckled kingsnake (*Lampropeltis getula holbrooki*)	0	0	X	NA
Eastern coachwhip (*Masticophis flagellum)*	0	0	X	NA
Gopher tortoise (*Gopherus polyphemus*)	0	0	0	NA
Eastern box turtle (*Terrapene Carolina)*	2	1	X	NA
Total reptiles	149	35	8	NA
Total captures	198	69	10	NA
Species Richness	12	12	5	17

[a] Effort = [# captures/# functional traps] x 20 per array per survey period per year.
[b] Effort = total number of surveys conducted during study period.
[c] Highest index value (0-3) assigned: 1=individuals, 2=calls overlapping, 3=full chorus;

Table 3.29. Mean values ($\bar{X}$) and standard errors (SE) of microhabitat variables measured at drift fence arrays in stands burned at least once every 2 years at Sandy Hollow WMA, LA.

Microhabitat	$\bar{X}$ [a]	SE
% Deciduous leaf litter	4.43	0.53
% Conifer litter	16.32	1.11
% Bare ground	7.24	0.78
% Rock [b]	0.21	0.13
% Water [b]	0	0
% Moss [b]	0.55	0.22
% Herbaceous cover	52.98	1.80
% Woody cover	13.08	1.31
% Vine cover	2.18	0.51
% Fern cover	2.64	0.90
% Canopy closure	46.91	1.64
Vegetation height (cm)	30.06	1.06
Litter depth (cm)	0.83	0.07
# Downed woody debris	9.88	1.31
# Trees ($\geq$ 8.0 cm)	4.96	0.51
Tree species richness	1.39	0.12
# Shrubs (<8.0cm dbh)	48.26	4.50
Shrub species richness	8.50	0.36

[a] $\bar{X}$ (% Deciduous leaf litter- % Canopy) = average of 4 measurements per array, season, year.
$\bar{X}$ (Vegetation height, Litter depth) = average of 16 measurements per array, season, year.
$\bar{X}$ (# Downed woody debris – Shrub species richness) = value per array, per year.
[b] Insufficient data for analysis.

to represent a gradient from areas greater in herbaceous vegetation height and percentage herbaceous ground cover (positive loadings), to areas with greater percentage woody ground cover (negative loading; Table 3.30). Percentage leaf litter and vine cover, and tree species richness each loaded positively on component 2. Component 3 was interpreted to represent areas with many trees, downed woody debris, percentage conifer litter, and a closed canopy. Shrub species richness and density loaded positively on component 4. Component 5 was interpreted as a gradient from percentage bare ground (negative loading) to litter depth (positive loading). Finally, fern cover positively loaded on component 6.

Associations between microhabitat and occurrence of anurans, lizards, eastern narrowmouth toad, ground skink, and eastern fence lizard captured at drift fence arrays were

examined using the 6 principal components as explanatory variables in tests of all possible models (n = 63). Seven models received substantial empirical support in explaining occurrence of anurans in relation to microhabitat characteristics. Five of the 6 components were present at least once among the 7 candidate models. However, the best approximating model retained 2 components, shrub richness and density (component 4), and percentage fern cover (component 6), which negatively influenced occurrence of anurans (Table 3.33).

Occurrence of lizards was best explained by the model with only shrub richness and density (component 4). In contrast to the influence of shrub richness and density on anurans, occurrence of lizards increased with increasing shrub richness and density. Although every component was present among candidate models with substantial empirical support, the best approximating model for occurrence of eastern narrowmouth toad included percentage herbaceous cover and height (component 1), shrub richness and density (component 4), and litter depth (component 5). Both percentage herbaceous cover and height and litter depth positively influenced occurrence of eastern narrowmouth toad, whereas shrub richness and density had a negative influence.

In contrast, occurrence of ground skink was negatively influenced by percentage herbaceous cover and height (component 1), the only component variable retained in the best approximating model. Ten models with varying combinations of tree density and downed woody debris (component 3), shrub richness and density (component 4), litter depth (component 5), and percentage fern cover (component 6), gained substantial empirical support in explaining occurrence of eastern fence lizards. Occurrence of eastern fence lizard increased with increasing tree density and downed woody debris, and shrub richness and density, and decreasing litter depth and percentage fern cover. However, Akaike weights small for all variables, suggesting that none of these variables may strongly influence occurrence of this species (Table 3.33).

Table 3.30. Principal components (PC) analysis results for microhabitat variables measured at drift fence arrays (n=48) at Sandy Hollow WMA, LA.

	PC1	PC2	PC3	PC4	PC5	PC6
Eigenvalue:	3.70	2.02	1.66	1.36	1.14	1.03
Variance explained:	0.25	0.13	0.11	0.09	0.08	0.07
Variables:						
% Leaf litter	2[a,b]	**74**	5	4	12	12
% Other litter	-50	-23	**54**	3	43	2
% Herbaceous cover	**77**	-20	-27	-9	0	-43
% Woody cover (2004 only)	**-52**	21	1	26	-11	-35
% Vine cover (2004 only)	-29	**64**	-20	12	7	-14
% Fern cover	0	5	-10	-4	-1	**93**
% Bare ground	-40	-19	-9	-6	**-72**	19
% Canopy closure	-43	43	**45**	22	24	1
Vegetation height (cm)	**79**	-3	-12	17	-4	7
Litter depth (cm)	-21	0	9	1	**84**	11
# Downed woody debris	-10	0	**85**	19	-6	-6
# Trees	-10	20	**80**	-15	21	-5
Tree species richness	-3	**79**	26	9	-10	1
# Shrubs	-1	-4	2	**90**	1	4
Shrub species richness	2	36	9	**76**	6	-14

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted as a better representation of variance explained.
[c] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 3.31. Principal components (PC, eigenvalues $\geq$ 1.0) derived from microhabitat variables measured at drift fence arrays at Sandy Hollow WMA, LA. Associated variables are those with a correlation coefficient of $\geq$ |0.40| with each respective PC.

PC	Loaded Variables[a]	Interpretation
PC1	% Herbaceous cover, Vegetation height (cm), (-) % Woody cover,	Herbaceous cover and height
PC2	% Leaf litter, % Vine cover, Tree species richness	Tree richness, vine, litter cover
PC3	% Conifer litter, % Canopy closure, # Downed woody debris, # Trees	Number of trees, downed woody debris
PC4	# Shrubs, Shrub species richness	Shrub richness and density
PC5	Litter depth (cm), (-) % Bare ground	Litter depth
PC6	% Fern cover	Fern cover

[a] Variables are positively related to the principal component unless otherwise noted.

Table 3.32. Model selection results for select herpetofauna detected at drift fence arrays at Sandy Hollow, WMA, LA, 2003-2004. Subsets include models with substantial empirical support (ΔAICc $\leq$ 2) in explaining occurrence associated with microhabitat. K is number of model parameters, AICc equals Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and best model, and w_i is Akaike weight.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Anuran occurrence	PC4 PC6	3	140.44	146.50	0	0.101
	PC4	2	143.18	147.21	0.71	0.071
	PC4 PC5 PC6	4	139.34	147.44	0.94	0.063
	PC1 PC4 PC6	4	139.47	147.57	1.07	0.059
	PC2 PC4 PC6	4	139.87	147.97	1.47	0.048
	PC4 PC5	3	142.24	148.30	1.80	0.041
	PC1 PC4 PC5	4	138.22	148.36	1.86	0.040
Lizard occurrence	PC4	2	354.10	358.13	0	0.335
	PC4 PC5	3	354.05	360.11	0.37	0.125
Eastern narrowmouth toad occurrence	PC1 PC4 PC5	4	105.06	113.16	0	0.081
	PC4 PC5	3	107.42	113.48	0.33	0.069
	PC1 PC3 PC4 PC5	5	103.47	113.62	0.46	0.064
	PC4	2	110.12	114.14	0.99	0.049
	PC1 PC4	3	108.27	114.32	1.17	0.045
	PC1 PC3 PC4	4	106.66	114.75	1.59	0.037
	PC2 PC4 PC5	4	106.78	114.88	1.72	0.034
	PC1 PC2 PC4 PC5	5	104.91	115.05	1.89	0.031
	PC1 PC4 PC5 PC6	5	104.97	115.11	1.96	0.030
Ground skink occurrence	PC1	2	125.63	129.65	0	0.151
	PC1 PC5	3	125.10	131.16	1.51	0.071
	PC1 PC3	3	125.50	131.56	1.90	0.058
Eastern fence lizard occurrence	PC3	2	216.61	220.64	0	0.094
	PC6	2	216.63	220.65	0.01	0.094
	PC4	2	216.76	220.79	0.15	0.088
	PC5	2	216.77	220.79	0.15	0.088
	PC5 PC6	3	215.28	221.33	0.69	0.067
	PC3 PC6	3	215.52	221.58	0.94	0.059
	PC4 PC6	3	215.56	221.61	0.97	0.058
	PC3 PC5	3	216.33	222.39	1.75	0.039
	PC3 PC4	3	216.41	222.46	1.82	0.038
	PC4 PC5	3	216.54	222.59	1.95	0.036

[a]PC1= herbaceous cover and height; PC2= tree richness, vine and litter cover; PC3= tree density, downed woody debris; PC4=shrub richness and density; PC5= litter depth; PC6= fern cover.

Table 3.33. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of microhabitat, for select herpetofauna captured at drift fence arrays at Sandy Hollow WMA, LA, 2003-2004.

Species	Model variable [a]	Estimate [b]	Standard error	$\sum w_i$
Anuran occurrence	PC1	0.2711	0.1720	0.374
	PC2	-0.0538	0.1332	0.302
	PC3	-0.0083	0.1287	0.274
	PC4	-0.3375	0.1408	0.836
	PC5	0.1545	0.1658	0.382
	PC6	-0.2727	0.1465	0.612
Lizard occurrence	PC1	-0.0727	0.1248	0.166
	PC2	0.2642	0.1047	0.000
	PC3	0.0473	0.1111	0.239
	PC4	0.2577	0.1190	0.895
	PC5	0.0180	0.1240	0.287
	PC6	0.0625	0.1030	0.225
Eastern narrowmouth toad occurrence	PC1	0.5239	0.1815	0.586
	PC2	-0.1062	0.1442	0.307
	PC3	-0.2381	0.1452	0.405
	PC4	-0.4114	0.1525	0.787
	PC5	0.3889	0.1782	0.597
	PC6	-0.0176	0.1415	0.272
Ground skink occurrence	PC1	-0.3281	0.1650	0.669
	PC2	0.2413	0.1063	0.190
	PC3	-0.0522	0.1296	0.305
	PC4	0.0760	0.1594	0.293
	PC5	0.1907	0.1632	0.351
	PC6	0.0185	0.1379	0.278
Eastern fence lizard occurrence	PC1	-0.1726	0.1432	0.127
	PC2	0.1059	0.1230	0.128
	PC3	0.0751	0.1256	0.377
	PC4	0.0717	0.1368	0.358
	PC5	-0.0548	0.1433	0.376
	PC6	-0.1500	0.1276	0.455

[a] PC1= herbaceous cover and height; PC2= tree richness, vine and litter cover; PC3= tree density, downed woody debris; PC4=shrub richness and density; PC5= litter depth; PC6= fern cover.
[b] Abundance data analyzed with logistic ANOVA. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003).

Associations of Herpetofauna with Landscape Characteristics

Drift Fence Arrays

A total of 22 landscape variables (5 composition variables representing each cover class, 3 class-specific configuration variables for each of the 5 classes, distance to nearest road, and distance to nearest body of water [pond or stream]) were included in analyses of associations of occurrence of select herpetofauna with landscape characteristics (Table 3.34). Based on eigenvalues >1 and a proportion of variance > 0.05, PCA resulted in 5 principal components, which accounted for 90.0 % of the variance (Table 3.35).

All landscape attributes associated with openings positively loaded onto component 1, which was thus interpreted to represent openings (Table 3.36). Alternatively, percentage composition, MEDPS, and ED of pine forest patches were positive loadings on component 2. Therefore, component 2 was interpreted to represent pine forests. On component 3, strongly correlated positive loadings included each landscape characteristic associated with patches of mixed pine-hardwood forest, and patch shape of pine forests, whereas negative loadings included composition and size of patches of longleaf pine. As a result, component 3 was interpreted to represent patches of mixed pine-hardwood forest. Further, component 4 consisted of edge and shape patch characteristics of both longleaf pine and longleaf savannah (positive loadings), and distance to nearest body of water (negative loading. Finally, composition and size of patches of longleaf pine savannah, as well as distance to nearest road, all positive loadings, loaded onto component 5.

Associations between landscape and occurrence of anurans, lizards, eastern narrowmouth toad, ground skink, and eastern fence lizard were examined using the 5 principal components as explanatory variables in tests of all possible models (n = 31). The proportion of the landscape comprised of openings (component 1) was the most important landscape characteristic associated

with occurrence of lizard, which decreased with increasing proportion of landscape in openings. Similarly, the proportion of openings in the landscape also negatively influenced occurrence of ground skink, and was the only component variable of importance to occurrence of ground skink. In contrast, the best approximating model for occurrence of eastern fence lizard retained openings (component 1) and amount of edge and shape of longleaf pine and longleaf savannah (component 4), both of which negatively influenced occurrence of eastern fence lizard.

Table 3.34. Mean ($\bar{X}$) values and standard errors (SE) of landscape variables generated within 500 m radius circular buffer zones centered on drift fence arrays (*n*=36) at Sandy Hollow WMA, LA.

Variables	$\bar{X}$	SE
Landscape composition		
% Longleaf pine	26.37	1.61
% Longleaf savannah	43.17	1.36
% Mixed pine-hardwood forest	17.01	2.10
% Pine forest	7.48	1.54
% Openings	5.97	0.95
Landscape configuration		
Median patch size, longleaf pine (ha)	20.61	1.26
Edge density, longleaf pine (m/ha)	0.009	0.0004
Area weighted mean shape index[a], longleaf pine	4.51	0.13
Median patch size, longleaf savannah (ha)	33.75	1.06
Edge density, longleaf savannah (m/ha)	0.01	0.0003
Area weighted mean shape index, longleaf savannah	4.34	0.10
Median patch size, mixed pine-hardwood forest (ha)	13.29	1.64
Edge density, mixed pine-hardwood forest (m/ha)	0.004	0.0004
Area weighted mean shape index, mixed pine-hardwood forest	2.32	0.16
Median patch size, pine forest (ha)	5.85	1.21
Edge density, pine forest (m/ha)	0.002	0.0003
Area weighted mean shape index, pine forest	1.84	0.17
Median patch size, openings (ha)	4.67	0.74
Edge density, openings (m/ha)	0.002	0.0002
Area weighted mean shape index, openings	1.59	0.15
Other landscape aspects		
Distance to nearest body of water (m)	459.99	46.87
Distance to nearest road (m)	312.08	42.02

[a] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 3.35. Principal components (PC) analysis results for landscape variables generated within 500 m radius circular buffers centered on drift fence arrays (*n*=36) at Sandy Hollow WMA, LA.

	PC1	PC2	PC3	PC4	PC5
Eigenvalue:	6.30	4.43	4.30	2.67	2.10
Variance explained:	0.29	0.20	0.20	0.12	0.10
Variables:					
% Longleaf pine	-26	-53	**-60**	49	-5
% Longleaf savannah	1	10	-12	9	**95**
% Mixed pine-hardwood forest	-13	-31	**64**	-34	-58
% Pine forest	-13	**97**	-10	-4	2
% Openings	**93**	-14	-6	-17	-2
Median patch size, longleaf pine (ha)	-26	-53	**-60**	49	-5
Edge density, longleaf pine (m/ha)	-14	-35	-32	**84**	13
Area weighted mean shape index[c], longleaf pine	0	0	13	**84**	31
Median patch size, longleaf savannah (ha)	1	10	-12	9	**95**
Edge density, longleaf savannah (m/ha)	-49	-5	13	**62**	53
Area weighted mean shape index, longleaf sav.	-62	-14	16	**66**	5
Median patch size, mixed pine-hardwood (ha)	-13	-31	**64**	-34	-58
Edge density, mixed pine-hardwood (m/ha)	2	-20	**89**	-13	-34
Area weighted mean shape index, mixed p-h	13	11	**87**	26	-12
Median patch size, pine forest (ha)	-13	**97**	-10	-4	2
Edge density, pine forest (m/ha)	-2	**90**	26	-3	11
Area weighted mean shape index, pine forest	-2	50	**72**	5	7
Median patch size, openings (ha)	**93**	-14	-6	-17	-2
Edge density, openings (m/ha)	**97**	-6	11	3	1
Area weighted mean shape index, openings	**84**	9	32	5	13
Distance to nearest water (m)	1	-19	17	**-68**	38
Distance to nearest road (m)	3	-34	-24	-41	**63**

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.
[c] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Landscape attributes associated with patches of pine forest (component 2) and patch edge and shape of longleaf pine and longleaf savannah (component 4) were more important than other component variables in explaining occurrence of anurans. Frequency of occurrence of anurans decreased with an increase in the proportion of pine forest in the landscape, and increased with respect to patch edge and shape of longleaf pine and longleaf savannah. In contrast to occurrence of anurans, occurrence of eastern narrowmouth toad was influenced more by landscape features associated with openings (component 1), mixed pine-hardwood forests (component 3), and patch edge and shape of longleaf pine and longleaf savannah (component 4), than by other landscape component variables. Occurrence of eastern narrowmouth toad was positively associated with openings, and patch edge and shape of longleaf pine and longleaf savannah, and negatively associated with mixed pine-hardwood forests.

Table 3.36. Principal components (PC, eigenvalues $\geq$ 1) derived from landscape variables associated with drift fence arrays at Sandy Hollow WMA, LA, 2003-2004. Associated variables are those with a correlation coefficient of $\geq$ 0.40 with each respective PC.

PC	Associated Variables[a,b]	Interpretation
PC1	Openings [% composition, MEDPS, ED, AWMSI]	Openings
PC2	Pine forests [% composition, MEDPS, ED]	Pine forest
PC3	Mixed pine-hardwood [% composition, MEDPS, ED, AWMSI]; Pine forest [ASMSI]; (-) Longleaf pine [% composition, MEDPS]	Mixed pine-hardwood
PC4	Longleaf pine [ED, AWMSI]; Longleaf savannah [ED, AWMSI]; (-) Distance to water	Patch edge and shape of longleaf pine and savannah
PC5	Longleaf savannah [% composition, MEDPS]; Distance to nearest road	Patch size of longleaf savannah; distance to road

[a] Variables are positively related to the principal component unless otherwise noted
[b] SMZ: streamside management zone; MEDPS=median patch size; ED=edge density; AWMSI=area weighted mean patch shape.

Table 3.37. Model selection results for select herpetofauna captured at drift fence arrays at Sandy Hollow, WMA, 2003-2004. Subsets include models with substantial empirical support ($\Delta AIC_c \leq 2$) in explaining associations between herpetofauna detections and landscape attributes. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and best model, w_i is Akaike weight.

Species or Group	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Anuran occurrence	PC4	2	146.97	151.00	0	0.111
	PC2	2	147.10	151.12	0.12	0.104
	PC1	2	147.18	151.20	0.20	0.100
	PC3	2	147.94	151.97	0.97	0.068
	PC2 PC4	3	146.18	152.24	1.23	0.060
	PC1 PC4	3	146.32	152.38	1.38	0.056
	PC5	2	148.58	152.60	1.60	0.050
Lizard occurrence	PC1	2	355.11	359.14	0	.0.461
	PC1 PC4	3	354.63	360.69	1.55	0.212
Eastern narrowmouth toad occurrence	PC4	2	113.38	117.41	0	0.115
	PC3	2	113.79	117.82	0.41	0.094
	PC3 PC4	3	112.06	118.12	0.71	0.081
	PC1	2	114.13	118.15	0.74	0.079
	PC2	2	114.85	118.88	1.47	0.055
	PC1 PC4	3	112.84	118.89	1.48	0.055
	PC1 PC3	3	113.16	119.22	1.80	0.047
Ground skink occurrence	PC1	2	127.50	131.52	0	0.187
	PC3	2	128.49	132.52	0.99	0.114
	PC2	2	129.11	133.14	1.61	0.084
Eastern fence lizard occurrence	PC1 PC4	3	208.04	214.10	0	0.246
	PC1	2	210.62	214.65	0.56	0.186
	PC1 PC3 PC4	4	207.52	215.61	1.52	0.115
	PC1 PC3	3	209.89	215.95	1.85	0.097

[a]PC1=openings; PC2=pine forest; PC3=mixed pine-hardwood forests; PC4=Patch edge, shape of longleaf pine and savannah; PC5=Patch size of longleaf savannah; distance to nearest road.

Table 3.38. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of landscape, for select herpetofauna captured at drift fence arrays at Sandy Hollow WMA, LA, 2003-2004.

Species	Model variable [a]	Estimate [b]	Standard error	$\sum w_i$
Anuran occurrence	PC1	0.1664	0.1741	0.387
	PC2	-0.2968	0.2001	0.409
	PC3	-0.0845	0.1937	0.332
	PC4	0.1977	0.1834	0.452
	PC5	-0.1339	0.1886	0.260
Lizard occurrence	PC1	-0.3240	0.1336	0.881
	PC2	0.1717	0.1295	0.134
	PC3	-0.1711	0.1390	0.149
	PC4	0.0857	0.1339	0.352
	PC5	-0.5575	0.1405	0.009
Eastern narrowmouth toad occurrence	PC1	0.2003	0.1854	0.403
	PC2	-0.3342	0.2249	0.341
	PC3	-0.3694	0.2273	0.472
	PC4	0.3422	0.2104	0.522
	PC5	-0.2152	0.2108	0.234
Ground skink occurrence	PC1	-0.7038	0.2126	0.553
	PC2	0.0030	0.1994	0.317
	PC3	-0.2403	0.2243	0.321
	PC4	0.3743	0.2058	0.256
	PC5	-0.5217	0.2501	0.237
Eastern fence lizard occurrence	PC1	-0.2425	0.1633	0.891
	PC2	0.0899	0.1688	0.190
	PC3	0.0865	0.1805	0.344
	PC4	-0.3456	0.1765	0.584
	PC5	-0.4184	0.1746	0.104

[a]PC1=openings; PC2=pine forest; PC3=mixed pine-hardwood forests; PC4=Patch edge, shape of longleaf pine and savannah; PC5=Patch size of longleaf savannah; distance to nearest road.
[b] Abundance data analyzed with logistic ANOVA. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003).

Anuran Calling Surveys

A total of 22 landscape variables (5 composition variables representing each cover class, 3 class-specific configuration variables for each of the 5 classes, distance to nearest road, and distance to nearest body of water [pond or stream]) were included in analyses of associations of occurrence of select anurans with landscape characteristics (Table 3.39). Based on eigenvalues >1 and a proportion of variance > 0.05, PCA resulted in 5 principal components, which accounted for 92.5 % of the variance (Table 3.40).

Component one was positively loaded with landscape attributes associated with mixed pine-hardwood (Table 3.41). Alternatively, landscape characteristics associated with pine forest were positive loadings on component 2, whereas percentage composition and shape of longleaf pine were negative loadings. Thus, component 2 was interpreted to represent pine forest. All landscape characteristics of openings loaded positively onto component 3. Further, amount of edge and shape of longleaf pine and longleaf savannah were positive loadings on component 4. Finally, percent composition and size of longleaf savannah, distance to nearest body of water, and distance to nearest road, all positive loadings, were strongly correlated with component 5.

Associations between landscape characteristics and occurrence of anuran, northern and southern cricket frog, and upland chorus frog were examined using the 5 principal components as explanatory variables in tests of all possible models (n = 31). Increasing amount of edge and shape of longleaf pine and longleaf savannah (component 4) best explained variation in occurrence of anurans (Table 3.42), which was positively associated with this component. In contrast, the proportion of the landscape comprised of mixed pine-hardwood forest (component 1) was more important explaining variation in occurrence of northern crocket frog, which was negatively influenced by mixed pine-hardwood forest. Neither occurrence of southern cricket frog nor upland chorus frog was well explained by landscape characteristics.

Table 3.39. Mean ($\bar{X}$) values and standard errors (SE) of landscape variables generated within 500 m radius circular buffer zones centered on the center anuran calling survey station of each transect (*n*=12) at Sandy Hollow WMA, LA.

Variables	$\bar{X}$	SE
Landscape composition		
% Longleaf pine	26.50	3.03
% Longleaf savannah	43.11	2.38
% Mixed pine-hardwood forest	16.23	3.67
% Pine forest	7.44	2.59
% Openings	6.71	1.79
Landscape configuration		
Median patch size, longleaf pine (ha)	20.71	2.37
Edge density, longleaf pine (m/ha)	0.009	0.0008
Area weighted mean shape index[a], longleaf pine	4.55	0.24
Median patch size, longleaf savannah (ha)	33.70	1.86
Edge density, longleaf savannah (m/ha)	0.011	0.0006
Area weighted mean shape index, longleaf savannah	4.32	0.19
Median patch size, mixed pine-hardwood forest (ha)	12.69	2.87
Edge density, mixed pine-hardwood forest(m/ha)	0.004	0.0007
Area weighted mean shape index, mixed pine-hardwood forest	2.22	0.29
Median patch size, pine forest (ha)	5.82	2.03
Edge density, pine forest (m/ha)	0.002	0.0005
Area weighted mean shape index, pine forest	1.75	0.29
Median patch size, openings (ha)	5.25	1.40
Edge density, openings (m/ha)	0.002	0.0004
Area weighted mean shape index, openings	1.49	0.29
Other landscape aspects		
Distance to nearest body of water (m)	446.81	77.12
Distance to nearest road (m)	282.20	75.50

[a] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 3.40. Principal components (PC) analysis results for landscape variables generated within 500 m radius circular buffers centered on the center anuran calling survey station of each transect (*n*=12) at Sandy Hollow WMA, LA.

	PC1	PC2	PC3	PC4	PC5
Eigenvalue:	7.08	4.47	4.03	2.83	1.93
Variance explained:	0.32	0.20	0.18	0.13	0.09
Variables:					
% Longleaf pine	-58	**-61**	-24	32	-30
% Longleaf savannah	-51	14	-15	26	**75**
% Mixed pine-hardwood forest	**94**	-18	0	-18	-19
% Pine forest	-24	**93**	-22	-14	-3
% Openings	8	-12	**92**	-31	-5
Median patch size, longleaf pine (ha)	-58	**-61**	-24	32	-30
Edge density, longleaf pine (m/ha)	-44	-39	-11	**76**	-24
Area weighted mean shape index[c], longleaf pine	-6	2	4	**95**	0
Median patch size, longleaf savannah (ha)	-51	14	-15	25	**75**
Edge density, longleaf savannah (m/ha)	-13	7	-46	**82**	25
Area weighted mean shape index, longleaf sav.	10	-2	-48	**83**	-12
Median patch size, mixed pine-hardwood (ha)	**94**	-18	0	-18	-19
Edge density, mixed pine-hardwood (m/ha)	**96**	-2	17	7	-11
Area weighted mean shape index, mixed p-h	**72**	27	27	44	-14
Median patch size, pine forest (ha)	-24	**93**	-22	-14	-3
Edge density, pine forest (m/ha)	-4	**94**	10	14	2
Area weighted mean shape index, pine forest	29	**73**	35	36	-17
Median patch size, openings (ha)	8	-12	**92**	-31	-5
Edge density, openings (m/ha)	11	1	**98**	-6	2
Area weighted mean shape index, openings	14	20	**90**	12	20
Distance to nearest water (m)	18	2	16	-27	**66**
Distance to nearest road (m)	-28	-38	13	-11	**69**

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.
[c] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 3.41. Principal components (PC, eigenvalues ≥ 1) derived from landscape variables associated with anuran calling survey stations at Sandy Hollow WMA, LA, 2003-2004. Associated variables are those with a correlation coefficient of $\geq |0.40|$ with each respective PC.

PC	Associated Variables[a,b]	Interpretation
PC1	Mixed pine-hardwood [% composition, MEDPS, ED, AWMSI]	Mixed pine-hardwood forest
PC2	Pine forest [% composition, MEDPS, ED, AWMSI], (-) Longleaf pine [% composition, MEDPS]	Pine forest
PC3	Openings [% composition, MEDPS, ED, AWMSI]	Openings
PC4	Longleaf pine [ED, AWMSI], Longleaf savannah [ED, AWMSI]	Edge and shape of longleaf pine and longleaf savannah
PC5	Longleaf savannah [% composition, MEDPS], Distance to water, Distance to road	Size of longleaf savannah, distance to water and road

[a] Variables are positively related to the principal component unless otherwise noted
[b] SMZ: streamside management zone; MEDPS=median patch size; ED=edge density; AWMSI=area weighted mean patch shape.

Table 3.42. Model selection results for select anurans detected during anuran calling surveys at Sandy Hollow, WMA, 2003-2004. Subsets include models with substantial empirical support ($\Delta AIC_c \leq 2$) in explaining associations between herpetofauna detections and landscape attributes. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and best model, w_i is Akaike weight.

Species or Group	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Anuran occurrence	PC1	2	44.46	48.67	0	0.136
	PC2	2	44.50	48.71	0.05	0.132
	PC5	2	44.84	49.05	0.38	0.112
	PC4	2	45.15	49.36	0.70	0.096
	PC2 PC4	3	43.21	49.63	0.97	0.084
	PC2 PC4	3	43.29	49.72	1.05	0.080
	PC4 PC5	3	43.54	49.97	1.30	0.071
Northern cricket frog occurrence	PC1	2	21.14	25.35	0	0.184
	PC5	2	21.98	26.19	0.84	0.121
	PC3	2	22.04	26.25	0.90	0.117
	PC2	2	22.06	26.27	0.93	0.116
Southern cricket frog occurrence	PC2	2	18.72	22.93	0	0.116
	PC5	2	18.97	23.18	0.25	0.103
	PC4	2	19.02	23.23	0.30	0.100
	PC3	2	19.03	23.24	0.31	0.099
	PC1	2	19.07	23.28	0.35	0.097
Upland chorus frog occurrence	PC4	2	10.59	14.80	0	0.107
	PC2	2	10.60	14.82	0.01	0.106
	PC5	2	10.64	14.85	0.04	0.104
	PC1	2	10.64	14.85	0.05	0.104
	PC3	2	10.69	14.90	0.10	0.101

[a]PC1=Mixed pine-hardwood forest; PC2=Pine forest; PC3=Openings; PC4= Patch edge and shape of longleaf pine and longleaf savannah; PC5=Patch size of longleaf savannah; distance to nearest body of water and nearest road.

Table 3.43. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of landscape, for select anurans detected during anuran calling surveys at Sandy Hollow WMA, LA, 2003-2004.

Species	Model variable [a]	Estimate [b]	Standard error	$\sum w_i$
Anuran occurrence	PC1	0.1791	0.4205	0.367
	PC2	0.1537	0.4135	0.373
	PC3	0.7576	0.3480	0.098
	PC4	0.5002	0.4401	0.429
	PC5	0.3136	0.4075	0.314
Northern cricket frog occurrence	PC1	-1.7633	1.0589	0.519
	PC2	0.2537	0.6131	0.318
	PC3	0.1126	0.7910	0.331
	PC4	1.8332	0.8874	0.094
	PC5	-0.0378	0.8114	0.335
Southern cricket frog occurrence	PC1	-0.8122	0.9326	0.307
	PC2	1.8201	0.7924	0.355
	PC3	0.2335	0.8279	0.320
	PC4	1.0178	0.7480	0.318
	PC5	-0.6759	0.8970	0.328
Upland chorus frog occurrence	PC1	-0.0551	1.0050	0.323
	PC2	1.2153	0.9921	0.330
	PC3	3.0095	0.9570	0.320
	PC4	1.2968	1.0887	0.331
	PC5	0.0898	1.0201	0.323

[a]PC1=Mixed pine-hardwood forest; PC2=Pine forest; PC3=Openings; PC4= Patch edge and shape of longleaf pine and longleaf savannah; PC5=Patch size of longleaf savannah; distance to nearest body of water and nearest road.
[b] Abundance data analyzed with logistic ANOVA. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003).

DISCUSSION

Sampling Methods

Although drift fences with pitfalls or funnel traps, or a combination of both, appears to be the most common method used to survey amphibians and reptiles (Renken et al. 2004, Dodd and Cade 1998, Gibbs 1998, Enge 1998, Adams et al. 1996, McLeod and Gates 1998), a number of researchers have combined this technique with visual encounters, cover boards, and anuran calling surveys (i.e. Block and Morrison 1998, Ryan et al. 2002, Cromer et al 2002, Kolozsvary

and Swihart 1999, Nuzzo and Mierzwa 2000). Indeed, captures in pitfalls and funnel traps associated with our drift fences comprised the greatest proportion of individual detections across survey methods and study areas, which is consistent with other studies incorporating multiple methods (Ryan et al. 2002, Kolozsvary and Swihart 1999). Detections of a number of species, including eastern narrowmouth toad, *Rana* spp., ground skink, and green anole occurred with multiple survey methods, but were greatest using drift fences. In our case, 1 and 2 species were unique to drift fences at Sandy Hollow and Sherburne, respectively, and 5 species were unique to drift fences at Ben's Creek. Captures at drift fences accounted for 63% of species detected at Sherburne, 50% at Ben's Creek, and 34% at Sandy Hollow. It is important to note, however, that total captures with drift fences in other studies of similar length far exceed the present study, ranging from 2 to 19 times as many captures (McLeod and Gates 1998, Mitchell et al. 1997, Aubry 2000, Russell et al. 2002). Two factors, including continuous trapping over a longer period of time, and association of trap schedule with rain events, perhaps explain the difference in capture rate. Implementation of either running traps in close association with rain events or keeping traps open for a longer period of time may have resulted in higher rates of capture in the present study.

Contribution to species richness and number of detections by other methods was variable by study area. Relative to other capture methods (excluding anuran calling surveys), cover boards were more effective at Sandy Hollow, accounting for 25% of total captures, than Ben's Creek (11%) or Sherburne (6%). However, total number of captures at Sandy Hollow (n=69) and Sherburne (n=67) were similar and exceeded captures at Ben's Creek (n=20). This does not take into account trapping effort, which was greatest at Ben's Creek (3,312 trap days) and least at Sandy Hollow (2,476 trap days). Surveys of cover boards at Sandy Hollow and Sherburne each contributed 1 species to total species richness, whereas no species was unique to cover board

surveys at Ben's Creek. Our success with cover boards is similar to other studies (Ryan et al. 2002, Kolozsvary and Swihart 1999). After establishment of cover boards, approximately 6 months passed before we began to record use of cover boards by herpetofauna. Additionally, a large proportion of cover boards either partially or completely deteriorated prior to the end of the study, presumably due to weather damage, decomposition by termites, or both. Low detection rates and contribution to species richness, as well as decomposition prior to completion of the study, likely made the cost and effort of producing and distributing cover boards the least worthwhile technique in our study.

Visual encounter surveys at Sandy Hollow and Ben's Creek resulted in few individual detections (*n*=10 and *n*=19, respectively), and detection of 4 and 6 species, respectively, none of which were unique to this survey method. The cost in terms of person-hours was not worth the result, which has been the conclusion of other researchers (Ryan et al. 2002) and thus, visual encounter surveys were not repeated at these 2 study areas in 2004. In contrast, 44% of detections at Sherburne occurred during visual encounter surveys. No species was unique to this technique at this study area either, although 19 species were detected. Approximately half (48%) of these detections were of spring peeper and 38% of *Rana* spp., both of which were detected by all other methods. Logistical constraints prevented visual encounter surveys during fall, which likely would have increased both species richness and number of detections using this survey technique.

At all study sites, anuran calling surveys detected more species of anuran than all other methods. Among survey methods, anuran calling surveys made the largest contribution to species richness at Sandy Hollow, where 13 species were unique to this monitoring technique, compared to Ben's Creek (9 unique species) and Sherburne (3 unique species). Most unique species were members of the Hylidae family, including Cope's gray treefrog, squirrel treefrog,

and barking treefrog. This is not surprising, as members of the Hylidae family either are arboreal (Conant and Collins 1998) or have the ability to climb or jump out of pitfall traps (LeGrand, personal obs., deMaynadier and Hunter 1998). However, a number of these species either were heard during one survey period (i.e. Woodhouse's toad and bullfrog, summer 2003, Ben's Creek), at one stand replicate (bird-voiced treefrog, Ben's Creek), or detected as a single individual calling (eastern spadefoot, June 2004, Sandy Hollow), which limited inferences about relationships to forest management or habitat. It is quite possible that increasing survey frequency (2-3 surveys per season instead of 1) may have made this method more effective (Stevens et al. 2002).

Although not a standardized survey, chance encounters (aural and visual) of amphibians and reptiles contributed to the total number of species detected at each study area. For example, although captures at drift fences accounted for detections of 2 and 5 species of snake at Sandy Hollow and Sherburne, respectively, detections of 9 species of snake were strictly as chance encounters. Additionally, detection at Ben's Creek and Sandy Hollow of gopher tortoise, an endangered species (Louisiana Natural Heritage Program 2005), was limited to chance encounters. Because the likelihood of detecting certain species of amphibian and reptile differs by survey technique, using a combination of methods increased the probability of detecting more individuals and more species within the herpetofaunal community (Block and Morrison 1998). Based on our results, drift fences with pitfalls and funnel traps proved to be relatively effective in monitoring herpetofauna, and perhaps would be the most efficient method when use of multiple techniques is not feasible.

Landscape-level Associations

Clearcuts, riparian buffers, forest area or cover, and distances to nearest road and body of water have influenced occurrence of herpetofauna to varying degrees in previous studies

(Renken et. al 2004, Vesley and McComb 2002, Hecnar and M'Closkey 1998, Gibbs 1998, Petranka et al. 2003). In Missouri, abundance of spotted salamander (*Ambystoma maculatum*) was greater 50 m and 200 m from clearcuts than 0 m from clearcuts, whereas abundance of bronze frog was greatest 50 m from clearcuts (Renken et al. 2004). In contrast, we found that occurrence of amphibians and eastern narrowmouth toad at Ben's Creek increased with increasing amounts of patch edge associated with clearcuts. As well, amphibians also were positively associated with mean shape complexity of clearcut, whereas lizards were negatively associated with mean shape complexity of clearcut.

Occurrence of 10 species of amphibian was positively associated with riparian buffer zones on a conifer plantation in Oregon (Vesley and McComb 2002). Similarly, streambeds in oak forests in Connecticut positively influenced several amphibian species. Although dry most of the year, streambeds were characterized by a moister microclimate, wetter substrates and more adjacent cover than adjacent managed uplands, and thus, provided movement paths for amphibians (Gibbs 1998). Notably, SMZs at Ben's Creek negatively influenced occurrence of amphibians at drift fences, whereas occurrence of lizards at drift fences increased with increasing proportion of SMZs in the landscape. However, detection of spring peeper during calling surveys was positively associated with amount of edge of SMZs relative to the landscape and mean shape complexity of SMZs, as well as mean shape complexity of clearcuts. This contrasts with studies in other locations, where spring peepers were ubiquitous and thus not associated with any landscape variables (Stevens et al. 2002, Guerry and Hunter 2002). It is possible that detections may have been too infrequent, perhaps due to an inappropriate survey schedule (see **Sampling Methods** above), to make inferences associated with landscape in this study (Stevens et al. 2002).

Amount of regional woodlands was the single most important variable, among both microhabitat and landscape variables, associated with species richness of amphibians in Ontario (Hecnar and M'Closkey 1998). Similarly, area of forest was among landscape variables important to several species of amphibian in a mosaic of agriculture and patches of forest in Indiana (Kolozsvary and Swihart 1999). Furthermore, reproductive success of gray treefrog and spring peeper in agricultural ponds in Minnesota was positively associated with total area of forest within 1000 m and 2500 m, respectively, of the pond center (Knutson et al. 2004). Across both Ben's Creek and Sandy Hollow, associations between occurrence and forest area were fairly consistent with this trend. Among other landscape variables, occurrence of amphibians at Ben's Creek was positively associated with proportion of pine stands 24-63 years old, the oldest forest age class of loblolly pine at this study area. At Sandy Hollow, overall occurrence of anurans was negatively associated with area of pine forest, and positively associated with amount of edge of longleaf pine relative to the landscape and median shape complexity of longleaf pine and longleaf savannah. In contrast, occurrence of eastern narrowmouth toad was positively associated with all landscape attributes (amount of edge relative to the landscape, shape complexity, patch size, proportion of landscape in this habitat type) of longleaf pine, longleaf savannah, and mixed pine-hardwood forest.

Openings (ponds, residential areas, agricultural fields) in the landscape at Sandy Hollow were important predictors of occurrence for eastern narrowmouth toad, lizards, ground skink, and eastern fence lizard. Eastern narrowmouth toad occurred on every study area, and in every treatment or stand type, and thus, a positive association with this landscape feature in the landscape was not unexpected. Alternatively, presence of openings was the only determinant at the landscape scale of occurrence for both lizards and ground skink, which were negatively influenced by openings. Over 50% of lizards captured in drift fences at Sandy Hollow were

skinks, woodland species typically associated with moisture (Conant and Collins1998), so negative relationship with openings was not unexpected.

Association of herpetofauna with proximity to roads has been variable (Petranka et al. 2003, Gibbs 1998), whereas proximity to water generally has been positively associated with species richness, abundance, and occurrence of herpetofauna (Nuzzo and Mierzwa 2000, Knutson et al. 2004, Stevens et al. 2002, Kolozsvary and Swihart 1999). Species richness and abundance of amphibian egg masses were determined to be independent of distance to nearest forest, paved road, or source pond in North Carolina (Petranka et al. 2003). Conversely, roads adjacent to forest stands created edge that either reduced or blocked passage of amphibians en route to breeding pools in Connecticut (Gibbs 1998). Species both at Ben’s Creek and Sandy Hollow were variably associated with distance to nearest road and distance to nearest body of water. Occurrence of amphibians at Ben’s Creek was positively associated with distance to road, whereas lizards were negatively associated. Occurrence of both groups increased with increasing distance to water. At Sandy Hollow, occurrence of anurans and eastern narrowmouth toad increased with decreasing distance to nearest body of water, whereas occurrence of eastern fence lizard increased with increasing distance to nearest body of water, trends which we expected to see, since most anurans are semi-aquatic and lizards used xeric and open habitats elsewhere (Block and Morrison 1998).

Stand-level Associations

Effects of uneven-aged management on species richness, abundance, and distribution of herpetofauna have varied (Renken et al. 2004, Messere and Ducey 2004, Adams et al. 1996, Harpole and Haas 1999, Frederickson and Frederickson 2004). Effects of uneven-aged management on abundance of amphibians were not detected in Missouri Ozark forests, although canopy closure was reduced in treatment stands relative to control stands (Renken et al. 2004).

Similarly, newly formed gaps resulting from selection cuts (group and individual) did not strongly affect abundance and distribution of salamanders in New York (Messere and Ducey 1998). Further, although percentage canopy cover was higher in uncut stands, abundance and richness of amphibians were similar across high-leave harvest, low-leave harvest, clearcut harvest, and no harvest stands in Kentucky (Adams et al. 1996).

In contrast, relative abundance of salamanders was lower on group-selection and clearcut stands than on control stands in southwestern Virginia, where canopy cover was greater on control than group-selection and clearcut stands, and leaf litter was similar across treatments, but most variable on sites with trees removed. Further, soil temperature and moisture were higher on group-selection and clearcut than control stands (Harpole and Haas 1999). Abundance and richness of amphibians was similar in individual-selection and undisturbed stands in Bolivia. Canopy cover was significantly less in disturbed than undisturbed sites, but understory cover was similar (Frederickson and Frederickson 2004).

Generally, we detected more species and individuals of amphibian in uncut and individual-selection stands than group-selection stands at Sherburne. Uncut and individual-selection stands were characterized by greater estimates of canopy closure, deciduous litter, and bare ground compared to group-selection stands. On sites with greater canopy cover, amphibians have more foraging opportunities because retention of moisture in the leaf litter continues for several days after a sufficient rain event (Harpole and Haas 1999). At increasing depths, leaf litter maintains a moist forest floor, serves as cover for amphibians and invertebrates, and influences forest floor chemistry (Faccio 2003).

However, species-specific amphibian responses at the stand level varied. We did not detect any difference in relative abundance of gulf coast toad, southern leopard frog, northern cricket frog, spring peeper, or Cope's gray treefrog across stand type at Sherburne. Abundance

of bronze frog, the most frequently trapped species in a bottomland hardwood forest in South Carolina, did not differ between control and group-selection stands, where estimates of canopy and leaf litter were lower than in control stands. Yet more Hylid individuals were captured in group-selection stands than in controls (Cromer et al. 2002). Likewise, abundance of bronze frog at Sherburne was independent of stand type based on visual encounter and anuran calling surveys. According to captures at drift fences, however, abundance of bronze frog mimicked that of amphibians. Similar to Cromer et al. (2002), we detected more calling individuals of squirrel treefrog in stands where selection cutting with groups occurred than in other stand types. Though not evident in measures of water cover, ephemeral pools were abundant on Sherburne. The short-term nature of ephemeral pools may exclude fish, and thus, are ideal breeding pools for anurans susceptible to fish predation (Babbitt and Tanner 2000). Also, herbaceous vegetation in group-selection stands may have provided locations for call perches or egg deposition (Cromer et al 2002).

Much of the research examining effects of even-aged management and forest age on amphibians during their terrestrial phase has focused on abundance and richness of salamanders (Grialou et al. 2000, Petranka et al. 1994, Dupuis et al. 1995), although at least one also has included anurans (Aubry 2000). In Washington, red-backed salamanders (*Plethodon vehiculum*) were captured more often in even-aged 45-60-year-old stands than 2-5-year-old stands over 2 years, whereas captures of ensatinas (*Ensatina eschscholtzii*) were similar across forest age in 1994 and were reduced in 2-5-year-old stands in 1995, suggesting that population responses of salamander species to forest age are variable (Grialou et al. 2000). Similarly, abundance and richness of southern Appalachian salamander communities in managed forests increased with increasing forest age in both wet and dry plots, with 0-5-year stands extremely depauperate of detections. Coarse woody debris also increased with forest age, but oldest stands (>120 years

old) and 0-5-year stands had the highest amount of coarse woody debris, though this lacked decay and was elevated in 0-5-year stands, providing unsuitable habitat for salamanders (Petranka et al. 1994). Researchers in Canada compared abundance of salamanders in old growth forests with 5-year, 17-18-year, and 54-72-year managed stands and concluded that even-aged practices such as clearcutting reduced amphibian populations by up to 70% in this area. A decrease in availability of moist microhabitats was believed to be the culprit and preservation of streamside buffers was suggested (Dupuis et al. 1995). Further, twice as many amphibians were captured in 50-70-year stands compared to 2-3-year, 12-20-year, and 30-40-year-old stands on an even-aged conifer plantation in Washington, and in contrast to other studies, variation in abundance of amphibians was not related to amount of coarse woody debris on the forest floor (Aubry 2000).

At Ben's Creek, we captured more species and individuals of amphibian in 1-year and 13-23-year stands than in other stand age classes. Other than percentage fern cover and woody cover, which were similar across stand age, none of the estimates of habitat characteristics we measured were unique to 1-year and 13-23-year stands. In 2 of the 13-23-year stand replicates, 1 of the 3 drift fence arrays was in close proximity to a stream. Captures in these 2 stands accounted for most detections of amphibian in this forest age class. All long-tailed salamanders (n=6) and 7 of 9 slimy salamanders were captured in one of these 2 arrays, which is consistent with studies that have recorded greater numbers of salamanders in older age stands that are in close proximity to riparian areas (Petranka et al. 1994, Aubry 2000, Dupuis et al. 1995).

Additionally, 3 of 4 1-year stand replicates were bordered by a streamside management zone, where running water was observed at least once during the study period. Although not evident by measurement of standing water during microhabitat surveys, we observed vernal pools in at least one of the stands. This contributed to our detection of individuals of calling

anurans in 1-year stands. Further, several species of calling anurans were detected for the first time in June 2004, during our final round of calling surveys at Ben's Creek. This survey followed several days of continuous precipitation, resulting in numerous pools of standing water in an otherwise dry area. To some extent, we believe that variation in captures and detections may be due to proximity and quality of adjacent aquatic habitats, as well as timing of surveys, rather than an effect related specifically to stand age (Russell et al. 2002, Babbitt and Tanner 2000).

We were unable to detect any differences in relative abundance and richness of reptiles associated with selection cutting at Sherburne, which agrees with other studies (Renken et al. 2004). This trend was consistent for abundance of green anole, ground skink, and *Agkistrodon* spp., each of which were detected by at least one method in every stand type. At Ben's Creek, however, we detected more species of reptile in 13-23-year stands, whereas relative abundance of lizards was independent of forest age. Green anole and eastern fence lizard were captured or encountered in every stand type, whereas 24 of 26 ground skinks were detected in 13-23-year stands, with the remainder in 7-9-year stands. Additionally, except for a single chance encounter in 1-year stands, five-lined and broadhead skinks (*Eumeces* spp.) were detected in only the older 2 age classes. It is possible that our rate of capture was too low to make any inference on relative abundance of reptiles at either study area. Furthermore, broad-level relationships with habitat characteristics at the stand level may have limited predictive value associated with occurrence of a species (Block and Morrison 1998). Basing conclusions on analyses of pooled species data could be misleading and potentially dangerous because unique species effects are easily masked by the captures of a few abundant species (deMaynadier and Hunter 1995).

Microhabitat Associations

Association of abundance of amphibians with understory vegetation, both herbaceous and woody, as well as canopy and forest floor characteristics such as coarse woody debris, has been documented. Microhabitat variables positively associated with number of species of amphibian in upland forests in Chicago included leaf litter and herbaceous cover (Nuzzo and Mierzwa 2000). Similarly, litter depth and understory vegetation positively influenced detections of salamander in managed conifer stands in New York (Pough et al. 1987). In addition to understory vegetation, distribution of salamanders in Canada was positively associated with soil moisture, logs, and ferns (Dupuis et al. 1995). On a Monterey pine (*Pinus radiata*) plantation in Australia, species of anuran were positively correlated with both herbaceous cover and shrubs in emergent, fringing vegetation of marshes and swamps (Parris and Lindenmayer 2004). Further, cane toads (*Bufo marinus*) on an Australian island used dense vegetation in more open areas in wetter conditions, whereas hollows under live trees and logs under a closed canopy were used in cooler and drier conditions. The authors noted that closed canopy conditions reduce wind speed and thus slow the rate of evaporation substantially (Seebacher and Alford 1999).

Abundance of anurans in a managed bottomland hardwood forest in South Carolina decreased with increasing herbaceous, vine, and ground cover, vegetation height, and vine vertical structure, but increased with increasing canopy and tree basal area. Abundance of salamanders similarly decreased with increasing herbaceous and ground cover, but increased with litter depth and canopy cover (Cromer et al. 2002), whereas diversity and richness of amphibians in pine and hardwood forests in Virginia were unrelated to number of canopy trees, tree diversity, or underground rocks (Mitchell et al. 1997).

Shrub cover and downed woody debris were unimportant to abundance of ensatinas and red-legged frogs detected on a conifer plantation in Washington, where abundance of these

species was positively correlated with litter depth and negatively correlated with cool, moist aspects and exposed rock (Aubry 2000). Abundance of some species of amphibian in Maine either were independent or negatively associated with downed woody debris, but positively associated with percent cover by snags, stumps, and associated root channels (deMaynadier and Hunter 1998). Prior to shelterwood harvests in Canada, abundance of salamanders was unrelated to understory vegetation and positively related to downed woody debris. After harvest, which resulted in twice as much downed woody debris on harvested sites, abundance of salamanders was positively related to understory vegetation and unrelated to downed woody debris, the influence of which may be confounded with changes in over- and understory cover (Morneault et al. 2004).

In the present study, detection of amphibians was associated with decreasing shrub cover at each of the 3 study areas. At Sherburne, detection of amphibians also was associated with increasing herbaceous cover (drift fences), and increasing canopy and bare ground (visual encounter surveys). Most amphibians detected at Sherburne were anurans, which need overstory canopy for temperature and moisture regulation (Bull and Wales 2001). Decreasing shrub cover was the most important determinant at the microhabitat level of occurrence of amphibians at Ben’s Creek, whereas occurrence of amphibians (specifically anurans) at Sandy Hollow increased with increasing herbaceous cover and decreasing fern cover, in addition to decreasing shrub cover. We were unable to detect an association between occurrence of amphibians and downed woody debris based on pooled species capture data, which is similar to previous findings (deMaynadier and Hunter 1998, Aubry 2000).

Occurrence of spring peeper was unrelated to habitat variables on Prince Edward Island, Canada, where this species was ubiquitous in both control and restored wetland study sites (Stevens et al. 2002). Conversely, abundance of spring peeper in upland forests in Chicago were

significantly positively associated with herbaceous cover and reduced distance to nearest pond (Nuzzo and Mierzwa 2000). Likewise, we found increasing herbaceous cover and water to be the strongest determinant of abundance of spring peeper during visual encounter surveys at Sherburne. Detections of spring peeper during visual encounter surveys were greater in uncut and individual-selection stands, which suggest that within stands where canopy closure generally was high, we detected more spring peepers in the vicinity of herbaceous cover and water.

Detections of Ranid species in other studies have been associated with herbaceous cover and canopy, among other microhabitat variables. Abundance of northern leopard frog (*Rana pipiens*) in upland forests in Chicago was positively related to herbaceous cover, and reduced distance to nearest pond (Nuzzo and Mierzwa 2000). Likewise, 2 Ranid species in habitat comprised of woodland, a conifer plantation, and farmland in the Netherlands were associated with grassy areas, whereas another species of Ranid remained close to water (Strijbosch 1980). Captures of aquatic ranids in managed forests in Maine were positively associated with canopy cover, conifer litter, and bole root, and negatively associated with ambient light. However, the best microhabitat model for these aquatic ranids included hardwood canopy and conifer litter (deMaynadier and Hunter 1998). Finally, occurrence of calling green frog (*Rana clamatans*) was positively associated with percent cattail (*Typha latifolia* L.) in restored wetlands in Canada, whereas occurrence of wood frog (*Rana sylvatica* Le Conte) in restored wetlands was ubiquitous and thus unrelated to habitat variables (Stevens et al. 2002).

Although we found no difference in abundance of *Rana* spp. at the stand level at Sherburne, examination of habitat associations at the microhabitat level suggests that variation in habitat influenced occurrence to some extent. Similar to abundance of amphibians, abundance of southern leopard frog and bronze frog captured during visual encounter surveys increased with increasing herbaceous cover and decreasing shrub cover, but increasing water and coarse woody

debris also were important. Increasing herbaceous cover and decreasing litter depth and shrub cover were primary determinants of occurrence of bronze frog, and also influenced occurrence of southern leopard frog. However, increasing canopy and decreasing vine cover, and increasing litter cover and decreasing vegetation height also were important to southern leopard frog, and may suggest either more widespread distribution or opportunistic foraging behavior of this species (Babbitt and Tanner 2000).

Dodd and Cade (1998) monitored amphibian movements from a temporary breeding pond in Florida and determined that eastern narrowmouth toads were likely to use a wide range of habitats. Similar to *Rana* spp., occurrence of eastern narrowmouth toad at Sherburne increased with increasing canopy cover and bare ground, but increasing tree richness and density, and decreasing downed woody debris also were important. Although eastern narrowmouth toads will utilize woody debris for shelter and moisture (Conant and Collins 1998), a negative association with downed woody debris may indicate that available downed woody debris lacked decay or sufficient size, or was isolated from the synergistic effects of other important variables (deMaynadier and Hunter 1998). In contrast, decreasing shrub cover was the most important determinant at the microhabitat level of occurrence of eastern narrowmouth toad at Ben's Creek. Decreasing shrub cover also positively influenced occurrence of eastern narrowmouth toad at Sandy Hollow, but other important microhabitat variables were increasing herbaceous cover (most important), increasing litter depth and decreasing bare ground, and, in contrast to occurrence at Sherburne, decreasing tree density and downed woody debris. That eastern narrowmouth toads were associated with a wide variety of habitat characteristics across study areas supports the generalist nature of this species (Conant and Collins 1998).

Abundance of snakes in managed bottomland hardwood forests in South Carolina was positively associated with canopy cover (Cromer et al. 2002). In contrast, Conroy (1999)

concluded that lizard assemblages did not seem to be related to variation in habitat in Australia. Instead, associations with habitat likely were species specific. In the present study, occurrence of reptiles at drift fences at Sherburne was associated with increasing tree richness and density as well as increasing vine cover and shrub richness. In slight contrast, increasing canopy, coarse woody debris, and bare ground, and decreasing fern and shrub cover influenced abundance of lizards detected during visual encounter surveys at Sherburne. At Ben's Creek, occurrence of lizards captured was associated with increasing canopy, tree density, litter cover, and vine cover, and decreasing shrub cover. In contrast, increasing shrub cover and shrub richness was the only microhabitat component associated with occurrence of reptiles at Sandy Hollow.

Two species of reptile, ground skink and eastern fence lizard, were detected with sufficient frequency at Sandy Hollow to examine association of occurrence with habitat at the microhabitat scale. Occurrence of ground skink increased with increasing shrub cover and decreasing herbaceous cover. Conversely, occurrence of eastern fence lizard was positively associated with increasing herbaceous cover and decreasing fern cover. Similarly, litter depth, development of grass and forbs, and cover provided by downed woody debris and rocks were important variables in habitat models explaining variation in occurrence of western fence lizard (*Sceloporus occidentalis*) in oak-pine woodlands of California (Block and Morrison 1998). In contrast, distributions of 2 lizard species, mountain spiny lizard (*Sceloporus jarrovii*) and tree lizard (*Urosaurus ornatus*), were not associated with vegetation type, but juveniles of both species were found farther from water and with less vegetative cover (Morrison et al. 1995). Eastern fence lizards in New Mexico were more frequently detected in arboreal locations than on the ground or under vegetation and were closely associated with plant cover at the microhabitat scale (Hager 2001). We also had a greater number of chance encounters with eastern fence lizards in arboreal locations, including both horizontal and vertical trunks and limbs of trees.

Study Limitations

The length of the study period may have prevented us from accurately detecting variation in distribution and abundance associated with habitat characteristics related to treatment effects. Instead, variation actually may be related to natural fluctuations in abundance over time (Renken et al. 2004). Though we tested for differences across season and between years, we did not incorporate rainfall or hydroperiod data into statistical analysis, which may have revealed trends in detection associated with variation rainfall or hydroperiod rather than treatment (Means et al. 2004). Fluctuations in the amphibian community are closely related to fluctuations in the environment, especially precipitation, and thus, more years of data are needed to gain a better understanding of community abundance associated with habitat characteristics at all scales (deMaynadier and Hunter 1995). It is likely that 2 years was insufficient to thoroughly assess the herpetofauna community.

In addition to the array of methods that we used to survey herpetofauna, surveys of vernal pools would have provided additional information for amphibians (Hecnar and M'Closkey 1998, Babbitt and Tanner 2000) as reproductive failure associated with changes in habitat characteristics may not be detected for several years for species primarily detected as adults (Pilliod et al. 2003). Furthermore, given a sufficient number of captures or detections, examination of associations between occurrence of herpetofauna and habitat characteristics at the microhabitat scale within treatments and across seasons would have provided more insight into responses to habitat features (Duguay and Wood 2002, Nuzzo and Mierzwa 2000). Finally, we did not assess models that included both microhabitat and landscape variable components to determine potential importance of 1 scale over another, nor did we test predictive models against actual data. In our case, this would have resulted in many component variables in the global model, which would have lead to poor goodness of fit, given our overall low and infrequent rate

of detection. Poorly fit models may have resulted in spurious outcomes, and lead to meaningless inferences on association of herpetofauna with habitat factors at the different scales we assessed.

Management Implications

Timber management which results in a mosaic of small-scale disturbances across the landscape would enable species sensitive to habitat fragmentation to maintain source populations. This also would maintain or restore herpetofauna diversity and community composition (Cromer et al. 2002). Presence of mature hardwood and either wetland or riparian habitat appears to be particularly important for amphibians. Thus, in managed landscapes such as even-aged pine plantations (Ben's Creek), preservation of streamside management zones is essential and would make forest management and maintenance of biodiversity more compatible (Renken et al. 2004). Additionally, preserving cool moist habitats, such as maintaining an even distribution of logs and snags as stable moist microclimates, or retaining understory sources of shade, would increase suitability of habitat, since amphibians, which have a low tolerance for hot, dry conditions, maximize foraging and breeding activity during cool moist conditions (Dupuis et al. 1995). Further, because anurans need overstory canopy for regulation of temperature and moisture (Bull and Wales 2001), management strategies that increase the proportion of older-aged stands within managed landscapes are likely to enhance the long-term habitat quality of intensively managed forest landscapes for terrestrial amphibians (Aubry 2000). However, our results suggest that responses to forest management are species-specific and may even differ for a single species across different habitat types, limiting the ability to make generalizations about effects of forest management on amphibian and reptile communities (deMaynadier and Hunter 1998).

CHAPTER 4. ASSOCIATIONS OF AVIFAUNA WITH FOREST MANAGEMENT, MICROHABITAT AND LANDSCAPE CHARACTERISTICS: RESULTS AND DISCUSSION

RESULTS: SHERBURNE WMA

Associations of Avifauna with Selection Cutting

We detected 8,167 individuals (44 species) within 50 m radius circular plots during April, May, and June of 2003 and 2004 (456 point surveys), excluding wading, nocturnal and incidental (<3 detections) species, and flyovers (Table 4.1). Mean species richness in group-selection (17.60 ± 0.33), individual-selection (18.36 ± 0.35), and uncut (17.83 ± 0.30) sites was similar ($F_{2,68.3}=0.66$, $P=0.521$). Mean relative diversity (J`) based on the Shannon-Weiner index in group-selection (0.623 ± 0.005), individual-selection (0.626 ± 0.006), and uncut (0.617 ± 0.004) sites also was similar.

Both of the migratory guilds differed across stand type. Relative abundance of the resident guild was greater ($F_{2,67.1}=5.02$, $P=0.009$) in individual and group-selection than uncut sites. Relative abundance of the migrant guild was greater ($F_{2,68.8}=6.07$, $P=0.004$) in uncut than group-selection sites, and similar in individual-selection sites to both. All of the habitat guilds differed across harvest strategy. Relative abundance of the forest habitat guild was greater ($F_{2,61.1}=38.74$, $P<0.001$) in uncut and individual-selection than group-selection sites. In contrast, relative abundance of the guild including second growth and forest opening inhabitants was greater ($F_{2,71.1}=0.66$, $P<0.001$) in group-selection compared to individual-selection and uncut sites. Relative abundance of the early-successional habitat guild was greatest ($F_{2,58.5}=87.28$, $P<0.001$) in group-selection sites and lowest in uncut sites. Finally, relative abundance of the generalist (or edge) habitat guild was greater in individual-selection ($F_{2,63.3}=3.51$, $P=0.036$) than group-selection sites, with uncut stands similar to both.

Of the 5 nesting guilds, relative abundances of 4 nesting guilds varied across stand type. Relative abundance of the forest canopy or subcanopy nesting guild was greatest in uncut sites and lowest in group-selection sites ($F_{2,66.3}$=27.76, P<0.001). However, relative abundance of the forest understory or ground nesting guild did not differ by stand type ($F_{2,58.2}$=2.16, P=0.125). Like the early-successional habitat guild, relative abundance of the early-successional understory or ground nesting guild was greatest in group-selection sites and lowest in uncut sites ($F_{2,65.6}$=93.51, P<0.001). In contrast, relative abundance of the cavity nesting guild was greater ($F_{2,72.1}$=13.03, P<0.001) in uncut and individual-selection than group-selection sites. Finally, relative abundance of the generalist (multi-habitat) nesting guild was greater ($F_{2,64.6}$=5.79, P=0.005) in individual-selection sites compared to uncut and group-selection sites, which were similar.

Relative abundance of all 6 foraging guilds differed across stand type. For both canopy gleaning and sallying guilds, relative abundance was greater ($F_{2,59.3}$=26.0, P<0.001 and $F_{2,72.7}$=10.14, P<0.001, respectively) in uncut and individual-selection sites than group-selection sites. Yet, relative abundance of the bark probing guild was greater ($F_{2,63.6}$=2.91, P=0.062) in individual-selection than uncut sites, with group-selection sites similar to both. In contrast, relative abundance of the ground gleaning guild was greater ($F_{2,67.2}$=5.0, P=0.009) in group and individual-selection sites compared to uncut sites, whereas relative abundance of aerial foragers was greater ($F_{2,71.2}$=3.50, P=0.035) in group-selection and uncut sites than individual-selection sites. Further, relative abundance of the shrub gleaning foraging guild was greater ($F_{2,67.1}$=20.79, P<0.001) in group-selection than either individual-selection or uncut sites (Table 4.2).

Table 4.1. Relative frequency of occurrence by stand type and guild classification of avian species detected within 50 m radius circular plots during point surveys at Sherburne WMA, LA, April-June 2003-04. Excludes wading, nocturnal and incidental species (≤3 detections), and flyovers. Species of concern indicated by asterisks:* (Audubon watchlist for LA);** (Audubon watchlist for NA).

	Stand Type			Guild[a]			
Species	Group (n^{b}=144)	Individ. (*n*=132)	Uncut (*n*=180)	Migratory	Habitat	Nesting	Foraging
Swallow-tailed Kite (*Elanoides forficatus*)**	0.021[c]	0	0	M	F	FC	AF
Mississippi Kite (*Ictina mississippiensis*)	0.028	0	0.017	M	F	FC	AF
Red-shouldered Hawk (*Buteo lineatus*)	0.042	0.045	0.078	R	F	FC	AF
Mourning Dove (*Zenaida macroura*)	0.007	0.045	0.033	R	E	EU	GG
Yellow-billed Cuckoo (*Coccyzus americanus*)*	0.201	0.394	0.283	M	F	FC	CG
Chimney Swift (*Chaetura pelagica*)	0.014	0	0	R	S	CA	AF
Ruby-throated Hummingbird (*Archilochus colubris*)	0.034	0.030	0.006	M	S	FU	SA
Red-headed Woodpecker (*Melanerpes erythrocephalus*)**	0.014	0	0.016	R	F	CA	BP
Red-bellied Woodpecker (*Melanerpes carolinus*)	0.053	0.652	0.467	R	F	CA	BP
Downy Woodpecker (*Picoides pubescens*)	0.257	0.250	0.222	R	F	CA	BP
Hairy Woodpecker (*Picoides villosus*)	0	0.008	0.011	R	F	CA	BP
Pileated Woodpecker (*Dryocopus pileatus*)	0.111	0.106	0.094	R	F	CA	BP
Eastern Wood-pewee (*Contopus virens*)	0.042	0	0	M	S	FC	SA
Acadian Flycatcher (*Empidonax virescens*)	0.507	0.864	0.889	M	F	FC	SA
Great crested Flycatcher (*Myiarchus crinitus*)	0.167	0.091	0.044	M	S	CA	SA
Blue Jay (*Cyanocitta cristata*)	0.125	0.023	0.022	R	S	FC	CG
American Crow (*Corvus brachyrhynchos*)	0.299	0.182	0.200	R	S	FC	GG
Fish Crow (*Corvus ossifragus*)	0.049	0.053	0.078	R	S	FC	GG
Carolina Chickadee (*Poecile carolinensis*)	0.292	0.455	0.472	R	G	CA	CG
Tufted Titmouse (*Baeolophus bicolor*)	0.569	0.697	0.750	R	G	CA	CG
Carolina Wren (*Thryothorus ludovicianus*)	0.882	0.947	0.911	R	G	GE	GG,SG

[a] Migratory guild: R (resident), M (migratory); habitat guild: F (forest), S (second growth forests or forests with openings), E (early successional), G (generalist or edge); nesting guild: FC (forest canopy or subcanopy), FU (forest understory or ground), EU (early successional understory or ground), CA (cavity), GE (multi-habitat); foraging guild: CG (canopy gleaner),SG (shrub gleaner), GG (ground gleaner), SA (sallier), BP (bark prober), AF (aerial forager).
[b] Total number of surveys conducted during study period.
[c] Number of surveys within harvest strategy in which species was detected divided by the total number of surveys in harvest strategy.
Table 4.1. continued.

	Stand Type			Guild			
Species	Group (*n*=144)	Individ. (*n*=132)	Uncut (*n*=180)	Migratory	Habitat	Nesting	Foraging
Blue-gray Gnatcatcher (*Polioptila caerula)*	0.375	0.447	0.417	R	F	FC	CG
Wood Thrush (*Hylocichla mustelina*)**	0.014	0.030	0.006	M	F	FU	GG
Gray Catbird (*Dumetella carolinensis*)	0.035	0.008	0	R	E	EU	SG
White-eyed Vireo (*Vireo griseus*)*	0.875	0.902	0.844	R	G	GE	SG
Yellow-throated Vireo (*Vireo flavifrons*)	0.049	0.045	0.089	M	S	FC	CG
Red-eyed Vireo (*Vireo olivaceus*)	0.375	0.689	0.767	M	F	FC	CG
Tennessee Warbler (*Vermivora peregrine*)	0.035	0	0.031	M	F	FU	CG
Northern Parula (*Parula Americana*)*	0.222	0.432	0.439	M	F	FC	CG
Yellow Warbler (*Dendroica petechia*)	0.007	0.008	0.006	M	S	FU	SG
Yellow-throated Warbler (*Dendroica dominica*)	0.014	0.053	0.050	M	F	FC	CG
American Redstart (*Setophaga ruticilla*)	0.396	0.500	0.628	M	F	FC	CG
Prothonotary Warbler (*Protonotaria citrea*)**	0.639	0.871	0.917	M	F	CA	SG
Swainson's Warbler (*Limnothlypis swainsonii*)**	0.118	0.212	0.04	M	F	FU	GG
Kentucky Warbler (*Oporornis formosus*)**	0.479	0.545	0.500	M	F	FU	GG
Common Yellowthroat (*Geothlypis trichas*)	0.479	0.015	0.011	R	E	EU	SG
Hooded Warbler (*Wilsonia citrine*)*	0.611	0.773	0.739	M	F	FU	SG
Yellow-breasted Chat (*Icteria virens*)	0.833	0.167	0.100	M	E	EU	SG
Summer Tanager (*Piranga rubra*)	0.160	0.136	0.161	M	S	FC	CG
Indigo Bunting (*Passerina cyanea*)	0.403	0.053	0.033	M	S	EU	SG
Painted Bunting (*Passerina ciris*)**	0.035	0.023	0.022	M	S	EU	SG
Northern Cardinal (*Cardinalis cardinalis*)	0.743	0.879	0.844	R	S	GE	SG,GG
Eastern Towhee (*Pipilo erthrophthalmus*)	0.528	0.280	0.194	R	E	EU	SG,GG
Brown-headed Cowbird (*Molothrus ater*)	0.201	0.106	0.089	R	G	GE	GG

Twenty-four of the 44 bird species detected during surveys met the minimum requirements of a minimum frequency of occurrence of 0.10 across all surveys and $\geq$ 50 detections. Of these 24 bird species, relative frequency of occurrence varied across stand type for 18 species (Table 4.3). Occurrence was greater in uncut and individual-selection relative to group-selection stands for 6 species (Acadian Flycatcher: $F_{2,73.4}$=18.24, *P*<0.001; Carolina Chickadee: $F_{2,455}$=6.04, *P*=0.003; Tufted Titmouse: $F_{2,64.6}$=5.05, *P*=0.009; Red-eyed Vireo: $F_{2,69}$=20.70, *P*<0.001; Northern Parula: $F_{2,63.1}$=5.1, *P*=0.009; Prothonotary Warbler: $F_{2,80.5}$=20.20, *P*<0.001 , scientific names are given in Table 4.1). Occurrence of Hooded Warbler and American Redstart was greater ($F_{2,450}$=3.39, *P*=0.034 and $F_{2,68.9}$=8.14, *P*<0.001, respectively) in uncut than group-selection stands, and similar in individual-selection stands to both.

In slight contrast, occurrence of Yellow-billed Cuckoo and Northern Cardinal was greater ($F_{2,66.1}$=5.47, *P*=0.006 and $F_{2,76.7}$=3.53, *P*=0.034, respectively) in individual and group-selection stands, with uncut stands similar to both. Further, occurrence of Red-bellied Woodpecker and Swainson's Warbler was greater ($F_{2,72}$=4.42, *P*=0.016 and $F_{2,67.3}$=4.91, *P*=0.01, respectively) in individual-selection than uncut stands, and similar in group-selection stands to both. Occurrence of Great-crested Flycatcher and Brown-headed Cowbird, was greater ($F_{2,75.2}$=3.92, *P*=0.024 and $F_{2,61.4}$=4.1, *P*=0.021, respectively) in group-selection than uncut stands, with individual-selection stands similar to both. Finally, occurrence of 4 species was greater in group-selection stands compared to both individual-selection and uncut stands (common yellowthroat: $F_{2,115}$=40.59, *P*<0.001; Yellow-breasted Chat: $F_{2,455}$=81.30, *P*<0.001; Indigo Bunting: $F_{2,80.3}$=23.59, *P*<0.001; Eastern Towhee: $F_{2,65.4}$=10.12, *P*<0.001).

Table 4.2. Mean relative abundance ($\bar{X}$) and standard error (SE) of migratory (2), habitat (4), nesting (5), and foraging (6) guilds by stand type based on detections during point count surveys at Sherburne WMA, LA, April-June 2003-2004.

	Stand Type					
	Group (*n*=144)		Individual (*n*=132)		Uncut (*n*=180)	
Guild Type	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Migratory Guild						
Resident	9.87A [a]	0.25	10.00A	0.25	9.01B	0.20
Migrant	7.73B	0.21	8.36AB	0.20	8.82A	0.20
Habitat Guild						
Forest	6.79B	0.25	9.70A	0.27	9.97A	0.23
Second growth or forest openings	2.96A	0.14	2.42B	0.13	2.21B	0.10
Early-successional	3.08A	0.16	0.66B	0.08	0.40C	0.05
Edge or generalist	4.77B	0.15	5.58A	0.16	5.25AB	0.14
Nesting Guild						
Forest canopy or subcanopy	3.81C	0.16	5.13B	0.19	5.93A	0.17
Forest understory or ground	1.74A	0.11	2.21A	0.13	1.74A	0.09
Early-successional understory	3.73A	0.19	0.75B	0.09	0.46C	0.06
Cavity	3.58B	0.19	4.73A	0.19	4.88A	0.16
Generalist (multi-habitat)	4.70B	0.16	5.55A	0.14	4.77B	0.12
Foraging Guild						
Canopy gleaning	3.74B	0.18	5.18A	0.20	5.97A	0.18
Shrub gleaning	10.11A	0.25	8.96B	0.19	8.18B	0.18
Ground gleaning	5.14A	0.19	5.43A	0.18	4.51B	0.13
Sallying	0.99B	0.08	1.42A	0.08	1.52A	0.06
Bark probing	1.13AB	0.09	1.28A	0.09	0.92B	0.07
Aerial foraging	0.13A	0.04	0.05B	0.02	0.12A	0.03

[a] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons (alpha=0.10).

Table 4.3. Mean relative frequency of occurrence by stand type for bird species with ≥ 0.10 frequency of occurrence during point count surveys at Sherburne WMA, LA, April-June 2003-2004.

	Stand Type					
	Group (*n*=144)		Individual (*n*=132)		Uncut (*n*=180)	
Species	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Yellow-billed Cuckoo	0.20B[a]	0.03	0.39A	0.04	0.28AB	0.03
Red-bellied Woodpecker	0.52AB	0.04	0.65A	0.04	0.47B	0.04
Downy Woodpecker	0.26A	0.04	0.25A	0.04	0.22A	0.03
Acadian Flycatcher	0.53B	0.04	0.84A	0.03	0.89A	0.02
Great-crested Flycatcher	0.17A	0.03	0.08AB	0.02	0.04B	0.02
Carolina Chickadee	0.29B	0.04	0.46A	0.04	0.47A	0.04
Tufted Titmouse	0.57B	0.04	0.69A	0.04	0.75A	0.03
Carolina Wren	0.88A	0.03	0.95A	0.02	0.90A	0.02
Blue-gray Gnatcatcher	0.37A	0.04	0.45A	0.04	0.42A	0.04
White-eyed Vireo	0.88A	0.03	0.89A	0.03	0.84A	0.03
Red-eyed Vireo	0.38B	0.04	0.69A	0.04	0.77A	0.03
Northern Parula	0.22B	0.04	0.43A	0.04	0.44A	0.04
American Redstart	0.38B	0.04	0.50AB	0.04	0.63A	0.04
Prothonotary Warbler	0.64B	0.04	0.87A	0.03	0.92A	0.02
Swainson's Warbler	0.12AB	0.03	0.21A	0.04	0.04B	0.02
Kentucky Warbler	0.48A	0.04	0.54A	0.04	0.50A	0.04
Common Yellowthroat	0.48A	0.04	0.02B	0.01	0.01B	0.01
Hooded Warbler	0.61B	0.04	0.76AB	0.04	0.74A	0.03
Yellow-breasted Chat	0.83A	0.03	0.39B	0.04	0.10B	0.02
Summer Tanager	0.16A	0.03	0.14A	0.03	0.16A	0.03
Indigo Bunting	0.40A	0.04	0.06B	0.02	0.03B	0.01
Northern Cardinal	0.74B	0.04	0.88A	0.03	0.85AB	0.03
Eastern Towhee	0.52A	0.04	0.28B	0.04	0.19B	0.03
Brown-headed Cowbird	0.20A	0.03	0.11AB	0.03	0.09B	0.02

[a] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons (alpha=0.10).

Associations of Microhabitat with Selection Cutting

Nine of 12 microhabitat variables measured at avian point count stations differed across stand type (Table 4.4). Percentages of litter ($F_{2,24}$=12.07, P<0.001), bare ground ($F_{2,24}$=9.11, P=0.001), fern cover ($F_{2,24}$=7.36, P=0.003) and canopy cover ($F_{2,24}$=32.56, P<0.001), number of trees ($F_{2,24}$=15.80, P<0.001) and tree species richness ($F_{2,24}$=6.90, P=0.004), were greater in both uncut and individual-selection sites compared to group-selection sites. Number of shrubs and

shrub species richness were greater ($F_{2,24}$=23.77, P<0.001 and $F_{2,24}$=8.16, P=0.02, respectively) in group and individual-selection sites compared to uncut sites. Finally, percentage vine cover, based on 2004 measurements only, was greater ($F_{2,11}$=13.73, P=0.001) in group-selection stands than either individual-selection or uncut stands, which were similar.

Table 4.4. Mean value ($\bar{X}$) and standard error (SE) of variables associated with avian point count stations by stand type at Sherburne WMA, LA, 2003-2004.

	Stand Type					
	Group		Individual		Uncut	
Variable	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
% Litter	18.33B[a]	1.68	34.65A	1.47	32.31A	1.67
% Herbaceous cover	13.56A	1.79	11.50A	1.15	8.43A	0.64
% Fern cover	11.35B	1.71	19.26A	2.10	25.53A	1.97
% Woody cover (2004 only)	8.24A	0.60	8.18A	1.21	7.28A	0.82
% Vine cover (2004 only)	29.95A	3.21	10.33B	1.10	9.52B	1.18
% Bare ground	2.90B	0.50	6.71A	0.66	7.74A	0.80
% Coarse woody debris	8.65A	0.64	7.76A	0.49	7.63A	0.58
% Canopy cover	79.05B	1.84	93.8A	0.37	94.12A	0.25
Total # shrubs	114.02A	7.35	169.80A	17.66	41.6B	1.94
Shrub species richness	11.96A	0.52	10.40A	0.23	8.77B	0.20
Total # trees	19.56B	1.26	31.00A	1.10	31.73A	1.09
Tree species richness	7.15B	0.35	9.00A	0.25	8.73A	0.21

[a] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons (alpha=0.10).

Associations of Avifauna with Microhabitat

None of the 11 microhabitat variables retained for analysis were highly correlated (>0.80). Thus, principal components analysis (PCA) was performed with all variables included. Based on eigenvalues $\geq$ 1.0, 4 principal components were retained, accounting for 66% of the variance (Table 4.5). Component 1 was interpreted as a gradient from areas with many trees, a closed canopy, and abundant litter (positive loadings), to areas where vine cover (primarily *Rubus* spp., negative loading) was common (Table 4.6). Component 2 was interpreted as a gradient from areas with a variety of shrubs (positive loading) to areas with bare ground (negative loading). Shrub density and woody cover were positively loaded on component 3,

whereas herbaceous cover was negatively loaded. Finally, coarse woody debris was positively loaded on component 4, while fern cover was negatively loaded.

Table 4.5. Principal components (PC) analysis results for microhabitat variables surveyed along transects bisecting avian point count stations at Sherburne WMA, LA.

	PC1	PC2	PC3	PC4
Eigenvalue	3.55	1.88	1.44	1.02
Variance explained	0.30	0.16	0.12	0.08
Variable:				
% Litter	**74**[a,b]	-7	1	40
% Herbaceous cover	-1	37	**-64**	0
% Fern cover	-1	-51	30	**-67**
% Woody cover	-2	18	**77**	-6
% Vine cover	**-66**	54	0	22
% Bare ground	6	**-65**	-13	-3
% Coarse woody debris	-3	-6	6	**83**
% Canopy cover	**71**	-42	20	-11
# Shrubs	-3	18	**69**	0
# Trees	**81**	-19	-3	5
Shrub species richness	-26	**74**	8	-3
Tree species richness	**81**	11	-17	-16

[a] Correlation coefficients are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.

Table 4.6. Principal components (PC, eigenvalues $\geq$ 1) derived from microhabitat variables associated with avian point count stations at Sherburne WMA, LA, 2003-2004. Associated variables are those with a correlation coefficient of $\geq |0.40|$ with each respective PC.

PC	Associated Variables[a]	Interpretation
PC1	# Trees, Tree richness, % Canopy cover, % Litter, (-) % Vine cover	Tree richness and density, litter
PC2	Shrub richness, (-) % Bare ground	Shrub richness
PC3	% Woody cover, # Shrubs, (-) % Herbaceous cover	Shrub density, woody cover
PC4	% Coarse woody debris, (-) % Fern cover	Coarse woody debris

[a] Variables are positively related to the principal component unless otherwise noted.

Occurrence of each the 24 species was examined using the 4 principal components as explanatory variables. The best approximating model for occurrence of Prothonotary and Hooded Warblers retained one variable, tree richness and density and litter (component 1), which positively influenced occurrence, and was present in all models with substantial empirical support for both species. Nine models with varying combinations of the 4 principal components gained substantial empirical support in explaining variation in occurrence of Swainson's Warbler. Tree richness and density and litter (component 1) and shrub density and woody cover (component 3) were retained within the best approximating model, in which both variables positively influenced occurrence of Swainson's Warbler, and had higher estimates of importance relative to other variables.

Three models were retained in the candidate model subset for Acadian Flycatcher. The best approximating model included tree richness and density and litter (component 1), shrub richness, vine and herbaceous cover (component 2), and shrub density and woody cover (component 3). Tree richness and density and litter, the strongest variable, and shrub density and woody cover each had a positive influence on occurrence of Acadian Flycatcher, whereas shrub richness, vine and herbaceous had a negative influence. The best predictor of occurrence of Downy Woodpecker was the model with tree richness and density and litter (component 1), which had a negative influence. However, the slope of this variable was weak, suggesting that habitat variables associated with component 1 may not be very important in relation to occurrence Downy Woodpecker.

Based on the best model, occurrence of Indigo Bunting was negatively related to tree richness and density and litter (component 1), and positively related to shrub density and woody cover (component 3), and coarse woody debris and litter (component 4). Estimates of relative importance were highest for tree richness and density and litter, followed by coarse woody debris

and litter and shrub density and woody cover. Similarly, occurrence of Common Yellowthroat was strongly and negatively influenced by tree richness and density and litter (component 1), but, in contrast to Indigo Bunting, strongly and negatively influenced by shrub density and woody cover, based on the best approximating model. The only other model in the candidate model subset included shrub richness, vine and herbaceous cover (component 2), which had a positive influence but was substantially less important.

The best approximating model for occurrence of both Yellow-billed Cuckoo and Blue-gray Gnatcatcher included shrub richness, vine and herbaceous cover (component 2), which positively influenced frequency of occurrence. However, the slope for shrub richness, vine and herbaceous cover for both species was weak, suggesting that influence by this variable may not be important. In contrast, frequency of occurrence of Carolina Wren decreased with increasing shrub richness, vine and herbaceous cover, according to the best approximating model.

The best approximating model in explaining variation in occurrence of Tufted Titmouse and Northern Cardinal each included shrub density and woody cover (component 3), which was supported by estimates of relative importance for this variable for each species. Frequency of occurrence of both species increased with increasing shrub density and woody cover. However, the slope for this variable associated with occurrence of Tufted Titmouse was close to zero, suggesting that the association between occurrence of this species and shrub density and woody cover may be weak.

Occurrence of Red-eyed Vireo, American Redstart, and Yellow-breasted Chat, were each strongly and negatively associated with shrub density and woody cover (component 3). For Red-eyed Vireo and American Redstart, the model with shrub density and woody cover was the only model which gained substantial empirical support, and was 1 of only 2 models which gained support in explaining occurrence of Yellow-breasted Chat. Estimates of relative importance for

Table 4.7. Model selection results with respect to microhabitat for select bird species detected during avian point counts at Sherburne, WMA. Models reported are those with ΔAICc ≤ 2. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and best model, and w_i is Akaike weight.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Yellow-billed Cuckoo	PC2 [a]	2	467.90	471.93	0	0.432
Red-bellied Woodpecker	PC4	2	546.88	550.90	0	0.369
	PC2 PC4	3	546.46	552.51	1.61	0.165
Downy Woodpecker	PC1	2	398.50	402.53	0	0.188
	PC4	2	399.36	403.38	0.86	0.122
	PC3	2	399.46	403.49	0.96	0.166
	PC2	2	399.55	403.57	1.05	0.111
	PC1 PC2	3	398.03	404.08	1.56	0.086
	PC1 PC4	3	398.387	404.44	1.91	0.384
Acadian Flycatcher	PC1 PC2 PC3	4	330.33	338.42	0	0.423
	PC1 PC2 PC3 PC4	5	329.77	339.90	1.48	0.202
	PC1 PC2	3	334.19	340.25	1.82	0.170
Great-crested Flycatcher	PC4	2	153.46	157.48	0	0.198
	PC3	2	153.99	158.01	0.53	0.152
	PC1	2	154.58	158.61	1.13	0.113
	PC2	2	155.04	159.06	1.58	0.090
	PC3 PC4	3	153.24	159.29	1.81	0.080
Carolina Chickadee	PC4	2	599.15	603.18	0	0.457
	PC3	2	600.49	604.52	1.34	0.234
Tufted Titmouse	PC3	2	445.03	449.05	0	0.360
	PC4	2	445.83	449.86	0.80	0.240
	PC2	2	446.40	450.42	1.37	0.181
Carolina Wren	PC2	2	189.26	193.29	0	0.310
	PC2 PC4	3	188.28	194.34	1.05	0.183
White-eyed Vireo	PC1	2	319.11	323.13	0	0.157
	PC4	2	319.45	323.48	0.34	0.132
	PC3	2	319.45	323.48	0.35	0.132
	PC2	2	319.45	323.48	0.35	0.132
	PC1 PC4	3	318.91	324.96	1.83	0.063
	PC1 PC2	3	318.97	325.02	1.89	0.061
	PC1 PC3	3	318.98	325.04	1.90	0.061

[a] PC1=tree richness and density, litter; PC2=shrub richness, vine and herbaceous cover; PC3= shrub density and woody cover; PC4= coarse woody debris and litter

Table 4.7. continued.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Blue-gray Gnatcatcher	PC2	2	454.76	458.78	0	0.912
Red-eyed Vireo	PC3	2	441.40	445.43	0	0.787
Northern Parula	PC4	2	383.86	387.89	0	0.280
	PC1 PC2 PC4	4	380.27	388.36	0.47	0.221
	PC3 PC4	3	383.21	389.27	1.38	0.140
	PC2 PC4	3	383.81	389.86	1.97	0.104
American Redstart	PC3	2	484.50	488.53	0	0.533
Prothonotary Warbler	PC1	2	297.49	301.52	0	0.475
	PC1 PC3	3	297.39	303.45	1.93	0.181
	PC1 PC4	3	297.49	303.54	2.03	0.173
Swainson's Warbler	PC1 PC3	3	199.70	205.76	0	0.173
	PC1 PC2 PC3	4	198.26	206.35	0.59	0.128
	PC1	2	202.49	206.52	0.76	0.118
	PC1 PC3 PC4	4	198.88	206.97	1.21	0.094
	PC1 PC2	3	201.21	207.26	1.51	0.081
	PC1 PC4	3	201.45	207.51	1.75	0.072
	PC3	2	203.57	207.60	1.84	0.069
	PC1 PC2 PC3 PC4	5	197.61	207.74	1.99	0.064
Kentucky Warbler	PC4	2	526.74	530.77	0	0.246
	PC2 PC4	3	526.29	532-34	1.57	0.112
	PC3 PC4	3	526.41	532.47	1.70	0.106
Common Yellowthroat	PC1 PC3	3	168.29	174.35	0	0.454
	PC1 PC2 PC3	4	168.04	176.13	1.78	0.186
Hooded Warbler	PC1	2	476.61	480.63	0	0.247
	PC1 PC3	3	475.22	481.28	0.65	0.179
	PC1 PC4	3	475.93	481.98	1.35	0.126
	PC1 PC2	3	476.38	482.43	1.80	0.100
	PC1 PC3 PC4	4	474.41	482.50	1.87	0.097
Summer Tanager	PC1	2	386.79	390.82	0	0.165
	PC4	2	387.05	391.07	0.25	0.146
	PC3	2	387.68	391.71	0.89	0.106
	PC2	2	387.69	391.71	0.89	0.106
	PC1 PC4	3	386.11	392.16	1.34	0.085
	PC1 PC3	3	386.69	392.74	1.92	0.063

Table 4.7. continued.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Yellow-breasted Chat	PC3	2	300.08	304.10	0	0.406
	PC4	2	301.28	305.31	1.21	0.222
Indigo Bunting	PC1 PC3 PC4	4	194.83	202.92	0	0.333
	PC1 PC4	3	196.98	203.03	0.11	0.315
Northern Cardinal	PC3	2	350.54	354.57	0	0.447
	PC4	2	352.26	356.29	1.72	0.189
Eastern Towhee	PC1	2	395.52	399.55	0	0.186
	PC2	2	395.67	399.69	0.14	0.173
	PC4	2	395.86	399.88	0.34	0.157
	PC3	2	397.00	401.03	1.48	0.089
	PC2 PC4	3	395.06	401.11	1.56	0.085
	PC1 PC4	3	395.29	401.34	1.79	0.076
Brown-headed Cowbird	PC4	2	255.05	259.07	0	0.443
	PC2 PC4	3	254.64	260.69	1.61	0.198

this variable were high for each of these species, providing support that these models were good predictors of occurrence for these species.

The best approximating model for occurrence of Northern Parula, Red-bellied Woodpecker, and Brown-headed Cowbird predicted that frequency of occurrence was positively associated with coarse woody debris and litter (component 4). In each case, this variable also carried the greatest relative weight. In contrast, occurrence of Great-crested Flycatcher, Carolina Chickadee and Kentucky Warbler were negatively influenced by percentage of coarse woody debris and litter, the only variable in the best approximating model for each of these 3 species. However, estimates of relative importance for occurrence of Great-crested Flycatcher indicate that the influence of coarse woody debris and litter may not be very important. As well, parameter estimates for great-crested flycatcher, Kentucky Warbler, and Carolina Chickadee indicate that the influence of coarse woody debris and litter may be relatively weak.

For White-eyed Vireo, Summer Tanager, and Eastern Towhee, the best approximating model included tree richness and density and litter (component 1), which had a negative association with occurrence of each species. However, model-averaged weights for tree richness and density and litter did not differ substantially from weights of the other 3 model variables for any of these species, suggesting that the association of tree richness and density and litter with occurrence of these species is not very strong. Further, parameter estimates for this variable for both Great-crested Flycatcher and White-eyed Vireo were relatively low, indicating a weak influence of tree richness and density and litter on their occurrence. For Eastern Towhee, the parameter estimate associated with tree richness and density and litter was relatively large, but estimates of relative weights contrasted with this result (Table 4.8).

Table 4.8. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory variable, representative of microhabitat, for select avian species detected during avian point counts at Sherburne WMA, LA, 2003-2004.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Yellow-billed Cuckoo	PC1	0.0184	0.0100	0.086
	PC2	0.0445	0.0876	0.767
	PC3	0.0056	0.0282	0.249
	PC4	0.0064	0.0358	0.333
Red-bellied Woodpecker	PC1	0.0492	0.0295	0.277
	PC2	-0.0062	0.0351	0.328
	PC3	0.0056	0.0258	0.240
	PC4	0.2571	0.0977	0.943
Downy Woodpecker	PC1	-0.0427	0.0616	0.513
	PC2	0.0539	0.0454	0.365
	PC3	-0.0039	0.0421	0.348
	PC4	-0.0021	0.0421	0.361
Acadian Flycatcher	PC1	0.6899	0.1158	0.904
	PC2	-0.4480	0.1124	0.877
	PC3	0.2154	0.1098	0.746
	PC4	-0.0397	0.0367	0.304
Great-crested Flycatcher	PC1	-0.0442	0.0552	0.349
	PC2	0.0190	0.0488	0.299
	PC3	-0.0647	0.0679	0.416
	PC4	-0.0761	0.0686	0.493
Carolina Chickadee	PC1	0.0041	0.0014	0.014
	PC2	-0.0108	0.0187	0.192
	PC3	-0.0265	0.0381	0.385
	PC4	-0.0437	0.0610	0.631
Tufted Titmouse	PC1	0.0001	0	0.000
	PC2	-0.0381	0.037	0.309
	PC3	0.0013	0.0622	0.532
	PC4	-0.0233	0.0438	0.394
Carolina Wren	PC1	0.0664	0.0364	0.247
	PC2	-0.2082	0.1345	0.876
	PC3	0.0045	0.0440	0.275
	PC4	-0.0458	0.0572	0.397

[a] PC1=tree richness and density, litter; PC2=shrub richness, vine and herbaceous cover; PC3= shrub density and woody cover; PC4= coarse woody debris and litter.

Table 4.8. continued.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Blue-gray Gnatcatcher	PC1	0.0017	0.0021	0.018
	PC2	0.0124	0.1078	0.936
	PC3	-0.0011	0.0039	0.033
	PC4	0.0122	0.0079	0.073
White-eyed Vireo	PC1	-0.0417	0.0589	0.424
	PC2	0.0243	0.0492	0.378
	PC3	-0.0265	0.0509	0.378
	PC4	0.0196	0.0514	0.381
Red-eyed Vireo	PC1	0.0084	0.0021	0.017
	PC2	-0.0057	0.0035	0.028
	PC3	-0.1694	0.1163	0.952
	PC4	-0.0226	0.019	0.173
Northern Parula	PC1	0.3179	0.0634	0.422
	PC2	-0.0511	0.0662	0.466
	PC3	0.0044	0.0382	0.292
	PC4	0.4009	0.1141	0.978
American Redstart	PC1	0.0133	0.0064	0.057
	PC2	-0.0664	0.025	0.218
	PC3	-0.1756	0.1000	0.869
	PC4	-0.0236	0.0284	0.266
Prothonotary Warbler	PC1	0.6188	0.1201	0.894
	PC2	-0.0002	0.0001	0.000
	PC3	-0.0411	0.0436	0.321
	PC4	-0.0078	0.0368	0.287
Swainson’s Warbler	PC1	0.1171	0.0796	0.774
	PC2	-0.0340	0.0411	0.410
	PC3	0.1821	0.0661	0.605
	PC4	0.0394	0.0391	0.368
Kentucky Warbler	PC1	0.0036	0.0321	0.294
	PC2	-0.0105	0.0313	0.364
	PC3	0.0074	0.0387	0.350
	PC4	-0.0808	0.0698	0.678
Common Yellowthroat	PC1	-0.8996	0.1346	0.940
	PC2	0.2349	0.0499	0.323
	PC3	-0.6073	0.1639	0.931
	PC4	-0.0033	0.0383	0.280

Table 4.8. continued.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Hooded Warbler	PC1	0.2457	0.0973	0.914
	PC2	-0.0158	0.0315	0.302
	PC3	0.0601	0.0516	0.444
	PC4	-0.0365	0.0387	0.354
Yellow-Breasted chat	PC1	-0.1310	0.0231	0.153
	PC2	0.0512	0.0138	0.089
	PC3	-0.1324	0.1018	0.654
	PC4	0.0203	0.0479	0.401
Summer Tanager	PC1	-0.0583	0.0596	0.473
	PC2	0.0031	0.0436	0.346
	PC3	0.0204	0.0448	0.353
	PC4	-0.0393	0.0568	0.436
Indigo Bunting	PC1	-1.0544	0.1512	0.997
	PC2	0.0657	0.0304	0.201
	PC3	0.1393	0.0784	0.559
	PC4	0.3132	0.1101	0.812
Northern Cardinal	PC1	0.0099	0.0060	0.048
	PC2	-0.0118	0.0203	0.158
	PC3	0.1230	0.0991	0.709
	PC4	0.0089	0.0496	0.406
Eastern Towhee	PC1	-0.2505	0.0501	0.390
	PC2	-0.0099	0.0550	0.421
	PC3	0.0200	0.0305	0.234
	PC4	-0.0274	0.0471	0.417
Brown-headed Cowbird	PC1	-0.0373	0.0230	0.174
	PC2	0.0897	0.0533	0.349
	PC3	0.0181	0.0279	0.209
	PC4	0.3884	0.1353	0.982

a Abundance data analyzed with linear regression. For every unit increase in the explanatory variable, the change in the species abundance is the parameter estimate (Perkins et al. 2003).
b Abundance data analyzed with logistic regression. For every unit increase in the explanatory variable, the odds of presence increase/decrease by exp(Estimate) (Perkins et al. 2003).

RESULTS: BEN'S CREEK WMA

Associations of Avifauna with Stand Age

We detected 6,354 individuals (44 species) within 50 m radius circular plots during April, May, and June of 2003 and 2004 (576 point count surveys), excluding wading, nocturnal and incidental ($\leq$3 detections) species, as well as flyovers (Table 4.9). Mean species richness in 4-5 year (7.50 $\pm$ 0.17) and 13-23 year (7.52 $\pm$ 0.18) stands was similar and greater ($F_{3,82.5}$=6.06, P=0.009) than in 7-9 year stands (6.32 $\pm$ 0.17), and similar in 1 year stands (6.66 $\pm$ 0.19) to other stand age classes. Likewise, mean relative diversity (J`) based on the Shannon-Weiner index in 4-5 year (0.496 $\pm$ 0.007) and 13-23 year (0.498 $\pm$ 0.007) stands was similar and greater ($F_{3,78.2}$=4.98, P=0.003) than 7-9 year stands (0.448 $\pm$ 0.008), and similar in 1 year stands (0.458 $\pm$ 0.009) to other stand age classes.

Relative abundance of both migratory guilds differed across stand age (Table 4.10). Relative abundance of residents was greater ($F_{3,81.7}$=6.66, P<0.001) in 4-5-year than 1-year stands, with abundance in other stand age classes similar and intermediate between both. In contrast, abundance of migrants was greater ($F_{3,85.9}$=5.35, P=0.002) in 1-year than 7-9-year stands, with occurrence in 4-5- and 13-23-year stands similar and intermediate to both.

Among the 4 habitat guilds, response to stand age was variable. Relative abundance of stand inhabitants was similar and greatest in 7-9- and 13-23-year stands, followed by 4-5-year stands, and was lowest in 1-year stands ($F_{3,72.1}$=71.94, P<0.001). In comparison, species more strongly associated with early-successional habitat were more abundant ($F_{3,69.3}$=61.23, P<0.001) in 1- and 4-5-year stands. Yet, relative abundance of inhabitants of second growth and/or stand openings was similar across all stand age classes ($F_{3,87.1}$=1.25, P=0.298). This was in slight contrast to relative abundance of generalist species, which was lower in 1-year stands compared to all other age classes ($F_{3,81.6}$=11.03, P<0.001).

As well, response of each of the 5 nesting guilds age differed across stand age. Relative abundance of species that nest in the forest canopy or subcanopy was greatest in 13-23-year stands, and, predictably, decreased with decreasing stand age ($F_{3,82.6}$=16.38, P<0.001). Following a somewhat similar trend, relative abundance of species that typically nest in forest understory or on the ground was similar and greater in 7-9- and 13-23-year stands, followed by 4-5-year stands, and lowest in 1-year stands ($F_{3,86.2}$=57.29, P<0.001). In direct contrast, relative abundance of species that typically nest in early-successional habitat was greatest in 1-year stands, decreased in 4-5-year stands, and was lowest in 7-9- and 13-23-year stands, which were similar ($F_{3,68.5}$=82.83, P<0.001). Not surprisingly, relative abundance of cavity nesters was greater in 13-23-year stands than other stand age classes ($F_{3,86.9}$=8.67, P<0.001). Finally, species in the generalist nesting guild displayed a pattern of relative abundance identical to that in the generalist habitat guild, in which relative abundance was lower in 1-year stands than other stand age classes ($F_{3,74.4}$=26.14, P<0.001).

Of the 6 foraging guilds, data was sufficient for all but aerial foragers, which was comprised of 2 species and too few detections for analysis. Thus, response to stand age is reported for the remaining 5. Relative abundance of canopy gleaners was greatest in 13-23-year stands, and decreased with decreasing stand age ($F_{3,92.1}$=12.54, P<0.001). Similarly, relative abundance of bark probers was greater in 13-23- than 1-year stands, with abundance in other age classes similar and intermediate between both ($F_{3,108}$=7.06, P<0.001). Salliers and shrub gleaners also shared similar response patterns to stand age. Relative abundance of salliers was greater in 1-year than 7-9-year stands, with abundance in other age classes similar and intermediate ($F_{3,88.5}$=2.53, P=0.063), whereas relative abundance of shrub gleaners was similar and greater in 1- and 4-5-year stands than 7-9- and 13-23-year stands ($F_{3,81}$= 9.70, P<0.001). Relative abundance of ground gleaners was similar across stand age ($F_{3,90.9}$= 1.79, P=0.163).

Table 4.9. Relative frequency of occurrence by stand age, and guild classification of birds detected within 50 m radius circular plots during point count surveys at Ben's Creek WMA, LA, April-June 2003-2004. Excludes wading, nocturnal and incidental species (≤3 detections), and flyovers. Species of concern indicated with asterisks: * (Audubon watchlist, LA); ** (Audubon watchlist, NA).

	Stand Age[a]				Guild[b]			
Species	1 yr	4-5 yr	7-9 yr	13-23 yr	Migratory	Habitat	Nesting	Foraging
Northern Bobwhite (*Colinus virginianus*)*	0.063	0.049	0.028	0.007	R	E	EU	GG
Mourning Dove (*Zenaida macroura*)	0	0.090	0.069	0.021	R	E	EU	GG
Yellow-billed Cuckoo (*Coccyzus americanus*)*	0.007	0	0.083	0.076	M	F	FC	CG
Common Nighthawk (*Chordeiles minor)*	0.007	0.070	0	0.007	M	G	EU	AF
Chimney Swift (*Chaetura pelagica*)	0.014	0.007	0	0.014	R	S	CA	AF
Ruby-throated Hummngbrd (*Archilochus colubris*)	0.007	0.014	0.007	0.007	M	S	FU	SA
Red-bellied Woodpecker (*Melanerpes carolinus*)	0.063	0.146	0.104	0.194	R	F	CA	BP
Downy Woodpecker (*Picoides pubescens*)	0.014	0.007	0.014	0.049	R	F	CA	BP
Northern Flicker (*Colaptes auratus)*	0.014	0.035	0.007	0.028	R	F	CA	BP
Pileated Woodpecker (*Dryocopus pileatus*)	0	0.021	0.007	0.056	R	F	CA	BP
Eastern Wood-pewee (*Contopus virens*)	0	0	0.014	0.028	M	S	FC	SA
Great crested Flycatcher (*Myiarchus crinitus*)	0.056	0.042	0.049	0.083	M	S	CA	SA
Eastern Kingbird (*Tyrannus tyrannus*)	0.125	0.014	0.007	0.014	M	E	EU	SA
Blue Jay (*Cyanocitta cristata*)	0.139	0.243	0.326	0.292	R	S	FC	CG
American Crow (*Corvus brachyrhynchos*)	0.035	0.076	0.021	0.111	R	S	FC	GG
Carolina Chickadee (*Poecile carolinensis*)	0.076	0.118	0.069	0.153	R	G	CA	CG
Tufted Titmouse (*Baeolophus bicolor*)	0.299	0.243	0.208	0.431	R	G	CA	CG
Brown-headed Nuthatch (*Sitta pusilla*)**	0.007	0	0.007	0.014	R	F	CA	CG
Carolina Wren (*Thryothorus ludovicianus*)	0.438	0.549	0.542	0.806	R	G	GE	GG,SG
Blue-gray Gnatcatcher (*Polioptila caerula)*	0.056	0.049	0.069	0.097	R	F	FC	CG
Eastern Bluebird (*Sialia sialis*)	0.028	0	0	0.007	R	E	CA	GG

[a] *N*=144 surveys conducted in each stand age class during study period.
[b] Migratory guild: R (resident), M (migratory); habitat guild: F (forest), S (second growth forests or forests with openings), E (early successional), G (generalist or edge); nesting guild: FC (forest canopy or subcanopy), FU (forest understory or ground), EU (early successional understory or ground), CA (cavity), GE (multi-habitat); foraging guild: CG (canopy gleaner),SG (shrub gleaner), GG (ground gleaner), SA (sallier), BP (bark prober), AF (aerial forager).
[c] Number of surveys within stand age in which species was detected, divided by the total number of surveys in harvest strategy.

Table 4.9. continued.

	Stand Age				Guild			
Species	1 yr	4-5 yr	7-9 yr	13-23 yr	Migratory	Habitat	Nesting	Foraging
Wood Thrush (*Hylocichla mustelina*)**	0.014	0.063	0.139	0.215	M	F	FU	GG
Gray Catbird (*Dumetella carolinensis*)	0.014	0.118	0.042	0	R	E	EU	SG
Northern Mockingbird (*Mimus polyglottos*)	0.194	0.056	0.007	0	R	G	EU	SG,GG
Brown Thrasher (*Toxostoma rufum*)	0.063	0.194	0.125	0.049	R	S	FU	GG
Cedar Waxwing (*Bombycilla cedrorum*)	0.007	0.042	0.014	0.014	M	S	N/A	CG,SG
White-eyed Vireo (*Vireo griseus*)*	0.285	0.785	0.736	0.417	R	G	GE	SG
Red-eyed Vireo (*Vireo olivaceus*)	0.049	0.021	0.076	0.257	M	F	FC	CG
Pine Warbler (*Dendroica pinus*)	0.028	0.063	0.118	0.361	R	F	FC	CG
Prairie Warbler (*Dendroica discolor*)**	0.701	0.458	0.007	0.097	M	E	EU	SG,GG
American Redstart (*Setophaga ruticilla*)	0	0.007	0.035	0.007	M	F	FC	CG
Swainson's Warbler (*Limnothlypis swainsonii*)**	0	0	0.153	0.069	M	F	FU	GG
Kentucky Warbler (*Oporornis formosus*)**	0.021	0.028	0.111	0.250	M	F	FU	GG
Common Yellowthroat (*Geothlypis trichas*)	0.813	0.715	0.153	0.236	R	E	EU	SG
Hooded Warbler (*Wilsonia citrine*)*	0.090	0.431	0.875	0.750	M	F	FU	SG
Yellow-Breasted chat (*Icteria virens*)	0.681	0.729	0.319	0.271	M	E	EU	SG
Summer Tanager (*Piranga rubra*)	0.042	0.007	0.021	0.132	M	S	FC	CG
Indigo Bunting (*Passerina cyanea*)	0.632	0.208	0.028	0.201	M	S	EU	SG
Blue Grosbeak (*Passerina caerulea*)	0.049	0.014	0	0.014	M	S	EU	GG
Northern Cardinal (*Cardinalis cardinalis*)	0.396	0.597	0.785	0.688	R	S	GE	SG,GG
Eastern Towhee (*Pipilo erthrophthalmus*)	0.806	0.917	0.854	0.819	R	E	EU	SG,GG
Bachman's Sparrow (*Aimophila aestivalis*)**	0.056	0.007	0	0.139	R	S	FU	GG
Brown-headed Cowbird (*Molothrus ater*)	0.083	0.208	0.083	0.042	R	G	GE	GG
Orchard Oriole (*Icterus spurious*)	0.194	0.063	0.007	0	M	S	FC	SG

Table 4.10. Mean relative abundance ($\bar{X}$) and standard error (SE) of migratory (2), habitat (4), nesting (5), and foraging (6) guilds by stand age based on detections during point count surveys at Ben's Creek WMA, LA, April-June 2003-2004.

	Stand Age [a]							
	1 yr		4-5 yr		7-9 yr		13-23 yr	
Guild Type	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Migratory Guild								
Resident	6.34C	0.25	8.13A [b]	0.23	7.02BC	0.22	7.79AB	0.26
Migrant	4.43A	0.18	3.74AB	0.30	3.08B	0.14	3.66AB	0.14
Habitat Guild								
Forest	0.42C	0.07	1.18B	0.11	2.88A	0.14	3.58A	0.16
Second growth/forest openings	2.49A	0.14	2.16A	0.26	2.16A	0.12	2.47A	0.12
Early-successional	6.08A	0.21	5.68A	0.19	2.63B	0.15	2.58B	0.15
Edge or generalist	1.79B	0.12	2.84A	0.12	2.42A	0.12	2.83A	0.13
Nesting Guild								
Forest canopy or subcanopy	0.62C	0.08	0.69BC	0.09	1.05B	0.10	1.79A	0.12
Forest understory or ground	0.25C	0.04	1.04B	0.10	2.40A	0.12	2.33A	0.14
Early-successional understory	7.56A	0.25	6.07B	0.21	2.67C	0.15	2.86C	0.17
Cavity	0.71B	0.09	0.71B	0.08	0.53B	0.07	1.26A	0.10
Generalist (multi-habitat)	1.63B	0.13	3.07A	0.13	3.43A	0.12	3.21A	0.15
Foraging Guild								
Canopy gleaning	0.85C	0.09	1.28BC	0.24	1.35B	0.12	2.37A	0.14
Shrub gleaning	9.07A	0.27	9.71A	0.37	7.64B	0.21	7.47B	0.28
Ground gleaning	4.97A	0.18	5.24A	0.18	4.88A	0.17	5.50A	0.20
Sallying	0.25A	0.06	0.08AB	0.02	0.10B	0.04	0.15AB	0.03
Bark probing	0.11B	0.03	0.22AB	0.04	0.14B	0.03	0.35A	0.05
Aerial foraging [c]	0.04	0.03	0.08	0.02	0.00	0.00	0.03	0.02

[a] *N*=144 surveys conducted in each stand age class during study period.
[b] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons (alpha=0.10).
[c] Insufficient data to analyze.

Nineteen of the 44 bird species met the minimum requirements of a minimum frequency of occurrence of 0.10 across all surveys and $\geq$ 50 detections. Of these 19 bird species, frequency of occurrence varied across stand age for 17 species (Table 4.11). Relative frequency of occurrence was greater in 13-23-year stands, and similar in all other stand age classes, for Carolina Wren ($F_{3,87.3}$=9.58, P<0.001), Red-eyed Vireo ($F_{3,111}$=10.94, P<0.001), and Pine Warbler ($F_{3,106}$=13.47, P<0.001). In slight contrast, relative occurrence of the Blue Jay was greater and similar in 7-9- and 13-23-year than 1-year stands, and similar in 4-5-year stands to all

three ($F_{3,88.7}$=3.74, P=0.014). An even stronger positive trend associated with stand age was apparent in response of Red-bellied Woodpecker, Wood Thrush, and Kentucky Warbler, for which relative occurrence was greater in 13-23-year stands and decreased with decreasing stand age ($F_{3,99.2}$=3.51, P=0.0.018; $F_{3,107}$=5.45, P=0.002; and $F_{3,111}$=9.29, P<0.001, respectively).

Relative frequency of occurrence of Hooded Warbler was greatest in 7-9- and 13-23-year stands, decreased in 4-5-year stands, and was lowest in 1-year stands ($F_{3,95.7}$=42.33, P<0.001). Likewise, occurrence of Northern Cardinal was lowest in 1-year, increased in 4-5-year, was even greater in 7-9-year, and was similar in 13-23-year stands to the latter 2 stand age classes ($F_{3,84.6}$=16.03, P<0.001). In contrast, frequency of occurrence of White-eyed Vireo was similar and greater in 4-5- and 7-9-year stands relative to other stand age classes, which had similar frequencies of occurrence of this species ($F_{3,84}$=18.61, P<0.001). Relative occurrence of Tufted Titmouse was greater in 13-23-year stands than 4-5- and 7-9-year stands, and similar in 1-year stands to other stand age classes ($F_{3,90.3}$=6.01, P<0.001). Alternatively, 4-5-year stands had a greater frequency of occurrence of Brown Thrasher relative to 13-23-year stands, with other stand age classes similar to both ($F_{3,91.8}$=2.78, P=0.045). Further, relative frequency of occurrence of Brown-headed Cowbird was greater in 4-5-year stands than all other stand age classes ($F_{3,89.3}$=4.94, P=0.003).

In general, the trend in occurrence of Prairie Warbler, Common Yellowthroat, Yellow-breasted Chat, and Indigo Bunting was opposite to trends in occurrence of species reported above. For both Common Yellowthroat and Yellow-breasted Chat, relative frequency of occurrence was similar and greater ($F_{3,87}$=35.42, P<0.001 and $F_{3,82.8}$=20.23, P<0.001, respectively) in 1- and 4-5-year stands than in 7-9- and 13-23-year stands. Alternatively, relative occurrence of Indigo Bunting was greatest in 1-year stands, decreased in 4-5- and 13-23-year stands, which were similar, and was lowest in 7-9-year stands ($F_{3,71.4}$=25.21, P<0.001). Finally,

among the 19 species examined, response of Prairie Warbler associated with stand age demonstrated the strongest trend. Relative frequency of occurrence was greatest in 1-year stands, decreased in 4-5-year stands, decreased further in 13-23-year stands, and was lowest in 7-9-year stands ($F_{3,97}$=32.09, P<0.001).

Table 4.11. Mean relative frequency of occurrence by stand age for bird species with ≥ 0.10 frequency of occurrence during point count surveys at Ben's Creek, LA, April-June 2003-2004.

	Stand Age[a]							
	1 yr		4-5 yr		7-9 yr		13-23 yr	
Species	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
Red-bellied Woodpecker	0.06B[b]	0.02	0.15AB	0.03	0.10AB	0.03	0.19A	0.03
Blue Jay	0.14B	0.03	0.24AB	0.04	0.33A	0.04	0.29A	0.04
Carolina Chickadee	0.08A	0.02	0.12A	0.03	0.07A	0.02	0.15A	0.03
Tufted Titmouse	0.30AB	0.04	0.24B	0.04	0.21B	0.03	0.43A	0.04
Carolina Wren	0.44B	0.04	0.55B	0.04	0.54B	0.04	0.81A	0.03
Wood Thrush	0.01C	0.01	0.06BC	0.02	0.14AB	0.03	0.22A	0.03
Brown Thrasher	0.06AB	0.02	0.19A	0.03	0.13AB	0.03	0.05B	0.02
White-eyed Vireo	0.29B	0.04	0.79A	0.03	0.74A	0.04	0.42B	0.04
Red-eyed Vireo	0.05B	0.02	0.02B	0.01	0.08B	0.02	0.26A	0.04
Pine Warbler	0.03B	0.01	0.06B	0.02	0.12B	0.03	0.36A	0.04
Prairie Warbler	0.70A	0.04	0.46B	0.04	0.01D	0.01	0.10C	0.03
Kentucky Warbler	0.02C	0.01	0.03BC	0.01	0.11AB	0.03	0.25A	0.04
Common Yellowthroat	0.81A	0.03	0.72A	0.04	0.15B	0.03	0.24B	0.04
Hooded Warbler	0.09C	0.02	0.43B	0.04	0.88A	0.03	0.75A	0.04
Yellow-breasted Chat	0.68A	0.04	0.73A	0.04	0.32B	0.04	0.27B	0.04
Indigo Bunting	0.63A	0.04	0.21B	0.03	0.03C	0.01	0.20B	0.03
Northern Cardinal	0.40C	0.04	0.60B	0.04	0.79A	0.03	0.69AB	0.04
Eastern Towhee	0.81A	0.03	0.92A	0.02	0.85A	0.03	0.82A	0.03
Brown-headed Cowbird	0.08B	0.02	0.21A	0.03	0.08B	0.02	0.04B	0.02

[a] N=144 surveys conducted in each stand age during study period.
[b] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons (alpha=0.10).

Associations of Microhabitat with Stand Age

Fourteen of 16 microhabitat variables measured at point count stations differed across stand age at Ben's Creek (Table 4.12). Tree species richness increased with increasing stand age, such that richness was greatest in 13-23-year stands and lowest in 1-year stands ($F_{3,27}$=33.84, P<0.001). Yet total number of trees was greatest in 7-9-year stands, followed by 13-23-, 4-5-, and 1-year stands ($F_{3,27}$=47.50, P<0.001). Both percentage canopy cover and conifer litter were similar and greatest in 7-9- and 13-23-year stands ($F_{3,27}$=120.81, P<0.001 and $F_{3,27}$=34.66, P<0.001, respectively), decreased in 4-5-year stands, and was lowest in 1-year stands.

Similarly, and predictably, percentage leaf litter, number of non-pine trees, and shrub species richness, were similar and greater in 13-23- and 7-9-year stands ($F_{3,27}$=13.81, P<0.001; $F_{3,27}$=17.05, P<0.001; and $F_{3,27}$=9.98, P<0.001, respectively) than in 4-5- and 1-year stands. However, total number of shrubs was greater in 7-9-year stands than all other stand age classes ($F_{3,27}$=6.19, P=0.002), and number of non-pine shrubs was greater in 7-9-year stands than 4-5- and 1-year stands ($F_{3,27}$=6.23, P=0.002).

Percentage grass cover was greater in 1- and 4-5 year stands than other stands ($F_{3,27}$=24.62, P<0.001), which was converse to the general trend of variables reported above. Likewise, percentage forb cover was greater in 1-year stands than 7-9- and 13-23-year stands, with percentage forb cover in 4-5-year stands similar to other stand age classes ($F_{3,27}$=5.87, P=0.003). Interestingly, percentage coarse woody debris was similar and greater in the youngest and oldest stand age classes than in the 2 intermediate stand age classes ($F_{3,27}$=15.34, P<0.001). In slight contrast, 1-year stands had the greatest percentage of vine cover, followed respectively by 4-5-, 13-23-, and 7-9-year stands ($F_{3,27}$=16.33, P<0.001). Finally, percentage bare ground was greater in 1-year stands than other stand age classes ($F_{3,27}$=17.44, P<0.001).

Table 4.12. Mean value ($\bar{X}$) and standard error (SE) of microhabitat variables associated with avian point count stations by stand age at Ben's Creek, LA, 2003-2004.

	Stand Age							
	1 yr		4-5 yr		7-9 yr		13-23 yr	
Variable	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE	$\bar{X}$	SE
% Canopy	15.16C	2.09	66.32B	2.28	88.74A	0.84	88.59A	1.38
% Leaf litter	5.87B	1.07	8.38B	0.83	20.30A	1.44	18.40A	2.40
% Conifer litter	4.43C	0.61	16.68B	1.46	35.54A	1.23	27.55A	1.84
% Grass cover	17.97A	1.77	15.19A	2.09	2.12B	0.51	3.02B	0.72
% Forb cover	8.90A	1.04	5.56AB	0.45	2.45B	0.36	3.37B	0.64
% Fern cover	1.63A	0.33	0.64A	0.32	1.16A	0.41	0.71A	0.22
% Woody cover	21.51A	1.70	22.50A	1.77	23.02A	1.60	21.59A	2.20
% Bare ground	8.21A	0.95	2.92B	0.49	1.67B	0.28	1.55B	0.43
% Woody debris	4.92A	0.57	1.67B	0.24	2.58B	0.36	7.17A	0.79
% Vine	29.16A	2.61	21.79AB	1.53	9.47C	1.18	18.52B	2.63
# Shrubs	236.54B	12.41	274.08B	13.93	342.50A	9.14	254.98B	11.20
# Non-pine shrubs	212.44B	12.53	247.50B	12.78	322.67A	9.15	264.29AB	9.73
Shrub richness	16.25B	0.56	17.15B	0.46	20.21A	0.43	20.54A	0.45
# Trees	3.04D	1.36	32.67C	3.71	86.67A	3.07	51.60B	3.68
# Non-pine trees	2.69B	1.28	3.13B	1.79	17.08A	4.71	20.88A	3.54
Tree richness	0.83D	0.29	1.17C	0.07	2.71B	0.27	5.23A	0.53

[a] Mean values followed by different letters across rows are different using Tukey-Kramer pairwise comparisons (alpha=0.10).

Associations of Avifauna with Microhabitat

Two of 16 microhabitat variables were highly correlated: number of shrubs was highly correlated (0.98) with number of non-pine saplings, and number of trees was highly correlated (0.83) with percentage conifer litter. This resulted in removal of number of non-pine shrubs and percentage conifer litter from the data set. Thus, PCA was performed on the 14 remaining variables. Four principal components received eigenvalues >1, accounting for 62% of the variance in the data set, and were thus retained for further analyses (Table 4.13). Component 1 was interpreted to represent areas with a partially or fully closed canopy, dense trees and a variety of trees and shrubs (Table 4.14). Number of shrubs loaded positively on component 2, whereas percentage coarse woody debris loaded negatively. Alternatively, component 3 was

interpreted to represent areas with an abundance of bare ground and occasional fern cover. Finally, woody cover was the only variable strongly correlated with component 4.

Table 4.13. Principal components (PC) analysis results for microhabitat variables surveyed along transects bisecting avian point count stations at Ben's Creek, LA.

	PC1	PC2	PC3	PC4
Eigenvalue	4.73	1.86	1.11	1.05
Variance explained	0.34	0.13	0.08	0.07
Variable:				
% Canopy cover	**78**[a.b]	25	-41	3
% Leaf litter	**81**	-3	7	-15
% Grass cover	**-69**	-1	20	-11
% Forb cover	**-49**	-33	18	-1
% Woody cover	-8	5	-14	**86**
% Vine cover	**-68**	-21	-17	-44
% Fern cover	3	12	**69**	-11
% Bare ground	-29	-30	**68**	5
% Coarse woody debris	31	**-75**	5	-4
# Trees	**73**	36	-18	9
# Non-pine trees	**59**	-8	5	-21
Tree species richness	**72**	-23	-10	-36
# Shrubs	25	**78**	4	12
Shrub species richness	**45**	43	-16	-29

[a] Correlation coefficients are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.

Table 4.14. Principal components (PC, eigenvalues $\geq$ 1) derived from microhabitat variables associated with avian point count stations at Sherburne, LA, 2003-2004. Associated variables are those with a correlation coefficient of $\geq |0.40|$ with each respective PC.

PC	Associated Variables[a]	Interpretation
PC1	% canopy, % leaf litter, # trees, # non-pine trees, tree and shrub richness, (-) % grass cover, (-) % forb cover; (-) % vine	Canopy cover, tree richness and density
PC2	# shrubs, (-) % coarse woody debris	Shrub density
PC3	% fern cover, % bare ground	Bare ground, fern cover
PC4	% woody cover	Woody ground cover

[a] Variables are positively related to the principal component unless otherwise noted.

Occurrence of each the 19 species was examined using the 4 principal components as explanatory variables. The best approximating model for occurrence of Red-bellied Woodpecker, Pine Warbler, and Brown-headed Cowbird retained component 1, which represented canopy closure, tree richness and density, and leaf litter. Occurrence of each of these species was negatively associated with component 1, which is generally supported by estimates of relative importance. Estimates of relative importance associated with occurrence of Brown-headed Cowbird suggested that fern cover and bare ground (component 3) and woody ground cover (component 4) were as important as canopy closure, tree richness and density and leaf litter in explaining variation in occurrence of Brown-headed Cowbird.

In contrast, shrub density (component 2) was the only variable retained in the best approximating model for explaining occurrence of Carolina Wren, Prairie Warbler, and Kentucky Warbler. Occurrence of Carolina Wren was negatively associated with shrub density, whereas occurrence of both Prairie Warbler and Kentucky Warbler was positively associated with this variable.

The best approximating model for occurrence of Brown Thrasher and White-eyed Vireo was comprised of component 3, which represented percentage fern cover and bare ground, and negatively influenced both species. This was the only model which received substantial empirical support in explaining occurrence of White-eyed Vireo. In contrast, relative Akaike weights suggested that shrub density (component 2) and percentage woody ground cover (component 4) also influenced occurrence of Brown Thrasher.

Percentage woody ground cover strongly and positively influenced Common Yellowthroat, Yellow-breasted Chat, and Northern Cardinal, whereas Blue Jay, Carolina Chickadee, Tufted Titmouse, Wood Thrush, and Indigo Bunting were strongly negatively influenced. However, other components also influenced occurrence of certain species.

Table 4.15. Model selection results with respect to microhabitat for bird species detected during avian point counts at Ben's Creek, WMA. Models reported are those with ΔAICc ≤ 2. K is number of model parameters, AICc equals Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and the best model, and w_i is Akaike weight.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Red-bellied Woodpecker	PC1	2	351.40	355.42	0	0.257
	PC2	2	352.82	356.84	1.42	0.126
	PC1 PC4	3	351.00	357.04	1.62	0.114
	PC4	2	353.14	357.16	1.75	0.107
	PC3	2	353.26	357.28	1.86	0.101
Blue Jay	PC4	2	528.65	532.67	0	0.282
	PC3 PC4	3	526.79	532.83	0.16	0.260
	PC3	2	530.37	534.37	1.72	0.119
Carolina Chickadee	PC4	2	307.11	311.13	0	0.183
	PC3 PC4	3	305.95	312.00	0.86	0.119
	PC2 PC4	3	306.26	312.31	1.17	0.102
	PC2	2	308.41	312.43	1.30	0.095
	PC1	2	308.70	312.72	1.59	0.083
	PC1 PC4	3	307.05	313.09	1.95	0.069
Tufted Titmouse	PC4	2	570.86	574.88	0	0.225
	PC3	2	570.94	574.96	0.09	0.215
	PC3 PC4	3	569.15	575.19	0.31	0.192
	PC1	2	571.91	575.93	1.05	0.133
	PC1 PC3	3	570.64	576.68	1.80	0.091
Carolina Wren	PC2	2	596.58	600.60	0	0.397
	PC4	2	597.14	601.16	0.56	0.299
Wood Thrush	PC4	2	203.75	207.77	0	0.318
	PC3 PC4	3	203.63	209.67	1.90	0.123
	PC1 PC4	3	203.66	209.70	1.93	0.121
Brown Thrasher	PC3	2	241.34	245.36	0	0.161
	PC2	2	241.62	245.64	0.28	0.139
	PC4	2	242.19	246.21	0.85	0.105
	PC3 PC4	3	240.57	246.61	1.25	0.086
	PC2 PC4	3	240.68	246.72	1.36	0.082
	PC2 PC3	3	240.69	246.73	1.37	0.081
	PC1 PC3	3	241.31	247.35	1.99	0.060
White-eyed Vireo	PC3	2	482.85	486.87	0	0.474

[a] PC1=canopy cover, tree richness and density, litter; PC2= shrub density; PC3= fern cover, bare ground; PC4= woody cover

Table 4.15. continued.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Red-eyed Vireo	PC1 PC2 PC4	4	178.75	186.82	0	0.325
	PC1 PC2 PC3 PC4	5	177.21	187.32	0.49	0.254
	PC1 PC4	3	181.74	187.78	0.96	0.201
Pine Warbler	PC1	2	274.23	278.25	0	0.457
	PC1 PC4	3	273.99	280.03	1.78	0.188
	PC1 PC2	3	274.11	280.15	1.90	0.177
Prairie Warbler	PC2	2	298.59	302.61	0	0.575
	PC2 PC4	3	298.49	304.53	1.92	0.220
Kentucky Warbler	PC2	2	213.40	217.42	0	0.298
	PC4	2	215.22	219.24	1.82	0.120
Common Yellowthroat	PC4	2	429.71	433.73	0	0.537
	PC3 PC4	3	429.44	435.48	1.75	0.223
Hooded Warbler	PC3 PC4	3	402.78	408.83	0	0.327
	PC3	2	404.98	409.00	0.17	0.300
	PC4	2	406.36	410.38	1.55	0.150
Yellow-breasted Chat	PC4	2	427.80	431.82	0	0.329
	PC2 PC4	3	426.55	432.59	0.77	0.224
	PC3 PC4	3	427.60	433.64	1.82	0.132
Indigo Bunting	PC4	2	277.41	281.43	0	0.211
	PC2	2	277.42	281.44	0.01	0.210
	PC3	2	277.76	281.78	0.35	0.177
Northern Cardinal	PC4	2	645.44	649.46	0	0.617
Eastern Towhee	PC2 PC4	3	314.92	320.97	0	0.308
	PC1 PC2 PC4	4	314.37	322.44	1.48	0.147
	PC1 PC2	3	316.64	322.69	1.72	0.130
	PC2	2	318.90	322.92	1.96	0.116
Brown-headed Cowbird	PC1	2	264.94	268.96	0	0.223
	PC3	2	265.62	269.64	0.68	0.158
	PC4	2	265.84	269.86	0.90	0.142
	PC2	2	266.40	270.42	1.46	0.107
	PC1 PC3	3	264.90	270.94	1.98	0.083

Percentage fern cover and bare ground (component 3) negatively influenced occurrence of Blue Jay and Carolina Chickadee, and positively influenced Tufted Titmouse and Indigo Bunting. Occurrence of Indigo Bunting also was negatively associated with shrub density (component 2).

The best approximating model for occurrence of Red-eyed Vireo retained canopy closure, tree richness and density, and percentage litter cover (component 1), shrub density (component 2), and percentage woody ground cover (component 4). Canopy closure, tree richness and density, percentage litter cover, and shrub density positively influenced occurrence of Red-eyed Vireo, whereas percentage woody ground cover had a negative influence. Conversely, occurrence of Hooded Warbler was best explained by the model which retained percentage fern cover and bare ground (component 3, negative influence) and percentage woody ground cover (component 4, positive influence). Finally, the best approximating model for the occurrence of Eastern Towhee retained shrub density (component 2) and percentage woody ground cover (component 4), both of which positively influenced occurrence of this species.

Associations of Avifauna with Landscape Characteristics

A total of 26 landscape variables (6 composition variables representing each cover class, 3 class-specific configuration variables for each of the 6 classes, distance to nearest road, and distance to nearest stream) were included in analyses of associations between occurrence of birds and landscape characteristics (Table 4.17). Streamside management zones made up the lowest proportion of the landscape (9.5 ± 0.90), followed by 24-63-year pine stands (11.30 ± 1.00), whereas 4-6-year stands comprised the largest proportion (24.00 ± 2.60), with other stand age classes of pine intermediate to these and similar in proportion to each other. Based on eigenvalues >1 and a proportion of variance > 0.05, PCA resulted in 5 principal components, which accounted for 70.9% of the variance (Table 4.18). All landscape attributes associated with

Table 4.16. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory component variable, representing microhabitat, for select birds detected during avian point count at Ben's Creek WMA, LA, 2003-2004.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Red-bellied Woodpecker	PC1	-0.0368	0.1331	0.534
	PC2	0.1837	0.1379	0.336
	PC3	-0.3238	0.1436	0.243
	PC4	-0.0694	0.1146	0.368
Blue Jay	PC1	0.3554	0.1135	0.275
	PC2	0.2132	0.1175	0.129
	PC3	-0.2736	0.1167	0.651
	PC4	-0.0864	0.0989	0.716
Carolina Chickadee	PC1	0.1714	0.1360	0.367
	PC2	-0.1207	0.1353	0.383
	PC3	-0.1608	0.1372	0.406
	PC4	-0.1867	0.1226	0.642
Tufted Titmouse	PC1	0.2150	0.1040	0.339
	PC2	-0.1697	0.1050	0.035
	PC3	0.0866	0.0976	0.556
	PC4	-0.1905	0.0967	0.543
Carolina Wren	PC1	0.2102	0.1138	0.044
	PC2	-0.0763	0.1070	0.573
	PC3	-0.2336	0.1010	0.133
	PC4	-0.0131	0.0928	0.468
Wood Thrush	PC1	0.5389	0.1872	0.368
	PC2	0.0821	0.1618	0.293
	PC3	-0.2674	0.1873	0.332
	PC4	-0.4126	0.1245	0.893
Brown Thrasher	PC1	-0.0257	0.1751	0.301
	PC2	-0.0994	0.1527	0.465
	PC3	-0.2303	0.1362	0.508
	PC4	0.2073	0.1174	0.426
White-eyed Vireo	PC1	0.1305	0.1373	0.290
	PC2	0.3716	0.1243	0.045
	PC3	-0.3742	0.1156	0.868
	PC4	0.1231	0.0996	0.287

[a] PC1=canopy cover, tree richness and density, litter; PC2= shrub density; PC3= fern cover, bare ground; PC4= woody cover

Table 4.16. continued.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Red-eyed Vireo	PC1	0.9874	0.1647	0.890
	PC2	0.1537	0.1628	0.598
	PC3	-0.1138	0.2006	0.421
	PC4	-1.1362	0.1618	0.999
Pine Warbler	PC1	-0.2554	0.1433	0.911
	PC2	0.0986	0.1469	0.307
	PC3	-0.4266	0.1656	0.051
	PC4	0.0741	0.1051	0.313
Prairie Warbler	PC1	-1.6985	0.1967	0.032
	PC2	0.4786	0.1523	0.979
	PC3	0.1334	0.1337	0.181
	PC4	0.0371	0.1069	0.277
Kentucky Warbler	PC1	0.3281	0.1700	0.250
	PC2	0.3130	0.1571	0.645
	PC3	-0.1188	0.1593	0.272
	PC4	-0.0773	0.1157	0.330
Common Yellowthroat	PC1	-1.1033	0.1515	0.000
	PC2	-0.0792	0.1313	0.219
	PC3	-0.0173	0.1190	0.306
	PC4	0.2465	0.1003	0.965
Hooded Warbler	PC1	0.8528	0.1809	0.000
	PC2	0.0726	0.1346	0.223
	PC3	-0.4140	0.1187	0.731
	PC4	-0.0808	0.1023	0.571
Yellow-breasted Chat	PC1	-1.0349	0.1476	0.000
	PC2	0.1224	0.1282	0.475
	PC3	-0.0304	0.1162	0.361
	PC4	0.2123	0.1017	0.790
Indigo Bunting	PC1	-0.9153	0.1904	0.190
	PC2	-0.3435	0.1818	0.463
	PC3	0.2001	0.1636	0.411
	PC4	-0.0330	0.1103	0.406
Northern Cardinal	PC1	0.5871	0.0988	0.000
	PC2	0.2110	0.1011	0.144
	PC3	-0.0621	0.0938	0.249
	PC4	0.0157	0.0986	0.699

Table 4.16. continued.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Eastern Towhee	PC1	-0.2963	0.1346	0.4374
	PC2	0.3675	0.1290	0.8918
	PC3	-0.0039	0.1322	0.2169
	PC4	0.2133	0.1168	0.6401
Brown-headed Cowbird	PC1	-0.4437	0.1685	0.440
	PC2	0.1517	0.1621	0.244
	PC3	-0.0480	0.1398	0.393
	PC4	0.0317	0.1217	0.354

7-9-year stands were positively loaded on component 1; thus component 1 was interpreted to represent stands which had achieved canopy closure (Table 4.19). On component 2, positive loadings included all landscape attributes associated with 4-6-year stands, whereas percentage composition, patch size, and edge density of 11-23-year stands were negatively loaded. Therefore, component 2 was interpreted to represent early successional communities with an open canopy. Component 3 included patch shape of 11-23-year stands (negative loading) and all attributes of 24-63-year stands (positive loadings), and was thus interpreted to represent stands with a pine-dominated overstory, and dense, woody understory. All landscape attributes associated with streamside management zones (SMZs) scored positively on component 4, whereas distance to nearest stream scored negatively. Finally, component 5 included all landscape attributes associated with 0-2-year stands (positive loadings) and distance to nearest road (negative loading), and thus, was interpreted to represent recently clearcut stands.

Occurrence of each the 19 species was examined using the 5 principal components as explanatory variables of the 26 landscape attributes. The best approximating model for explaining variation in occurrence of Brown Thrasher, Hooded Warbler, Yellow-breasted Chat, and Indigo Bunting retained one variable, stands with a closed canopy (component 1).

Occurrence of Brown Thrasher and Hooded Warbler increased as the proportion of stands in the landscape with a closed canopy increased, whereas occurrence of Yellow-breasted Chat and Indigo Bunting decreased. Additionally, stands with a predominately pine overstory and woody understory (component 3) and clearcuts (component 5) were equally as important in explaining occurrence of Brown Thrasher. In contrast, occurrence of Blue Jay, Carolina Chickadee, Carolina Wren, and Wood Thrush were negatively associated with pine overstory-woody understory stands (component 3). Occurrence of Blue Jay also was negatively influenced by early successional stands (component 2) and SMZs (component 4), and occurrence of Carolina Chickadee was negatively influenced by early successional communities (Table 4.21).

The best approximating model for occurrence of Tufted Titmouse, Pine Warbler, Kentucky Warbler, and Common Yellowthroat included 1 component variable, SMZs (component 4), which negatively influenced occurrence of Pine Warbler and Common Yellowthroat, and positively influenced occurrence of Tufted Titmouse and Kentucky Warbler. Additionally, stands with a closed canopy (component 1), stands with a pine-dominated overstory and woody understory (component 3), and clearcuts (component 5) also may explain variation in occurrence of Kentucky Warbler (Table 4.21).

The component which represented clearcuts (component 5) was retained in the best approximating model for occurrence of Brown-headed Cowbird, which was positively associated with this stand type. Interestingly, relative weights suggested that stands with a closed canopy (component 1) also may be important to occurrence of Brown-headed Cowbird. Conversely, the model with closed canopy stands (component 1) and SMZs (component 4) received the most substantial empirical support in explaining occurrence of Red-bellied Woodpecker. Canopy closure negatively influenced occurrence of Red-bellied Woodpecker, whereas SMZs had a

positive influence. This model also best explained occurrence of Northern Cardinal, which was positively associated with both canopy closure and SMZs.

Conversely, the best approximating model in explaining variation in occurrence of White-eyed Vireo included, canopy closure (component 1) and stands with a pine-dominated

Table 4.17. Mean values ($\bar{X}$) and standard errors (SE) of landscape variables generated within 500 m radius circular buffer zones (*n*=96) centered on avian point count stations at Ben's Creek, LA.

Variables	$\bar{X}$	SE
Landscape composition		
% Pine regenerated 2001-2003 (0-2 yr)	16.30	1.90
% Pine regenerated 1997-1999 (4-6 yr)	24.00	2.60
% Pine regenerated 1994-1996 (7-9 yr)	18.70	2.50
% Pine regenerated 1980-1992 (11-23 yr)	16.80	3.00
% Pine regenerated 1940-1979 (24-63 yr)	11.30	1.00
% Streamside management zone (hardwood)	9.50	0.90
Landscape configuration		
Median patch size, 0-2 yr (ha)	5.63	0.76
Edge density, 0-2 yr (m/ha)	0.002	0.0003
Area weighted mean shape index[a], 0-2 yr	1.19	0.08
Median patch size, 4-6 yr (ha)	8.85	1.62
Edge density, 4-6 yr (m/ha)	0.003	0.0002
Area weighted mean shape index, 4-6 yr	1.30	0.06
Median patch size, 7-9 yr (ha)	14.63	1.93
Edge density, 7-9 yr (m/ha)	0.002	0.0002
Area weighted mean shape index, 7-9 yr	0.85	0.08
Median patch size, 11-23 yr (ha)	3.62	0.80
Edge density, 11-23 yr (m/ha)	0.002	0.0004
Area weighted mean shape index, 11-23 yr	0.78	0.09
Median patch size, 24-63 yr (ha)	3.96	0.53
Edge density, 24-63 yr (m/ha)	0.002	0.0002
Area weighted mean shape index, 24-63 yr	1.21	0.08
Median patch size, hardwood (ha)	2.86	0.41
Edge density, hardwood (m/ha)	0.004	0.0003
Area weighted mean shape index, hardwood	2.58	0.11
Other landscape aspects		
Distance to stream (m)	1411.44	77.30
Distance to nearest road (m)	156.80	9.03

[a] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 4.18. Principal components (PC) analysis results for landscape variables generated within 500 m radius circular buffers centered on avian point count stations (*n*=96) at Ben's Creek WMA, LA, 2003-2004.

	PC1	PC2	PC3	PC4	PC5
Eigenvalue:	6.11	4.92	3.29	2.69	1.42
Variance explained:	0.23	0.19	0.13	0.10	0.05
Variables:					
% Pine stands regenerated 2001-2003 (0-2 yr)	-3 [a]	-4	38	13	**85** [b]
% Pine stands regenerated 1997-1999 (4-6 yr)	-13	**92**	-5	-22	-11
% Pine stands regenerated 1994-1996 (7-9 yr)	**93**	-2	-7	-12	-17
% Pine stands regenerated 1980-1992 (11-23 yr)	-50	**-57**	-41	11	-36
% Pine stands regenerated 1940-1979 (24-63 yr)	-11	-7	**84**	-14	18
% Streamside management zone (hardwood)	-30	-32	-15	**71**	-10
Median patch size, 0-2 yr (ha)	-21	6	-1	-24	**51**
Edge density, 0-2 yr (m/ha)	-4	-1	39	22	**84**
Area weighted mean shape index[c], 0-2 yr	-12	15	23	12	**71**
Median patch size, 4-6 yr (ha)	-2	**66**	-43	-5	13
Edge density, 4-6 yr (m/ha)	-3	**93**	12	-3	-7
Area weighted mean shape index, 4-6 yr	24	**59**	26	43	9
Median patch size, 7-9 yr (ha)	**93**	-2	-7	-12	-17
Edge density, 7-9 yr (m/ha)	**96**	1	-11	2	-14
Area weighted mean shape index, 7-9 yr	**83**	7	-4	8	-7
Median patch size, 11-23 yr (ha)	-41	**-47**	-14	33	-16
Edge density, 11-23 yr (m/ha)	-49	**-55**	-47	12	-35
Area weighted mean shape index, 11-23 yr	-24	-28	**-59**	23	-23
Median patch size, 24-63 yr (ha)	6	-14	**54**	-39	2
Edge density, 24-63 yr (m/ha)	-17	9	**87**	10	26
Area weighted mean shape index, 24-63 yr	-13	10	**68**	-5	14
Median patch size, hardwood (ha)	-22	-19	-5	**39**	-23
Edge density, hardwood (m/ha)	-12	-17	-8	**88**	11
Area weighted mean shape index, hardwood	6	16	-10	**77**	9
Distance to stream (m)	-42	44	18	**-60**	-7
Distance to nearest road (m)	10	13	4	29	**-38**

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.
[c] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 4.19. Principal components (PC, eigenvalues $\geq$ 1) derived from landscape variables associated with avian point count stations at Ben's Creek WMA, LA, 2003-2004. Associated variables are those with a correlation coefficient of $\geq |0.30|$ with each respective PC.

PC	Associated Variables[a,b]	Interpretation
PC1	7-9 yr [% composition, MEDPS, ED, AWMSI]	closed canopy
PC2	4-6 yr [% composition, MEDPS, ED, AWMSI]; (-) 11-23 yr [% composition, MEDPS, ED]	early successional plant communities, open canopy
PC3	24-63 yr [% composition, MEDPS, ED, AWMSI]; (-) 11-23 yr [AWMSI]	prominent pine overstory, dense woody understory
PC4	SMZ [% composition, MEDPS, ED, AWMSI]; (-) distance to nearest stream	SMZs
PC5	0-2 yr [% composition, MEDPS, ED, AWMSI]; (-) distance to nearest road	clearcuts

[a] Variables are positively related to the principal component unless otherwise noted.
[b] SMZ: streamside management zone, MEDPS=median patch size, ED=edge density, AWMSI=area weighted mean patch shape.

overstory and woody understory (component 3). Similar to occurrence of Northern Cardinal, stands with a closed canopy positively influenced occurrence of White-eyed Vireo, whereas stands with a pine-dominated overstory and woody understory had a negative influence. Alternatively, the best approximating model for occurrence of Red-eyed Vireo retained stands with a pine-dominated overstory and woody understory (component 3) and SMZs (component 4). Like occurrence of White-eyed Vireo, occurrence of Red-eyed Vireo was negatively associated with stands with a pine-dominated overstory and dense, woody understory. However, SMZs positively influenced occurrence of Red-eyed Vireo, and had the greatest relative Akaike weight, indicating that influence of SMZs on occurrence of Red-eyed Vireo may be most important.

Early successional stands (component 2) and SMZs (component 4) were retained in the model with the greatest support for occurrence of Prairie Warbler and Eastern Towhee, which was positively associated with early successional stands, and negatively associated with SMZs.

Table 4.20. Model selection results with respect to landscape characteristics for species detected during avian point counts at Ben's Creek WMA, LA. Models reported have a ΔAICc ≤ 2. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and best model, and w_i is Akaike weight.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Red-bellied Woodpecker	PC1 PC4	3	343.51	349.55	0	0.357
	PC4	2	346.47	350.50	0.94	0.222
	PC1 PC2 PC4	4	343.27	351.34	1.79	0.146
Blue Jay	PC3	2	530.03	534.05	0	0.184
	PC4	2	530.61	534.63	0.58	0.138
	PC2	2	530.98	535.00	0.95	0.114
	PC3 PC4	3	529.77	535.81	1.76	0.077
	PC2 PC3	3	529.89	535.93	1.88	0.072
Carolina Chickadee	PC3	2	309.66	313.69	0	0.227
	PC2	2	309.78	313.80	0.12	0.214
	PC5	2	310.50	314.52	0.84	0.149
Tufted Titmouse	PC4	2	567.36	571.38	0	0.387
	PC4 PC5	3	566.84	572.88	1.50	0.182
	PC3 PC4	3	567.14	573.18	1.81	0.156
Carolina Wren	PC3	2	595.10	598.12	0	0.570
Wood Thrush	PC3	2	204.81	208.84	0	0.167
	PC3 PC5	3	203.81	209.43	0.59	0.125
	PC2 PC3 PC5	4	201.62	209.69	0.86	0.109
	PC2 PC3	3	204.32	210.36	1.53	0.078
	PC3 PC4 PC5	4	202.75	210.82	1.98	0.062
Brown Thrasher	PC1	2	243.20	247.22	0	0.143
	PC5	2	243.22	247.24	0.03	0.142
	PC3	2	243.26	247.28	0.06	0.139
	PC2	2	243.92	247.94	0.72	0.100
	PC3 PC5	3	242.97	249.01	1.79	0.058
White-eyed Vireo	PC1 PC3	3	477.58	483.62	0	0.575
Prairie Warbler	PC2 PC4	3	299.57	305.62	0	0.423
	PC2 PC3 PC4	4	298.62	306.69	1.08	0.247

[a] PC1: stands with a closed canopy; PC2: early successional stands; PC3: stands with pine overstory and dense, woody understory; PC4: streamside management zones; PC5: clearcuts.

Table 4.20. continued.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Kentucky Warbler	PC4	2	211.16	215.18	0	0.128
	PC1 PC4	3	209.48	215.52	0.34	0.109
	PC3 PC5	3	209.62	215.66	0.48	0.101
	PC5	2	211.68	215.71	0.52	0.099
	PC1 PC5	3	210.10	216.14	0.96	0.080
	PC1 PC3 PC5	4	208.37	216.44	1.26	0.068
	PC4 PC5	3	210.97	217.02	1.83	0.051
	PC1 PC4 PC5	4	209.06	217.13	1.94	0.049
	PC3 PC4 PC5	4	209.07	217.14	1.95	0.048
Common Yellowthroat	PC4	2	434.36	438.38	0	0.463
	PC1 PC4	3	434.12	440.17	1.78	0.190
Hooded Warbler	PC1	2	402.57	406.59	0	0.341
	PC1 PC3	3	402.10	408.14	1.55	0.157
Yellow-breasted Chat	PC1	2	427.37	431.39	0	0.407
	PC1 PC4	3	426.47	432.51	1.11	0.233
Indigo Bunting	PC1	2	273.62	277.64	0	0.311
	PC1 PC2	3	273.40	279.44	1.80	0.126
Northern Cardinal	PC1 PC4	3	637.52	643.57	0	0.335
	PC4	2	640.20	644.22	0.66	0.241
Eastern Towhee	PC2 PC4	3	319.54	325.59	0	0.168
	PC2 PC3 PC4 PC5	5	316.84	326.94	1.36	0.085
	PC2 PC3 PC4	4	318.93	327.00	1.41	0.083
	PC2 PC4 PC5	4	319.06	327.13	1.54	0.078
	PC1 PC2 PC4	4	319.36	327.43	1.85	0.067
	PC2 PC3	3	321.44	327.48	1.90	0.065
Brown-headed Cowbird	PC5	2	264.81	268.83	0	0.154
	PC1	2	265.04	269.06	0.23	0.137
	PC4	2	266.11	270.13	1.29	0.080
	PC1 PC5	3	264.10	270.14	1.30	0.080
	PC3	2	266.68	270.70	1.87	0.060
Red-eyed Vireo	PC3 PC4	3	193.93	199.98	0	0.285
	PC2 PC3 PC4	4	192.45	200.52	0.54	0.217
Pine Warbler	PC4	2	280.20	284.22	0	0.410

Table 4.21. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory component variable, representing landscape, for select birds detected during avian point counts at Ben's Creek WMA, LA, 2003-2004.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Red-bellied Woodpecker	PC1	-0.2562	0.1460	0.623
	PC2	-0.0355	0.1414	0.291
	PC3	-0.1193	0.1306	0.160
	PC4	0.3937	0.1402	0.965
	PC5	-0.4114	0.1342	0.003
Blue Jay	PC1	0.1071	0.1199	0.192
	PC2	-0.0925	0.1204	0.354
	PC3	-0.1449	0.1147	0.500
	PC4	-0.0368	0.1152	0.394
	PC5	-0.1071	0.1173	0.171
Carolina Chickadee	PC1	-0.3283	0.1547	0.015
	PC2	-0.0075	0.1469	0.414
	PC3	-0.0078	0.1408	0.452
	PC4	0.2135	0.1424	0.177
	PC5	-0.1587	0.1431	0.317
Tufted Titmouse	PC1	-0.3391	0.1079	0.000
	PC2	-0.2599	0.1108	0.073
	PC3	-0.0181	0.1060	0.313
	PC4	0.3886	0.1094	0.881
	PC5	-0.1351	0.1068	0.397
Carolina Wren	PC1	-0.3635	0.1232	0.023
	PC2	-0.2681	0.1221	0.034
	PC3	0.0574	0.1122	0.779
	PC4	-0.1132	0.1184	0.293
	PC5	-0.3232	0.1150	0.131
Wood Thrush	PC1	-0.1832	0.2156	0.292
	PC2	-0.5540	0.2096	0.390
	PC3	-0.7461	0.2147	0.932
	PC4	-0.1055	0.2034	0.292
	PC5	-0.4970	0.1948	0.491

[a] PC1: stands with a closed canopy; PC2: early successional stands; PC3: stands with pine overstory and dense, woody understory; PC4: streamside management zones; PC5: clearcuts.

Table 4.21. continued.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Brown Thrasher	PC1	0.0174	0.1943	0.378
	PC2	0.1065	0.1827	0.280
	PC3	0.1622	0.1526	0.379
	PC4	-0.3388	0.1643	0.137
	PC5	0.0028	0.1787	0.377
White-eyed Vireo	PC1	0.5069	0.1638	0.749
	PC2	0.3880	0.1537	0.001
	PC3	-0.4881	0.1420	0.924
	PC4	-0.4892	0.1445	0.042
	PC5	-0.4411	0.1432	0.122
Red-eyed Vireo	PC1	-0.6625	0.2360	0.224
	PC2	-0.4533	0.2382	0.442
	PC3	-0.5274	0.1941	0.766
	PC4	1.6299	0.2048	1.000
	PC5	-0.0886	0.1997	0.181
Pine Warbler	PC1	-0.4956	0.1972	0.128
	PC2	-0.8606	0.1782	0.165
	PC3	-0.1855	0.1835	0.259
	PC4	-0.2178	0.1628	0.675
	PC5	-0.4586	0.1748	0.181
Prairie Warbler	PC1	-0.3832	0.2709	0.196
	PC2	0.8264	0.2286	0.871
	PC3	0.4942	0.1411	0.317
	PC4	-0.6365	0.2158	0.988
	PC5	0.0844	0.1754	0.087
Kentucky Warbler	PC1	-0.4605	0.2192	0.468
	PC2	-0.3245	0.2079	0.086
	PC3	-0.2909	0.2616	0.406
	PC4	0.3426	0.1954	0.554
	PC5	-0.8584	0.2081	0.577
Common Yellowthroat	PC1	-0.4818	0.1960	0.390
	PC2	0.6316	0.1771	0.052
	PC3	0.2945	0.1517	0.134
	PC4	-0.2580	0.1587	0.783
	PC5	0.6659	0.1900	0.023

Table 4.21. continued.

Species	Model variable	Estimate	Standard error	$\sum w_i$
Hooded Warbler	PC1	0.6657	0.2039	0.810
	PC2	-0.3823	0.1894	0.211
	PC3	-0.8351	0.2315	0.411
	PC4	-0.1286	0.1660	0.288
	PC5	-1.1673	0.1580	0.128
Yellow-breasted Chat	PC1	-0.4202	0.1772	0.837
	PC2	0.6326	0.1624	0.004
	PC3	0.3349	0.1440	0.250
	PC4	-0.3725	0.1488	0.451
	PC5	0.4368	0.1624	0.019
Indigo Bunting	PC1	-0.6350	0.2535	0.753
	PC2	0.1608	0.2175	0.357
	PC3	0.1799	0.1650	0.233
	PC4	0.2113	0.1791	0.280
	PC5	0.4270	0.1878	0.032
Northern Cardinal	PC1	0.2588	0.1126	0.634
	PC2	-0.0140	0.1076	0.277
	PC3	-0.2541	0.1015	0.038
	PC4	0.2011	0.1054	0.809
	PC5	-0.4440	0.0999	0.000
Eastern Towhee	PC1	0.0145	0.1640	0.283
	PC2	0.4219	0.1715	0.757
	PC3	-0.3261	0.1568	0.512
	PC4	-0.5229	0.1487	0.689
	PC5	-0.1177	0.1486	0.381
Brown-headed Cowbird	PC1	0.1453	0.1776	0.450
	PC2	0.4936	0.1693	0.218
	PC3	0.1615	0.1504	0.274
	PC4	-0.1104	0.1611	0.298
	PC5	0.1272	0.1587	0.514

RESULTS: SANDY HOLLOW WMA

Summary of Avifauna at Sandy Hollow

We detected 4,808 individuals (45 species) within 50 m radius circular plots during April, May, and June of 2003 and 2004 (432 point surveys), excluding wading, nocturnal and incidental (<3 detections) species, and flyovers (Table 4.22). Of the 45 species, 16 species (indicated in Table 4.22) were selected for analysis of associations with microhabitat (Table 4.23) and landscape (Table 4.28) variables.

Table 4.22. Relative frequency of occurrence across study area of species detected within 50 m radius circular plots during point count surveys at Sandy Hollow WMA, LA, April-June 2003-2004. Excludes flyovers, wading, nocturnal and incidental species ($\leq$3 detections). Species of concern indicated with asterisks:* (LA Audubon watchlist); ** (NA Audubon watchlist).

Species	Frequency of Occurrence
Northern Bobwhite (*Colinus virginianus*)*	0.144
Mourning Dove (*Zenaida macroura*)	0.037
Yellow-billed Cuckoo (*Coccyzus americanus*)*	0.032
Common Nighthawk (*Chordeiles minor)*	0.032
Chimney Swift (*Chaetura pelagica*)	0.007
Red-headed Woodpecker (*Melanerpes erythrocephalus*)	0.044
Red-bellied Woodpecker (*Melanerpes carolinus*) †	0.262
Downy Woodpecker (*Picoides pubescens*)	0.067
Northern Flicker (*Colaptes auratus)*	0.014
Pileated Woodpecker (*Dryocopus pileatus*)	0.016
Eastern Wood-pewee (*Contopus virens*) †	0.102
Great-crested Flycatcher (*Myiarchus crinitus*)	0.079
Eastern Kingbird (*Tyrannus tyrannus*)	0.090
Blue Jay (*Cyanocitta cristata*)	0.433
American Crow (*Corvus brachyrhynchos*)	0.120
Carolina Chickadee (*Poecile carolinensis*) †	0.271
Tufted Titmouse (*Baeolophus bicolor*) †	0.586
Brown-headed Nuthatch (*Sitta pusilla*)**	0.088
Sedge Wren (*Cistothorus platensis*)	0.007
Carolina Wren (*Thryothorus ludovicianus*) †	0.597
Blue-gray Gnatcatcher (*Polioptila caerula)* †	0.144
Eastern Bluebird (*Sialia sialis*)	0.039
Wood Thrush (*Hylocichla mustelina*)**	0.067
Gray Catbird (*Dumetella carolinensis*)	0.030

[a] Number of surveys within stand age in which species was detected, divided by the total number of surveys conducted on study area (n=432).
† Indicates species selected for analysis of associations with microhabitat and landscape.

Table 4.22. continued.

Species	Frequency of Occurrence
Northern Mockingbird (*Mimus polyglottos*)	0.118
Brown Thrasher (*Toxostoma rufum*) †	0.134
White-eyed Vireo (*Vireo griseus*)*	0.076
Red-eyed Vireo (*Vireo olivaceus*)	0.019
Pine Warbler (*Dendroica pinus*) †	0.590
Prairie Warbler (*Dendroica discolor*)**	0.009
Kentucky Warbler (*Oporornis formosus*)**	0.039
Common Yellowthroat (*Geothlypis trichas*) †	0.169
Hooded Warbler (*Wilsonia citrine*)*	0.083
Yellow-breasted Chat (*Icteria virens*) †	0.354
Summer Tanager (*Piranga rubra*) †	0.123
Indigo Bunting (*Passerina cyanea*) †	0.502
Blue Grosbeak (*Passerina caerulea*)	0.076
Northern Cardinal (*Cardinalis cardinalis*) †	0.590
Eastern Towhee (*Pipilo erthrophthalmus*) †	0.692
Bachman's Sparrow (*Aimophila aestivalis*)** †	0.271
Chipping Sparrow (*Spizella passerina*)	0.028
Field Sparrow (*Spizella pusilla*)	0.030
Savannah Sparrow (*Passerculus sandwichensis*)	0.009
Brown-headed Cowbird (*Molothrus ater*) †	0.289
Orchard Oriole (*Icterus spurious*)	0.021

Associations of Avifauna with Microhabitat

Thirteen microhabitat variables associated with avian point stations were considered in analysis of associations with occurrence of select bird species (Table 4.23). None of these microhabitat variables were highly correlated (>0.80). Thus, PCA was performed with all variables included. This resulted in 5 principal components with eigenvalues >1.0, which accounted for 77.9% of the variance within the data set (Table 4.24). Occurrence of each the 16 species was examined using the 5 principal components as explanatory variables.

Percentage leaf litter, tree species richness, and percentage hardwood trees loaded positively on component 1 (Table 4.25), which was interpreted to represent tree density. Alternatively, percentage canopy closure and conifer litter, and number of trees loaded positively on component 2, which was interpreted to represent canopy closure. High loadings on

Table 4.23. Mean value ($\bar{X}$) and standard error (SE) of microhabitat variables surveyed at point count stations (*n*=72) at Sandy Hollow WMA, LA, in 2003 and 2004.

Microhabitat Variable	$\bar{X}$	SE
% Canopy closure	46.34	1.48
% Leaf litter	5.60	0.54
% Conifer litter	18.12	0.77
% Bare ground	12.66	0.92
% Herbaceous cover	38.06	1.38
% Woody cover (2004 only)	14.74	0.69
# Shrubs	137.46	7.49
Shrub species richness	13.67	0.32
# Trees	24.88	1.36
Tree species richness	3.51	0.18
% Hardwood trees	21.18	1.91
% Longleaf pine (*Pinus palustris*)	58.76	3.10
% Loblolly pine (*Pinus taeda*)	15.88	2.01

Table 4.24. Principal components (PC) analysis results for microhabitat variables surveyed at avian point count stations (*n*=72) at Sandy Hollow WMA, LA in 2003 and 2004.

	PC1	PC2	PC3	PC4	PC5
Eigenvalue:	3.98	2.40	1.43	1.22	1.09
Variance explained:	0.31	0.18	0.11	0.09	0.08
Variables:					
% Canopy closure	35 [a]	**77** [b]	20	3	-3
% Leaf litter	**81**	19	20	-2	-7
% Conifer litter	-15	**83**	5	-9	10
% Bare ground	-7	-11	-6	4	**93**
% Herbaceous cover	-33	-50	-35	-15	**-58**
% Woody cover (2004 only)	-3	-7	**67**	38	-5
# Shrubs	19	3	**85**	3	8
Shrub species richness	50	6	**68**	3	0
# Trees	6	**74**	-21	0	-15
Tree species richness	**86**	18	13	6	-5
% Hardwood trees	**73**	-34	6	28	26
% Longleaf pine (*Pinus palustris*)	-47	20	-17	**-80**	-14
% Loblolly pine (*Pinus taeda*)	-5	8	13	**93**	2

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.

Table 4.25. Principal components (PC, eigenvalues ≥ 1) derived from microhabitat variables associated with avian point count stations at Sandy Hollow WMA, LA, 2003-2004. Associated variables are those with a correlation coefficient of $\geq |0.40|$ with each respective PC.

PC	Associated Variables[a]	Interpretation
PC1	% Leaf litter; Tree richness; % hardwood trees	Tree density
PC2	% Canopy closure; % Other litter; # Trees	Canopy closure
PC3	% Woody ground cover; # Shrubs; Shrub richness	Percentage shrub cover
PC4	% Loblolly pine trees; (-) % Longleaf pine trees	Percentage loblolly pine
PC5	% Bare ground; (-) % Herbaceous cover	Percentage bare ground

[a] Variables are positively related to the principal component unless otherwise noted.

component 3 included percentage woody ground cover, number of shrubs, and shrub species richness, each positive a loading. Therefore, component 3 was interpreted to represent shrub cover. Percentage loblolly pine trees scored positively on component 4, whereas percentage longleaf pine scored negatively. Finally, component 5 included 2 strong correlations, percentage bare ground (positively loaded) and herbaceous cover (negatively loaded).

The best approximating model in explaining occurrence of Indigo Bunting retained 1 component, tree density (component 1), which negatively influenced occurrence of Indigo Bunting and received the greatest relative weight among components. Alternatively, occurrence of Brown-headed Cowbird was negatively influenced by canopy closure (component 2), the single variable retained in the only model with substantial empirical support in explaining variation in occurrence of this species. Relative Akaike weights further indicated that canopy closure was the most important component variable associated with occurrence of Brown-headed Cowbird.

Canopy closure (component 2) also was important to occurrence of Carolina Chickadee and Brown Thrasher, but percentage shrub cover (component 3) influenced occurrence of these 2

species as well. Both canopy closure and percentage shrub cover negatively influenced occurrence of Carolina Chickadee and Brown Thrasher, but model-averaged Akaike weights indicate that influence of canopy closure on occurrence may be more important for both species.

Tree density (component 1), canopy closure (component 2) and percentage shrub cover (component 3) were each important to occurrence of Eastern Towhee. Varying combinations of these 3 component variables were retained in the set of candidate models for this species. Occurrence of Eastern Towhee was positively associated with tree density and percentage shrub cover, and negatively associated with canopy closure. However, as with Carolina Chickadee and brown thrasher, influence of canopy closure may be more important than other components on occurrence Eastern Towhee.

In contrast, percentage loblolly pine (component 4) gained the most support in explaining occurrence of Northern Cardinal, which was positively associated with percentage loblolly pine. Occurrence of Red-bellied Woodpecker also was positively associated with percentage loblolly pine, which was retained in the best approximating model for this species. However, percentage shrub cover (component 3) also appeared in the candidate set of models, and was further supported by Akaike weights as having some importance in explaining variation in occurrence of Red-bellied Woodpecker, which was positively associated with percentage shrub cover.

Although all components appeared across the set of candidate models for occurrence of Pine Warbler, the best approximating model included tree density (component 1), canopy closure (component 2), percentage shrub cover (component 3), and percentage loblolly pine (component 4). Occurrence of Pine Warbler increased with increasing canopy closure and percentage loblolly pine, but decreased with increasing tree density and percentage shrub cover. Relative Akaike weights suggested that tree density and canopy closure may be more important than percentage shrub cover and loblolly pine.

Occurrence of Summer Tanager was negatively influenced by percentage shrub cover (component 3), yet positively influenced by herbaceous cover (component 5), which may be more important (Table 4.27). Conversely, percentage loblolly pine (component 4) and herbaceous cover (component 5) were similarly important in explaining variation in occurrence of Eastern Wood-pewee, which was negatively influenced by both components.

Percentage herbaceous cover (component 5) was in the model with substantial empirical support in explaining occurrence of Tufted Titmouse, which decreased with increasing percentage herbaceous cover. In contrast, tree density (component 1) and percentage shrub cover (component 3), loblolly pine (component 4), and herbaceous cover (component 5) were important in explaining occurrence of both Yellow-breasted Chat and Bachman's Sparrow. Occurrence of Yellow-breasted Chat was positively associated with tree density and shrub cover, and negatively associated with loblolly pine and herbaceous cover, among which tree density and shrub cover and loblolly pine were more important. In contrast, occurrence of Bachman's Sparrow was negatively influenced by all 4 components, each equally important in explaining variation in occurrence of Bachman's Sparrow (Table 4.27).

Finally, canopy closure (component 2) and percentage shrub cover (component 3), loblolly pine (component 4), and herbaceous cover (component 5) influenced occurrence of Carolina Wren, Blue-gray Gnatcatcher, and Common Yellowthroat. Occurrence of Carolina Wren was positively associated with all 4 component variables, among which percentage shrub cover was most important. Occurrence of Blue-gray Gnatcatcher decreased with increasing canopy closure, and increased with increasing percentage shrub cover, loblolly pine, and herbaceous cover. Similarly, occurrence of Common Yellowthroat decreased with increasing canopy closure, and decreasing percentage shrub cover, loblolly pine, and herbaceous cover, among which shrub and herbaceous cover may be more important (Table 4.27).

Table 4.26. Model selection results with respect to microhabitat for species detected during avian point counts at Sandy Hollow WMA, LA, 2003-2004. Models reported are those with ΔAICc ≤ 2. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and best model, w_i is Akaike weight.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Red-bellied Woodpecker	PC4	2	427.46	431.49	0	0.180
	PC3 PC4	3	427.02	433.07	1.58	0.082
	PC3	2	429.07	433.10	1.61	0.079
Eastern Wood-pewee	PC4 PC5	3	196.46	202.52	0	0.139
	PC4	2	199.25	203.28	0.77	0.095
	PC5	2	199.31	203.34	0.82	0.092
	PC2 PC4 PC5	4	195.71	203.81	1.29	0.073
	PC3 PC4 PC5	4	196.06	204.15	1.64	0.061
	PC3 PC4	3	198.36	204.41	1.90	0.054
	PC2 PC4	3	198.46	204.51	2.00	0.051
Carolina Chickadee	PC2 PC3	3	425.68	431.74	0	0.186
	PC2	2	427.72	431.75	0.02	0.185
	PC2 PC3 PC4	4	425.27	433.36	1.63	0.082
	PC2 PC4	3	427.35	433.41	1.67	0.081
	PC1 PC2 PC3	4	425.40	433.49	1.76	0.077
	PC1 PC2	3	427.55	433.60	1.87	0.073
Tufted Titmouse	PC5	2	414.46	418.49	0	0.348
Carolina Wren	PC3 PC4	3	429.04	435.10	0	0.115
	PC3	2	431.23	435.26	0.16	0.106
	PC2 PC3	3	429.28	435.34	0.24	0.102
	PC3 PC4 PC5	4	427.57	435.66	0.57	0.086
	PC2 PC3 PC4	4	427.75	435.84	0.75	0.079
	PC3 PC5	3	429.98	436.04	0.94	0.072
	PC2 PC3 PC5	4	427.97	436.07	0.97	0.071
	PC2 PC3 PC4 PC5	5	426.23	436.37	1.27	0.061
Blue-gray Gnatcatcher	PC3 PC4 PC5	4	239.70	247.80	0	0.218
	P2 PC3 PC4 PC5	5	238.52	248.66	0.86	0.142
	PC3 PC4	3	243.01	249.07	1.27	0.115
Brown Thrasher	PC2	2	245.79	249.82	0	0.180
	PC2 PC3	3	244.65	250.71	0.89	0.115
	PC2 PC4	3	245.71	251.77	1.95	0.068

[a] PC1: tree density; PC2: canopy closure; PC3: percentage shrub cover; PC4: percentage *Pinus taeda* trees; PC5: percentage herbaceous cover.

Table 4.26. continued.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Pine Warbler	PC1 PC2 PC3 PC4	5	426.90	437.04	0	0.125
	PC1 PC2	3	431.23	437.28	0.24	0.110
	PC1 PC2 PC3	4	429.21	437.30	0.27	0.109
	PC1 PC2 PC4	4	429.73	437.82	0.78	0.084
	PC1 PC2 PC3 PC5	5	427.81	437.95	0.91	0.078
	PC1 PC2 PC3 PC4PC5	6	426.00	438.20	1.16	0.070
	PC1 PC2 PC5	4	430.44	438.53	1.49	0.059
Common Yellowthroat	PC2 PC3 PC4 PC5	5	192.26	202.40	0	0.463
	PC2 PC3 PC5	4	195.26	203.35	0.95	0.288
Yellow-breasted Chat	PC1 PC3 PC4	4	331.07	339.17	0	0.272
	PC1 PC3 PC4 PC5	5	329.45	339.59	0.43	0.219
	PC3 PC4	3	339.80	339.86	0.69	0.193
	PC3 PC4 PC5	4	331.88	339.97	0.81	0.181
Summer Tanager	PC5	2	207.87	211.90	0	0.170
	PC3 PC5	3	207.04	213.09	1.19	0.094
	PC4 PC5	3	207.09	213.14	1.24	0.091
	PC3	2	209.61	213.64	1.74	0.071
Indigo Bunting	PC1	2	278.16	282.19	0	0.187
	PC1 PC2	3	277.57	283.63	1.44	0.091
	PC2	2	279.94	283.97	1.78	0.077
	PC1 PC5	3	278.11	284.17	1.98	0.070
	PC1 PC3	3	278.13	284.19	1.99	0.069
Northern Cardinal	PC4	2	424.04	428.07	0	0.323
	PC2 PC4	3	423.66	429.72	1.65	0.142
	PC1 PC4	3	423.95	430.01	1.94	0.123
Eastern Towhee	PC1 PC2	3	343.97	350.02	0	0.128
	PC2	2	346.12	350.15	0.13	0.120
	PC1 PC2 PC3	4	342.26	350.36	0.33	0.109
	PC2 PC3	3	344.62	350.67	0.65	0.093
Bachman's Sparrow	PC1 PC3 PC4 PC5	5	373.37	383.51	0	0.366
	PC1 PC4 PC5	4	376.49	384.59	1.07	0.214
	PC1 PC2 PC3 PC4PC5	6	372.72	384.92	1.40	0.181
Brown-headed Cowbird	PC2	2	385.97	390.00	0	0.915

Table 4.27. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory component variable, representing microhabitat, for select birds detected during avian point counts at Sandy Hollow WMA, LA, 2003-2004.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Red-bellied Woodpecker	PC1	0.1615	0.1104	0.265
	PC2	0.0427	0.1096	0.313
	PC3	0.1934	0.1192	0.437
	PC4	0.1660	0.1119	0.660
	PC5	0.3140	0.1196	0.279
Eastern Wood-pewee	PC1	-0.3555	0.1682	0.181
	PC2	0.1740	0.1391	0.386
	PC3	0.0593	0.1560	0.352
	PC4	-0.2800	0.1573	0.661
	PC5	-0.2467	0.1641	0.604
Carolina Chickadee	PC1	-0.0150	0.1153	0.299
	PC2	-0.2087	0.1095	0.930
	PC3	-0.0786	0.1203	0.531
	PC4	0.1037	0.1125	0.306
	PC5	-0.0606	0.1190	0.179
Tufted Titmouse	PC1	0.1087	0.1137	0.263
	PC2	0.0753	0.1057	0.297
	PC3	0.3078	0.1241	0.008
	PC4	0.1274	0.1185	0.190
	PC5	-0.1064	0.1208	0.695
Carolina Wren	PC1	-0.0367	0.1096	0.287
	PC2	0.2033	0.1038	0.447
	PC3	0.2352	0.1123	0.965
	PC4	0.1467	0.1112	0.503
	PC5	0.0953	0.1110	0.416
Blue-gray Gnatcatcher	PC1	0.1516	0.1388	0.128
	PC2	-0.1496	0.1394	0.407
	PC3	0.3961	0.1536	0.742
	PC4	0.1413	0.1431	0.811
	PC5	0.2903	0.1731	0.617

[a] PC1: tree density; PC2: canopy closure; PC3: percentage shrub cover; PC4: percentage *Pinus taeda* trees; PC5: percentage herbaceous cover.

Table 4.27. continued.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Brown Thrasher	PC1	-0.1007	0.1572	0.266
	PC2	-0.4178	0.1324	0.761
	PC3	-0.1856	0.1553	0.450
	PC4	0.0795	0.1498	0.294
	PC5	-0.0113	0.1395	0.296
Pine Warbler	PC1	-0.1652	0.1156	0.808
	PC2	0.2888	0.1092	0.856
	PC3	-0.1201	0.1176	0.576
	PC4	0.2418	0.1260	0.477
	PC5	-0.0551	0.1214	0.365
Common Yellowthroat	PC1	-0.3840	0.1852	0.148
	PC2	-0.4927	0.1383	0.903
	PC3	0.4617	0.1550	0.981
	PC4	0.3523	0.1509	0.608
	PC5	0.4699	0.1403	0.989
Yellow-breasted Chat	PC1	0.1141	0.1231	0.560
	PC2	-0.1881	0.1181	0.073
	PC3	0.4127	0.1292	0.936
	PC4	-0.3582	0.1345	0.989
	PC5	-0.1714	0.1400	0.473
Summer Tanager	PC1	0.1052	0.1321	0.202
	PC2	-0.0112	0.1286	0.289
	PC3	-0.1839	0.1500	0.423
	PC4	0.1289	0.1416	0.378
	PC5	0.2733	0.1493	0.681
Indigo Bunting	PC1	-0.1142	0.1267	0.634
	PC2	-0.2238	0.1216	0.389
	PC3	-0.0411	0.1363	0.314
	PC4	0.1982	0.1406	0.195
	PC5	-0.0164	0.1396	0.314
Northern Cardinal	PC1	0.0727	0.1070	0.287
	PC2	-0.0417	0.1031	0.310
	PC3	0.1965	0.1129	0.135
	PC4	0.2471	0.1136	0.929
	PC5	-0.0633	0.1123	0.222

Table 4.27. continued.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Eastern Towhee	PC1	0.0985	0.1269	0.540
	PC2	-0.2284	0.1212	0.783
	PC3	0.1150	0.1355	0.481
	PC4	-0.0315	0.1296	0.272
	PC5	0.0600	0.1425	0.223
Bachman's Sparrow	PC1	-0.2671	0.1284	0.893
	PC2	-0.0975	0.1197	0.324
	PC3	-0.1917	0.1421	0.622
	PC4	-0.5566	0.1487	0.980
	PC5	-0.4481	0.1417	0.972
Brown-headed Cowbird	PC1	-0.2617	0.1209	0.031
	PC2	-0.2561	0.1096	0.933
	PC3	0.0396	0.1207	0.038
	PC4	0.0648	0.1163	0.030
	PC5	0.1500	0.1260	0.034

Associations of Avifauna with Landscape Characteristics

A total of 22 landscape variables (5 composition variables representing each cover class, 3 class-specific configuration variables for each of the 5 classes, distance to nearest road, and distance to nearest body of water [pond or stream]) were included in analyses of associations of occurrence of birds with landscape characteristics (Table 4.28). Based on eigenvalues >1 and a proportion of variance > 0.05, PCA resulted in 5 principal components, which accounted for 86.7 % of the variance (Table 4.29). Occurrence of each the 16 avian species was examined using the 5 principal components as explanatory variables, representative of landscape characteristics.

Landscape attributes associated with openings positively loaded onto component 1, which was thus interpreted to represent openings (Table 4.30). Likewise, component 2 was characterized by high positive scores associated with landscape attributes of pine forests. On component 3, landscape attributes related to mixed pine-hardwood forests were positively

loaded, whereas distance to nearest road was negatively loaded. Further, each attribute associated with longleaf pine, as well as amount of edge and shape of longleaf savannah, were highly and positively loaded onto component 4. Finally, component 5 was interpreted to represent occurrence and size of patches of longleaf savannah.

Table 4.28. Mean values ($\bar{X}$) and standard errors (SE) of landscape variables generated within 500 m radius circular buffer zones centered on avian point count stations (n=72) at Sandy Hollow WMA, LA.

Variables	$\bar{X}$	SE
Landscape composition		
% Longleaf pine	25.84	1.21
% Longleaf savannah	41.83	1.12
% Mixed pine-hardwood forest	17.82	1.52
% Pine forest	7.69	1.10
% Openings	6.82	0.77
Landscape configuration		
Median patch size, longleaf pine (ha)	20.20	0.94
Edge density, longleaf pine (m/ha)	0.009	0.0003
Area weighted mean shape index[a], longleaf pine	4.52	0.09
Median patch size, longleaf savannah (ha)	32.70	0.88
Edge density, longleaf savannah (m/ha)	0.011	0.0003
Area weighted mean shape index, longleaf savannah	4.17	0.08
Median patch size, mixed pine-hardwood (ha)	13.92	1.19
Edge density, mixed pine-hardwood (m/ha)	0.004	0.0003
Area weighted mean shape index, mxd pine-hardwd	2.29	0.10
Median patch size, pine forest (ha)	6.02	0.86
Edge density, pine forest (m/ha)	0.002	0.0002
Area weighted mean shape index, pine forest	1.70	0.12
Median patch size, openings (ha)	5.33	0.60
Edge density, openings (m/ha)	0.002	0.0002
Area weighted mean shape index, openings	1.68	0.11
Other landscape aspects		
Distance to nearest body of water (m)	428.39	32.08
Distance to nearest road (m)	292.76	31.83

[a] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 4.29. Principal components (PC) analysis results for landscape variables generated within 500 m radius circular buffers centered on avian point count stations (*n*=72) at Sandy Hollow WMA, LA.

	PC1	PC2	PC3	PC4	PC5
Eigenvalue:	7.55	4.17	3.73	1.95	1.67
Variance explained:	0.34	0.19	0.17	0.09	0.08
Variables:					
% Longleaf pine	-32[a]	-53	-46	**57** [b]	-20
% Longleaf savannah	-4	-5	-33	3	**93**
% Mixed pine-hardwood forest	-9	-19	**73**	-42	-45
% Pine forest	-15	**96**	-18	4	0
% Openings	**94**	-9	0	-17	-16
Median patch size, longleaf pine (ha)	-32	-53	-46	**57**	-20
Edge density, longleaf pine (m/ha)	-28	-26	-30	**85**	9
Area weighted mean shape index[c], longleaf pine	-10	21	6	**74**	45
Median patch size, longleaf savannah (ha)	-4	-5	-33	3	**93**
Edge density, longleaf savannah (m/ha)	-50	1	-7	**63**	52
Area weighted mean shape index, longleaf sav.	-58	3	10	**72**	4
Median patch size, mixed pine-hardwood (ha)	-9	-19	**73**	-42	-45
Edge density, mixed pine-hardwood (m/ha)	16	-9	**89**	-21	-28
Area weighted mean shape index, mixed p-h	36	-1	**76**	9	-4
Median patch size, pine forest (ha)	-15	**96**	-18	4	0
Edge density, pine forest (m/ha)	4	**92**	16	6	0
Area weighted mean shape index, pine forest	9	**60**	58	9	-10
Median patch size, openings (ha)	**94**	-9	0	-17	-16
Edge density, openings (m/ha)	**95**	-4	18	-14	4
Area weighted mean shape index, openings	**73**	7	27	-23	35
Distance to nearest water (m)	7	-31	-1	**-58**	14
Distance to nearest road (m)	9	-28	**-48**	-20	21

[a] Values are multiplied by 100 and rounded to the nearest integer.
[b] Values greater than |40| are considered primary component variables; if a variable appeared on more than one component, the component with which the variable had the strongest correlation (in bold) was interpreted to be a better representation of the variance explained by that variable.
[c] Average perimeter-to-area ratio for a class, weighted by the size of each stand (polygon).

Table 4.30. Principal components (PC, eigenvalues ≥ 1) derived from landscape variables associated with avian point count stations at Sandy Hollow WMA, LA, 2003-2004. Associated variables are those with a correlation coefficient of ≥ |0.40| with each respective PC.

PC	Associated Variables[a]	Interpretation
PC1	Openings [% composition, MEDPS, ED, AWMSI]	Openings
PC2	Pine forests [% composition, MEDPS, ED, AWMSI]	Pine forest
PC3	Mixed pine-hardwood [% composition, MEDPS, ED, AWMSI]; (-) Distance to nearest road	Mixed pine-hardwood
PC4	Longleaf pine [% composition, MEDPS, ED, AWMSI]; Longleaf savannah [ED, AWMSI]	Longleaf pine; edge and shape of longleaf savannah
PC5	Longleaf savannah [% composition, MEDPS]	Size and occurrence of longleaf savannah

[a] Variables are positively related to the principal component unless otherwise noted

Among the 5 component variables, occurrence of both Red-bellied Woodpecker and Tufted Titmouse were negatively influenced by openings (component 1) and positively influenced by pine forest (component 2). Alternatively, mixed pine-hardwood forest (component 3), was most important in explaining occurrence of Carolina Wren and Eastern Towhee, which increased in frequency with increasing proportion of mixed pine-hardwood forests.

Pine forest (component 2) and mixed pine-hardwood forest (component 3) positively influenced occurrence of Blue-gray Gnatcatcher. In contrast, occurrence of Indigo Bunting was most closely associated with openings (component 1) and longleaf pine, and amount of edge and shape of longleaf savannah (component 4). Occurrence of Indigo Bunting increased in frequency with an increase in proportion of openings in the landscape, and decreased in frequency with an increase in proportion longleaf pine and longleaf savannah.

Pine forest (component 2) and longleaf pine, and amount of edge and shape of longleaf savannah (component 4) influenced occurrence of both Summer Tanager and Brown-headed Cowbird, both negatively associated with proportion of pine forest in the landscape, and

positively associated with longleaf pine, and amount of edge and shape of longleaf savannah. Relative Akaike weights suggested that, longleaf pine and amount of edge and shape of longleaf savannah may be more important than pine forest in explaining variation in occurrence of both species.

Alternatively, pine forest (component 2), mixed pine-hardwood forest (component 3), and longleaf pine, and amount of edge and shape of longleaf savannah (component 4), were important to occurrence of Bachman's Sparrow. Occurrence of Bachman's Sparrow decreased in frequency with an increase in proportion of pine forest and mixed pine-hardwood forest in the landscape, and increased in frequency with an increase in longleaf pine-savannah. Influence of pine forests and mixed pine-hardwoods may be more important than longleaf pine-savannah in explaining variation in occurrence of Bachman's Sparrow.

Size and percentage of landscape in longleaf savannah (component 5) best explained occurrence of both Pine Warbler and Northern Cardinal, both of which were negatively associated with this component. On the contrary, occurrence of Carolina Chickadee and Yellow-breasted Chat was most strongly related to both openings (component 1) and size and percentage in landscape of longleaf savannah (component 5). Openings positively influenced occurrence of both Carolina Chickadee and Yellow-breasted Chat. However, size and percentage of landscape in longleaf savannah positively influenced Carolina Chickadee, whereas Yellow-breasted Chat was negatively influenced.

Openings (component 1), longleaf pine, and amount of edge and shape of longleaf savannah (component 4), and size and percentage in landscape of longleaf savannah (component 5) were important in explaining variation in occurrence of Brown Thrasher. Frequency of occurrence of Brown Thrashers decreased with increasing proportion of both openings and longleaf pine and amount of edge and shape of longleaf savannah in the landscape, and increased

with increasing proportion of landscape in longleaf savannah. Akaike weights indicated that longleaf pine, and amount of edge and shape of longleaf savannah, may have been more important than openings or size and percentage in landscape of longleaf savannah.

In slight contrast, occurrence of Eastern Wood-pewee was associated with pine forests (component 2), longleaf pine, and amount of edge and shape of longleaf savannah (component 4), and size and percentage in landscape of longleaf savannah (component 5). Pine forests and size and percentage in landscape of longleaf savannah negatively influenced occurrence of Eastern Wood-pewee, whereas landscape characteristics of longleaf pine, and amount of edge and shape of longleaf savannah had a positive influence.

Finally, openings (component 1), pine forests (component 2), longleaf pine, and amount of edge and shape of longleaf savannah (component 4), and size and percentage in landscape of longleaf savannah (component 5) were important in explaining variation in occurrence of Common Yellowthroat. Frequency in occurrence of Common Yellowthroat increased with an increase in proportion of openings and longleaf savannah in the landscape, and decreased with an increase in proportion of pine forests and longleaf pine in the landscape.

Table 4.31. Model selection results with respect to landscape characteristics for species detected during avian point counts at Sandy Hollow WMA, LA. Models reported are those with ΔAICc ≤ 2. K is number of model parameters, AICc is Akaike's Information Criterion for small sample size, ΔAICc is AICc difference between each model and best model, and w_i is Akaike weight.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Red-bellied Woodpecker	PC1 PC2	3	421.44	427.50	0	0.270
	PC2	2	424.71	428.74	1.24	0.146
Eastern Wood-pewee	PC4 PC5	3	181.42	190.47	0	0.302
	PC2 PC4 PC5	4	183.01	191.11	0.64	0.220
	PC3 PC4 PC5	4	184.07	192.16	1.69	0.130

[a] PC1: openings; PC2: pine forest; PC3: mixed pine-hardwood forest; PC4: longleaf pine, amount of edge and shape of longleaf savannah; PC5: size and occurrence of longleaf savannah.

Table 4.31. continued.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Carolina Chickadee	PC1	2	422.22	426.25	0	0.172
	PC5	2	423.33	426.35	0.10	0.163
	PC3	2	423.06	427.09	0.83	0.113
	PC2	2	423.51	427.54	1.29	0.090
	PC1 PC5	3	421.66	427.72	1.45	0.082
	PC4	2	424.11	428.14	1.88	0.067
Tufted Titmouse	PC1	2	422.95	426.97	0	0.251
	PC2	2	423.16	427.19	0.21	0.226
	PC4	2	424.67	428.70	1.73	0.106
Carolina Wren	PC3	2	424.70	428.73	0	0.210
	PC3 PC5	3	424.11	430.17	1.43	0.103
	PC2 PC3	3	424.64	430.70	1.96	0.079
Blue-gray Gnatcatcher	PC2 PC3	3	243.88	249.94	0	0.155
	PC1 PC2 PC3	4	242.72	250.82	0.88	0.099
	PC3	2	246.87	250.90	0.97	0.096
	PC2	2	246.88	250.91	0.97	0.095
	PC2 PC3 PC4	4	243.65	251.74	1.81	0.063
	PC1 PC3	3	245.70	251.75	1.82	0.063
Brown Thrasher	PC4 PC5	3	234.81	240.87	0	0.171
	PC3 PC4 PC5	4	233.17	241.27	0.40	0.140
	PC1 PC4	3	235.92	241.98	1.11	0.098
	PC1 PC2 PC4	4	234.31	242.41	1.54	0.079
Pine Warbler	PC5	2	425.59	429.61	0	0.293
	PC1 PC5	3	424.83	430.89	1.28	0.155
Common Yellowthroat	PC1 PC2 PC5	4	196.34	204.44	0	0.240
	PC1 PC2 PC4 PC5	5	194.38	204.52	0.08	0.231
	PC1 PC5	3	200.15	206.20	1.77	0.099
Yellow-breasted Chat	PC1	2	346.81	350.84	0	0.179
	PC1 PC3	3	345.92	351.98	1.13	0.102
	PC1 PC5	3	346.40	352.46	1.61	0.080
	PC5	2	348.51	352.53	1.69	0.077
Summer Tanager	PC4	2	206.99	211.02	0	0.197
	PC2 PC4	3	206.03	212.09	1.07	0.115

Table 4.31. continued.

Species	Model	K	Deviance	AIC_c	ΔAIC_c	w_i
Indigo Bunting	PC1	2	278.61	282.64	0	0.140
	PC1 PC4	3	277.41	283.47	0.83	0.093
	PC1 PC5	3	277.64	283.70	1.06	0.082
	PC4	2	279.78	283.81	1.17	0.078
	PC3	2	280.38	284.41	1.77	0.058
	PC5	2	280.39	284.42	1.78	0.058
	PC1 PC4 PC5	4	276.41	284.50	1.86	0.055
Northern Cardinal	PC5	2	425.93	429.96	0	0.274
	PC4 PC5	3	425.17	431.22	1.27	0.146
	PC3 PC5	3	425.75	431.81	1.85	0.109
Eastern Towhee	PC3	2	348.32	352.34	0	0.393
Bachman's Sparrow	PC2 PC3 PC4	4	375.63	383.73	0	0.344
	PC2 PC3	3	378.28	384.33	0.61	0.254
	PC2 PC3 PC4 PC5	5	375.32	385.46	1.74	0.144
Brown-headed Cowbird	PC4	2	392.88	396.91	0	0.266
	PC2 PC4	3	392.30	398.36	1.45	0.129
	PC2	2	394.74	398.77	1.87	0.105

Table 4.32. Model-averaged parameter estimates, unconditional standard errors, and relative Akaike weights ($\sum w_i$) of each explanatory component variable, representing landscape attributes, for select birds detected during point count surveys at Sandy Hollow WMA, LA, 2003-2004.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Red-bellied Woodpecker	PC1	-0.2636	0.1245	0.616
	PC2	0.2404	0.1212	0.831
	PC3	0.1106	0.1271	0.285
	PC4	0.0778	0.1191	0.054
	PC5	0.0016	0.1199	0.288
Eastern Wood-pewee	PC1	0.5981	0.1753	0.100
	PC2	-0.1256	0.2120	0.400
	PC3	0.0544	0.2190	0.300
	PC4	0.6498	0.1722	0.957
	PC5	-0.3150	0.1761	0.806
Carolina Chickadee	PC1	0.0886	0.1154	0.437
	PC2	0.1722	0.1151	0.214
	PC3	0.1520	0.1178	0.260
	PC4	-02010	0.1166	0.183
	PC5	0.1173	0.1134	0.407
Tufted Titmouse	PC1	-0.0797	0.1241	0.465
	PC2	0.0185	0.1366	0.419
	PC3	0.3838	0.1311	0.071
	PC4	0.0507	0.1237	0.212
	PC5	-0.1894	0.1210	0.199
Carolina Wren	PC1	0.0262	0.1095	0.270
	PC2	-0.0076	0.1137	0.331
	PC3	0.2790	0.1125	0.653
	PC4	-0.0574	0.1097	0.185
	PC5	-0.0788	0.1084	0.337
Blue-gray Gnatcatcher	PC1	0.0283	0.1524	0.376
	PC2	0.1497	0.1638	0.623
	PC3	0.2277	0.1655	0.706
	PC4	-0.2686	0.1600	0.373
	PC5	-0.3217	0.1450	0.085

[a] PC1: openings; PC2: pine forest; PC3: mixed pine-hardwood forest; PC4: longleaf pine, amount of edge and shape of longleaf savannah; PC5: size and occurrence of longleaf savannah.

Table 4.32. continued.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Brown Thrasher	PC1	-0.2252	0.1833	0.462
	PC2	-0.0419	0.1900	0.312
	PC3	-0.1674	0.1731	0.393
	PC4	-0.3659	0.1574	0.883
	PC5	0.5424	0.1520	0.651
Pine Warbler	PC1	0.1587	0.1375	0.358
	PC2	0.0440	0.1503	0.218
	PC3	0.0622	0.1454	0.220
	PC4	-0.0179	0.1282	0.253
	PC5	-0.4788	0.1311	0.980
Common Yellowthroat	PC1	1.0105	0.2122	0.854
	PC2	-0.6343	0.2830	0.781
	PC3	0.0240	0.2207	0.248
	PC4	-0.3815	0.1628	0.502
	PC5	1.1727	0.2062	0.864
Yellow-breasted Chat	PC1	0.2819	0.1365	0.661
	PC2	0.0530	0.1518	0.261
	PC3	0.0648	0.1523	0.368
	PC4	0.3683	0.1377	0.253
	PC5	-0.0821	0.1355	0.434
Summer Tanager	PC1	-0.1096	0.1649	0.235
	PC2	-0.1956	0.1945	0.426
	PC3	0.3913	0.1881	0.286
	PC4	0.2705	0.1660	0.785
	PC5	-0.1129	0.1632	0.346
Indigo Bunting	PC1	0.1584	0.1442	0.620
	PC2	0.0651	0.1568	0.200
	PC3	-0.0268	0.1550	0.320
	PC4	-0.1069	0.1413	0.433
	PC5	0.0614	0.1394	0.383
Northern Cardinal	PC1	0.1388	0.1155	0.203
	PC2	0.1178	0.1231	0.249
	PC3	0.2166	0.1200	0.283
	PC4	-0.0693	0.1150	0.348
	PC5	-0.1968	0.1136	0.924

Table 4.32. continued.

Species	Model variable [a]	Estimate	Standard error	$\sum w_i$
Eastern Towhee	PC1	0.2139	0.1343	0.028
	PC2	0.0623	0.1438	0.349
	PC3	0.1583	0.1444	0.668
	PC4	-0.3764	0.1319	0.025
	PC5	0.2390	0.1338	0.322
Bachman's Sparrow	PC1	0.0882	0.1395	0.090
	PC2	-0.3414	0.1564	0.903
	PC3	-0.4583	0.1542	0.997
	PC4	0.1735	0.1334	0.596
	PC5	0.2005	0.1333	0.293
Brown-headed Cowbird	PC1	0.2744	0.1245	0.252
	PC2	-0.0824	0.1387	0.415
	PC3	-0.0348	0.1353	0.288
	PC4	0.2790	0.1252	0.625
	PC5	0.1530	0.1248	0.034

DISCUSSION

Sherburne WMA

Previous studies have documented that uneven-aged forest management practices (i.e. selection cutting) generally positively affected early-successional bird species, whereas species typically associated with mature forests exhibited variable species-specific responses to management treatments (Gram et al. 2003, Annand and Thompson 1997, Rodewald and Smith 1998, Dickson et al. 1992). Gram et al. (2003) reported greater densities of Indigo Buntings and Yellow-breasted Chats (early-successional species), and Kentucky Warblers and Hooded Warblers (mature forest species), in areas treated with a combination of small-group and individual-tree selection cuts; densities of White-eyed Vireo (a generalist), and Acadian Flycatcher (a forest interior species), were similar in control and selection cutting with groups. Annand and Thompson (1997) found abundance of Indigo Buntings and Red-bellied Woodpeckers (a forest interior species) to be greater in stands with group-selection cuts than

single-tree selection and mature stands, and abundance of Hooded Warblers and Northern Parulas (forest interior species) greater in mature stands, and mature and group-selection stands, respectively. Additionally, their results indicated that 4 forest interior, 3 second growth, 3 early successional, and 4 generalist inhabitant species exhibited no differences in abundance with respect to single-tree, group selection, and mature stands. Rodewald and Smith (1998) found that abundance of understory nesters was greatest in mature stands versus treatment stands, which included group-selection cuts. Indigo Buntings in the study by Rodewald and Smith (1998) were most common on stands where group-selection cuts had occurred, whereas Acadian Flycatchers and Tufted Titmice were more abundant in mature stands relative to stands with group-selection cuts. However, placement and foraging behavior of 17 species in relation to juxtaposition of forested and logged (group-selection and clearcut) sites in Vermont used a mixture of disturbed and undisturbed forest, showing no preference for one type over the other (Lent and Capen 1995).

We found that at the stand-level, the combination of group and individual-selection cuts had a variety of effects on occurrence of breeding bird species at Sherburne. Six species, Common Yellowthroat, Yellow-breasted Chat, Eastern Towhee, Indigo Bunting, Great-crested Flycatcher, and Brown-headed Cowbird, occurred more frequently in stands where selection cutting with groups was prescribed than either individual-selection or uncut stands. Except for the latter 2 species, these species are associated with early-successional habitat (Hamel 1992), nest in early-successional understory, and are shrub and/or ground-gleaning foragers (Wiedenfeld and Swan 2000). Frequent canopy gaps in uneven-aged stands promote a well-developed understory and sub-canopy (Dickson et al. 1992). Stands where selection cutting with groups occurred had greater percentage *Rubus* spp., which we found to be the major constituent of impenetrable thicket-weeds that comprised these gaps. Such thickets are prime habitat for

numerous fauna, including low-nesting songbirds (Kellison and Young 1997). Some early successional species, such as Yellow-breasted Chat, may require large openings more typical of clearcuts (Dickson et al 1992), which may explain increased use of stands with group-selection cuts by early-successional species. Brown-headed Cowbird also was more frequently detected in areas with group-selection cuts in this study. Given that Cowbirds are attracted to forest margins, and parasitism is especially prevalent in fragmented forest lands, this was not unexpected. It is worth noting, however, that our detection of Brown-headed Cowbird across all stand types was low, which contrasts with reports in other hardwood forest systems, where abundance of cowbirds was common across mature, group selection, and single tree selection stands (Dickson et al. 1993).

Yellow-billed Cuckoo, Red-bellied Woodpecker, Swainson's Warbler, and Northern Cardinal were detected slightly more often in individual-selection stands, which in our study may be viewed as intermediate in seral stage between uncut stands and stands where a combination of individual-selection and group-selection cuts was performed. Individual-selection cuts occurred 17-18 years prior to the study, which was sufficient time for a majority of the stumps to decompose (Kenny Ribbeck, LDWF, pers. communication), and perhaps, for stands to attain structure similar to that of uncut stands, but with pockets of understory growth due to some light reaching the forest floor. Swainson's Warbler forages on the ground and breeds in dense thickets and vine tents in forests north of the Coastal Marsh Region in Louisiana (Wiedenfeld and Swan 2000). Greater occurrence of Swainson's Warbler in individual-selection stands suggests that these stands may provide a suitable combination of forest cover with patches of dense understory. Similarly, the Yellow-billed Cuckoo nests in dense vine tangles and forage in the canopy, particularly on cicadas and tent caterpillars (Wiedenfeld and Swan 2000) and thus, also may have found individual-selection stands more suitable than other stand types.

Four forest interior species, Acadian Flycatcher, Red-eyed Vireo, Northern Parula, and Prothonotary Warbler, as well as Carolina Chickadee and Tufted Titmouse, 2 generalist species, occurred more frequently in individual-selection and uncut stands, suggesting an association with a closed canopy forest, and perhaps other habitat variables such as tree density, litter, and bare ground. With the exception of Prothonotary Warbler, these forest interior species utilize the canopy for nesting and foraging (Wiedenfeld and Swan 2000), and thus may be sensitive to canopy openings. Our results for Red-eyed Vireo and Acadian Flycatcher contrasts slightly with Annand and Thompson (1997), who found occurrence to be similar across single tree, group-selection, and mature stand types for Acadian Flycatcher, and greater in the latter 2 stand types for Northern Parula. Prothonotary Warbler, Carolina Chickadee, and Tufted Titmouse nest in cavities, and may be impacted at the stand level by removal of overstory trees, which potentially alter stand structure and availability of hard and soft mast, ground-level vegetation, invertebrates, snags, and arboreal cavities (Wigley and Roberts 1994).

Hooded Warbler and American Redstart occurred more frequently in uncut stands than group-selection stands in our study. Although we identified both species as forest interior associates, and thus predicted this trend, other studies reported that densities of Hooded Warbler did not vary between control stands in combined single-tree and group-selection stands (Gram et al. 2003). Indeed, we detected Hooded Warbler in >60% of surveys conducted in group-selection stands, indicating substantial use of these areas, but to a relatively lesser extent than of uncut stands. In Oregon, 6 species, 3 forest interior, 2 generalists and 1 second-growth inhabitant, showed no noticeable difference in frequency of occurrence across stand types, perhaps because habitat upon which they relied or to which they were sensitive was not strongly impacted by these harvest methods at the stand level (Chambers et al 1999).

It has been suggested that most birds select habitat for breeding based on vegetation structure (Dickson et al. 1993). The degree to which habitat factors at the microhabitat scale have influenced avian species occurrence in previous research is variable (Martin 1989, Hennings and Edge 2003, Hagan and Meehan 2002, Berry and Bock 1998, Seoane et al. 2004; Hall and Mannan 1999, Weakland and Wood 2005, Saab 1999). Microhabitat components were the best determinant of occurrence for only 10 of 32 bird species examined in riparian cottonwood (*Populus angustifolia*) forests in Idaho, whereas landscape variables better predicted occurrence of 21 species. Herbaceous cover, shrub cover and dense subcanopies influenced occurrence of Yellow-breasted Chats; as well, dense subcanopies also were important to Brown-headed Cowbirds, whereas Downy Woodpeckers were associated with bare ground (Saab 1999).

In contrast, microhabitat variables performed better than landscape variables in explaining occurrence of 17 of 20 species in an industrial forest in Maine, with certain microhabitat variables positively related to occurrence of both early- and late-successional species (i.e. understory stem density negatively influenced both Blackburnian Warblers [late successional] and White-throated Sparrows [early successional]). Percentage canopy cover positively influenced occurrence of Cerulean Warbler (*Dendroica cerulea*) a species typically associated with mature deciduous forests, in West Virginia (Weakland and Wood 2005), whereas the Blue-gray Gnatcatcher, also a mature forest species, was not associated with any habitat variables in the Colorado foothills, leading the authors to suggest that this species may respond to a combination of associated variables (Berry and Bock 1998).

We found that models associating species with microhabitat components generally supported and paralleled relationships between species and stand type. Four forest interior and 1 forest opening species were positively associated with increasing tree density and canopy closure, of which Acadian Flycatcher and Prothonotary Warbler had the strongest association.

Tree density and canopy closure were both stand attributes of uncut and individual-selection stands, where these 2 species most frequently occurred. Canopy closure also was important for American Goldfinch and House Wrens in Idaho (Saab 1999), as well as for native birds breeding in riparian habitat in Oregon (Hennings and Edge 2003). In contrast, occurrence of Indigo Bunting and Common Yellowthroat had the strongest positive association with increasing percentage vine cover, which occurred in higher percentages in stands with group-selection cuttings. Increasing percentage bare ground was a determinant of occurrence of Acadian Flycatcher and Carolina Wren. Notably, occurrence of Carolina Wren (considered habitat generalists) did not differ at the stand level, suggesting that Carolina Wren may be selecting habitat based on microhabitat variables. Carolina Wrens sometimes forage at the ground level (Hamel 1992), and may have selected areas with bare ground, irrespective of stand type.

Increasing woody cover and shrub density were important determinants to occurrence of Northern Cardinal, Swainson's Warbler and Acadian Flycatcher, the latter 2 of which are considered forest interior birds. Swainson's Warbler is associated with dense woody understory habitat in forest areas (Wiedenfeld and Swan 2000, Hamel 1992), and the strong positive relationship with shrubs and woody cover further supports greater frequencies of occurrence of this species in individual-selection stands, where density of shrubs was greater than in uncut stands, and slightly greater than in stands where selection cutting with groups occurred. Acadian Flycatchers are salliers, and perhaps areas with woody cover provide suitable perching sites from which to sally for insects. Increasing shrub cover and decreasing number of trees also was associated with probability of breeding success of Elegant Trogons (*Trogon elegans*) in southeastern Arizona (Hall and Mannan 1999).

Occurrence of Common Yellowthroat, common across Louisiana in weedy fields and marshes (Wiedenfeld and Swan 2000), was more strongly associated with herbaceous cover than

any other species. Herbaceous cover was similar across stand types in this study, yet occurrence of common yellowthroat was substantially greater in group-selection stands (0.48) compared to individual-selection (0.02) and uncut stands (0.01), suggesting that Common Yellowthroats selected habitat at a finer scale within group-selection stands. This contrasts with Saveraid et al. (2001) who reported a negative influence of percentage forb cover on Common Yellowthroats. To a lesser extent, herbaceous cover positively influenced American Redstart, Red-eyed Vireo, and Yellow-breasted Chat. Yellow-breasted Chat also was strongly and positively associated with increasing herbaceous cover in Idaho (Saab 1999). Both American Redstart and Red-eyed Vireo nest and forage in the canopy; thus, occurrence of these species associated with herbaceous cover remains unclear. However, Hagan and Meehan (2002) reported a positive relationship of American Redstarts in Maine both with understory stems 2-4 m tall, and area of early-successional forest, which is typically characterized with herbaceous cover (Dickson et al 1993). Red-eyed Vireo was negatively associated with early successional forest in Maine (Hagan and Meehan 2002).

Coarse woody debris, which did not differ across stand types, was a strong positive determinant of occurrence of Red-bellied Woodpecker, Indigo Bunting, Brown-headed Cowbird and Northern Parula. Red-bellied Woodpecker typically utilizes snags and dead limbs for foraging and nesting (Conner et al. 1994), and percentage coarse woody debris on the ground likely was in close proximity to snags and trees with dead branches. As part of the harvest prescription, trees selected for harvest in group-selection cuts were felled and left in piles in the stands (K. Ribbeck, LDWF, pers. communication). Perhaps male Indigo Buntings and Brown-headed Cowbirds used these piles for singing or as lookout positions to defend territories or seek parasitism hosts, respectively. We are uncertain about the associations of coarse woody debris with occurrence of Northern Parula.

Our results at both the stand and microhabitat levels generally are consistent with other findings on use of selection cuttings by early and late successional breeding bird species. Group-selection cuttings seemed to favor early successional and edge species, whereas both individual-selection and group-selection cutting provided additional understory vegetation growth that supported forest associates that utilize understory, such as Kentucky Warbler and Swainson's Warbler, which has been documented in other bottomland hardwood forest systems that have implemented harvest prescriptions other than clearcuts (Dickson et al. 1993). Thompson et al. (1993) stated that uneven-aged harvest regimes retain a large proportion of the bird community associated with mature forest, as well as provide habitat for a number of early successional species that use the shrub-sapling layer.

Ben's Creek WMA

Habitat initially created as a result of even-aged management is important to bird species associated with early-successional and edge conditions, with an increase in suitability for late successional bird species as plant succession progresses in managed stands (Yahner 2000). Stands in an early-successional stage are structurally simple and characterized by minimal vertical foliage diversity. As even-aged stands succeed through different stages of development, these different stages are attractive and/or suitable to different species of bird (Dickson et al. 1993). Across the seral stage matrix, pine plantations provide herbaceous strata suitable for ground-nesting birds, seeds and invertebrate forage associated with trees and understory vegetation, and a dense woody understory for species dependent on shrubs (Allen et al. 1996). Thompson et al (1993) reported that density of early-successional Neotropical migrants was greater in landscapes under even-aged management than in wilderness areas with no timber harvest, whereas density of forest interior Neotropical migrants was slightly lower. Likewise, 10 of 37 Neotropical migrant species, including Mourning Warblers, Alder Flycatchers, and

Common Yellowthroats, were more abundant in even-aged stands <5 years old and 6-20 years old, whereas abundance of 13 other species, including Wood thrush and Red-eyed Vireos, was greater in mature stands in Maine (Hagan et al. 1997). Although, avian diversity was lowest in clearcuts (<5 years old) compared to even-aged regeneration cuts (6-20 years old) in Maine (Hagan et al. 1997), we found that bird diversity was similar and greater in 4-5- and 13-23-year stands relative to 7-9-year stands at Ben's Creek.

In Missouri, even-aged stands 3-6 years of age had greater detections of early successional and generalist species, including Indigo Bunting, Yellow-breasted Chat, Prairie Warbler, and Brown-headed Cowbird, than even-aged mature (~65 years old) stands, which had greater numbers of late successional species, such as Wood Thrush, Red-eyed Vireo, and Pine Warbler (Annand and Thompson 1997). Barber et al. (2001) documented nest predation and parasitism among regenerating (3-6 years old), mid-rotation (12-15 years old), and thinned (17-23 years old) pine plantations and reported that nests of Yellow-breasted Chat were predated and parasitized more in thinned stands, and mid-rotation and thinned stands, respectively, whereas parasitism levels for Common Yellowthroat, Indigo Bunting, Prairie Warbler, and Northern Cardinal did not differ across treatments. Barber et al. (2001) also found that Cowbird abundance was greater in regenerating and thinned stands than mid-rotation stands, and increased with increasing host abundance and decreasing canopy cover. Relative abundance of Blue Jay and American Crow, avian predators, was greater in mid-rotation, and mid-rotation and thinned stands, respectively, than regenerated stands. Other authors have stated that juxtaposition of different even-aged stands resulted in increased amounts of edge in the forest which may affect the reproductive success of Neotropical migrants (Thompson et al. 1993).

In general, we found trends in bird species composition across 1-, 4-5-, 7-9-, and 13-23-year, even-aged loblolly pine stands to be similar to previous studies (Yahner 2000 and

Thompson et al. 1993). However, comparison of our results to other studies regarding effects of even-aged management on birds should be interpreted cautiously, primarily because even-aged stands examined in previous studies were often much farther apart in seral stage, treated competing vegetation with burning and/or herbicides, were naturally regenerating deciduous forests versus pine plantations, or retained residual trees or residual patches of trees (Hagan et al. 1997, Schulte and Niemi 1998, Hestir and Cain 1999). At Ben's Creek, 5 of 6 species (Brown Thrasher, Brown-headed Cowbird, Common Yellowthroat, Prairie Warbler, Yellow-breasted Chat, Indigo Bunting) with greater frequencies in 1-year and/or 4-5-year stands were early-successional species, and 3 of these 5 were Neotropical migrants. This agrees with other reports of occurrence of early-successional species in loblolly pine regeneration stands <5 years old (Allen et al. 1996). Common Yellowthroats similarly occurred on 2-3 year clearcuts in Minnesota, whereas Brown Thrashers were completely absent from these stands (Schulte and Niemi 1998, Wiedenfeld and Swan 2000). Brown Thrasher was nearly absent on 1-year stands in our study, and occurred more frequently on 4-5-year stands than other forest age classes. Yahner (2003) suggested that Indigo Buntings used forested plots <3 years of age due to presence of residual trees, potentially used for singing or perching. Since occurrence of Indigo Bunting in our study was 3 times greater in 1-year stands than 4-year stands, Indigo Bunting perhaps utilized shrubs and coarse woody debris piles, or winged sumac (*Rhus copallina*) and yaupon (*Ilex vomitoria*), species that were abundant in the clearcuts, for perching and singing.

In contrast, 5 of 7 species with greater frequencies of occurrence in 11-23-year stands were late-successional species, including Wood Thrush, Red-eyed Vireo, Pine Warbler, and Kentucky Warbler, all Neotropical migrants. A substantial decrease in encounters with Red-eyed Vireo in stands prior to harvest compared to encounters 2-3 years after harvest in Pennsylvania provides additional support for the sensitivity to early-successional habitat of some

late-successional species, such as Red-eyed Vireo (Yahner 2003). In southern Arkansas, 15-year old even-aged loblolly pine stands supported greater densities of Carolina Wren, Hooded Warbler, Wood Thrush, Northern Cardinal, Common Yellowthroat, Eastern Towhee, White-eyed Vireo, and Yellow-breasted Chat, relative to 25-year old stands. The authors attributed this to dependence of these species on abundant deciduous vegetation in the understory for foraging and nesting (Hestir and Cain 1999). Our results contrast for the latter 4 species, which may be due to a combination of factors in our study, including similar percentage woody cover across forest age class, greater density of shrubs in 4-5- and 7-9-year stands, and potential differences in timing of commercial thinning between our study and the former. Indeed, occurrence of Eastern Towhee did not differ across seral stage, occurrence of White-eyed Vireo was greater in 4-5- and 7-9-year stands, Yellow-breasted Chat was similar in occurrence in both 1- and 4-5 year stands, and Common Yellowthroat is often more associated with herbaceous cover than dense woody shrubs in Louisiana (Wiedenfeld and Swan 2000).

Examining occurrence of avian species relative to microhabitat components revealed the relative importance of specific habitat conditions to distributions of breeding birds at an even finer scale at Ben's Creek. Barber et al. (2001) documented occurrence of nests of Red-eyed Vireos on late-rotation (>50 years) stands only, where overstory was dense (78.1% canopy cover), and also reported a negative association of Brown-headed Cowbird abundance with canopy cover. At Ben's Creek, occurrence of Red-eyed Vireo, a late successional Neotropical migrant, had the strongest association with increasing tree richness, tree density, and canopy cover, and decreasing herbaceous and vine cover, whereas occurrence of Brown-headed Cowbird and Eastern Towhee, edge and early-successional specialists, respectively, were inversely related to these habitat characteristics. Canopy cover and tree richness were similarly important for Red-eyed Vireo at the stand level, whereas stand level differences in canopy and herbaceous

cover were less important to Brown-headed Cowbird and Eastern Towhee. In Colorado, species richness was positively associated with shrub cover at the microhabitat scale (Berry and Bock 1998). Similarly, in our study, increasing density of shrubs and decreasing coarse woody debris were most important at the microhabitat level for Kentucky Warbler, Eastern Towhee and Prairie Warbler, species which forage and nest in the understory. Shrub density and coarse woody debris played a larger role at the microhabitat level compared to the stand level, where preference of forest age differed among these species. Occurrence of Indigo Bunting, which was greatest in 1-year stands, had the strongest inverse relationship to increasing density of shrubs and decreasing coarse woody debris, suggesting that within 1-year stands, Indigo Bunting selected areas with less shrubs and more coarse woody debris.

Bare ground and density of willow subcanopy were the most frequent significant predictors of species occurrence in cottonwood riparian forests in Idaho (Saab 1999). At Ben's Creek, among 8 species associated with increasing bare ground and decreasing canopy cover, occurrence of Hooded Warbler and White-eyed Vireo had the strongest negative relationship with these microhabitat variables, whereas Indigo Bunting had the strongest positive relationship. This generally corresponds with habitat selection at the stand level, at which canopy closure was greatest in the 2 older forest age classes, where Hooded Warbler occurred most often, and in which bare ground was greatest in 1-year stands, where occurrence of Indigo Bunting was substantially more frequent. This suggests that these variables held similar importance for these species at both the stand and microhabitat levels.

Finally, occurrence of 13 species (3 forest interior, 3 second growth/forest opening, 3 generalist, 4 early successional) was associated with increasing percentage woody cover, which was the most frequent predictor of species occurrence among microhabitat components at Ben's Creek. By far, frequency of occurrence of Red-eyed Vireo had the strongest negative association

(see Table 4.16) with this habitat characteristic, whereas occurrence of Yellow-breasted Chat, Eastern Towhee, and Common Yellowthroat, each early-successional understory nesters and shrub-gleaning foragers (Hamel 1992, Wiedenfeld and Swan 2000), had the strongest positive associations with increasing percentage woody cover. Stand level percentage of woody cover was similar across Stand Age, suggesting the greater importance of this variable at the microhabitat scale for each of these species.

Analysis of relationships between habitat characteristics and occurrence of select bird species at the landscape scale provided additional insight into habitat selection patterns of breeding birds on Ben's Creek. In Maine, 10 of 37 late-successional species were positively associated with medium-age or mature forest in the surrounding 1-km landscape, whereas 13 of 35 early-successional species were positively related to the proportion of early-successional habitat in the surrounding landscape (Hagan et al. 1997). In our study, White-eyed Vireo and Hooded Warbler exhibited the strongest positive relationship with closed canopy landscapes, which was of somewhat less importance at finer scales. Both species occurred at high frequencies in 7-9-year (closed canopy) stands, but not exclusively. In contrast, Indigo Bunting, Kentucky Warbler, and Yellow-breasted Chat each had the strongest negative association with closed canopy, providing further evidence of their general avoidance of closed canopy areas.

Hagan and Meehan (2002) reported that the total area of early-successional forest in the landscape was positively related to occurrence of American Redstart and Magnolia Warbler, and negatively related to presence of Red-eyed Vireo in Maine. We found that Eastern Towhee and Prairie Warbler, both early successional associates, had the strongest positive relationship with early successional plant communities. Early successional habitat positively influenced occurrence of Prairie Warbler at microhabitat, stand, and landscape scales, whereas preference of this habitat type by Eastern Towhee was only revealed at the microhabitat and landscape scales.

Thinning of even-aged 35-45 year old conifer stands increased species richness and density of 10 species in the Oregon Cascades (Hagar et al. 2004). We similarly found that occurrence of 7 species, Hooded Warbler, Wood Thrush, Red-eyed Vireo, Kentucky Warbler, Blue Jay, White-eyed Vireo, and Eastern Towhee, demonstrated a strong positive association with shape of stands commercially thinned, promoting canopy release (11-23-years old). Except for the latter 2 species, stands in this age category were important to these species at the stand level as well. Hooded Warbler and Wood Thrush, both forest associates, had the strongest relationships with this landscape attribute.

Turner et al. (2002) examined interactions between hardwood stands and the surrounding loblolly pine matrix in the lower coastal plain of South Carolina, and concluded that hardwood presence enabled forest interior species typical of hardwood stands to spill over into adjacent areas. Among 10 species influenced by SMZs at Ben's Creek, occurrence of Red-bellied Woodpecker, Kentucky Warbler, and Tufted Titmouse exhibited the strongest positive associations with SMZs, whereas 4 early successional species, Prairie Warbler, Eastern Towhee, Yellow-breasted Chat, and Common Yellowthroat, and one pine specialist, Pine Warbler, had the strongest negative association with SMZs, suggesting these species avoided stands adjacent to SMZs. Occurrence of Kentucky Warbler has been associated with riparian and wetland areas (Hamel 1992, Wiedenfeld and Swan 2000), and given the short stand rotation on Ben's Creek, SMZs probably had a larger source of snags or older deciduous trees suitable for woodpecker foraging and nesting (Moorman et al. 1999).

At Ben's Creek, occurrence of 2 forest species, Kentucky Warbler and Wood Thrush, decreased with increasing proportion of the landscape in 1-year stands, whereas occurrence of Brown-headed Cowbird increased. This underscores the importance of forested area for the former 2 species, both forest associates, which were virtually absent in 1-year stands. The fact

that both clearcuts and closed canopy stands, which were adjacent at some locations on the area, influenced the occurrence of Brown-headed Cowbird at the landscape scale demonstrates the attraction of this species to edge habitat, which is a common result of even age regeneration (Barber et al 2001).

A primary result of industrial forestry is conversion of large tracts of land to early-successional and intermediate-aged habitat relative to more mature natural forest regimes. Pine plantations managed on a pulpwood rotation of ~ 30 years provide suitable habitat for early-successional species, but habitat for species that require vegetative structure characteristic of stands beyond pole timber stage is limited (Allen et al. 1996). As a result, abundance of early-successional and late-successional species should increase and decrease, respectively, with this alteration in habitat. Species attracted to early-successional habitats include Neotropical migrants (e.g. Indigo Bunting, Yellow-breasted Chat, Prairie Warbler), and populations of these species have benefited from industrial forestry. Notably, some of these species (e.g. Prairie Warbler) are species of concern due to population declines on a regional, national, or even larger scale. However, potential ecological benefits of timber harvesting for such species is costly for late-successional species, many of which also are listed as species of concern either in this region, including Yellow-billed Cuckoo and Hooded Warbler, or on a national level, such as Brown-headed Nuthatch, Wood Thrush, Swainson's Warbler, and Kentucky Warbler (National Audubon Society 2002) .

Sandy Hollow WMA

Species composition of breeding birds detected at Sandy Hollow was similar to other studies in longleaf pine-savannah communities and open pine ecosystems (Krementz and Christie 1999; Conner et al. 2002, Provencher et al. 2002, Engstrom et al. 1996, Wilson et al. 1995), and included Pine Warbler, Brown-headed Nuthatch, and Bachman's Sparrow, specialists

of this habitat type (Hamel 1992), the latter 2 of which are species of national conservation concern (National Audubon Society 2002). Consistent with other studies, occurrence of bird species at Sandy Hollow was correlated with habitat characteristics at both landscape and microhabitat scales (Ribig and Sample 2001, Brotons and Herrando 2001, Moreira et al. 2003). Microhabitat- and landscape-level factors influenced occurrence of grassland bird density in south-central Wisconsin. Diversity of cover type in surrounding area, field habitat (i.e. grass cover), mean patch size, and distance to woody vegetation were important predictors of bird density, whereas size of field was not (Ribig and Sample 2001). As well, size of pine forest, and to a lesser extent, subarboreal vegetation and presence of large pine trees, were important predictors of bird species occurrence for forest interior species in the north-western Mediterranean basin (Brotons and Herrando 2001). In pine stands managed with prescribed fire in Portugal, abundance of shrub nesting bird species was associated with size of eucalyptus stands (landscape) and herbaceous cover, and size and richness of trees (microhabitat), whereas abundance of tree nesters was associated with size of shrublands and size of eucalyptus stands (Moreira et al. 2003).

We related occurrence of species to 5 microhabitat components at Sandy Hollow to determine habitat preferences at the microhabitat scale. We found increasing density of hardwood trees to be important to 4 early-successional and 1 late-successional species at the microhabitat level, among which Bachman's Sparrow had the strongest negative association, followed by Pine Warbler and Indigo Bunting, which is not surprising, given that the Pine Warbler is a specialist of pine forests and Indigo Bunting prefers early-successional habitat (Hamel 1992). In both Texas and Florida, abundance of Bachman's Sparrow in longleaf pine-savannah was associated with avoidance of hardwood vegetation (Conner 2002, Provencher et al. 2002). Increasing canopy closure was important to 3 early-successional, 3 second growth/generalist, and 2 late-successional

species, and predictably, had the greatest negative influence on occurrence of Common Yellowthroat and Brown Thrasher, both early successional associates, and the greatest positive influence on Pine Warbler, which nest and forage in the canopy (Hamel 1992).

Abundance of species such as Indigo Bunting and Yellow-breasted Chat was positively related to increasing density of shrubs in longleaf pine forests in Texas (Conner et al. 2002). Laiolo et al. (2004) similarly found percentage shrub cover and percentage vegetation to be important determinants of occurrence of species associated with grasslands and shrubs. We found increasing percentage shrub cover to be the most frequent predictor of species occurrence among microhabitat components at Sandy Hollow, influencing occurrence of 4 early-successional, 3 second growth/generalist, and 4 late-successional species. Among these, occurrence of Common Yellowthroat had the strongest positive association, followed by Yellow-breasted Chat, Blue-gray Gnatcatcher, and Carolina Wren. All of these species use the shrub layer in pine-savannah ecosystems for nesting, foraging and perching (Hamel 1992), which supports this relationship with shrubs at a finer scale.

Three early-successional, 3 second growth/generalist, and 3 late-successional species were strongly associated with increasing percentage of the landscape in loblolly pine and decreasing percentage longleaf pine, including Red-bellied Woodpecker and Northern Cardinal (positive), and Bachman's Sparrow, Yellow-breasted Chat and Eastern Wood-pewee (negative). The importance of increasing percentage of longleaf pine in the landscape to both Bachman's Sparrow (Hamel 1992, Provencher et al. 2002, Conner et al. 2002, Liu et al 1995) and Eastern Wood-pewee (Hamel 1992, Wilson et al. 1995) was not unexpected. Increasing percentage bare ground and decreasing herbaceous cover influenced occurrence of 8 species, including 3 early- and 2 late-successional species. Common Yellowthroat and Summer Tanager were among those positively associated, whereas Bachman's Sparrow, Eastern Wood-pewee and Tufted Titmouse

were negatively influenced by these variables. Similarly, abundance of Bachman's Sparrow and grass cover were positively correlated in stands under a spring prescribed burning regime in Florida (Provencher et al. 2002).

Habitat conditions at Sandy Hollow at the landscape scale influenced occurrence of early- and late-successional forest associates, as well as pine specialists and ubiquitous species. Openings (agriculture, ponds, residential areas) in the landscape impacted occurrence of 7 species, of which 3 early- and 1 late-successional species were positively influenced, and 1 late-successional and 2 second-growth/generalist species were negatively influenced. Patches of residential areas also influenced density of Savannah Sparrow (*Passerculus sandwhichensis*) in grasslands in Wisconsin, where density increased with decreasing proportion of residential areas in the landscape (Ribig and Sample 2001). Probability of occurrence of Bachman's Sparrow in pine-grassland restoration stands in the Homochitto National Forest, Mississippi, increased with decreasing pine sawtimber patch size and increasing pine sawtimber mean perimeter-area ratio (Wood et al. 2004).

Likewise, we found 8 species to be associated with increasing percentage of patches of pine forest in the landscape, among which Blue-gray Gnatcatcher and Red-bellied Woodpecker were positively associated, whereas Common Yellowthroat and Bachman's Sparrow had the strongest negative association with increasing pine forest. Occurrence of Bachman's Sparrow was the only species negatively influenced by increasing percentage of mixed pine-hardwood in the landscape, whereas occurrence of Eastern Towhee, Blue-gray Gnatcatcher, and Carolina Wren were positively influenced. In pine forests frequented by wildfire, and surrounded by a burnt shrubby matrix in the Mediterranean, fragment size, core area and fragment perimeter were positively associated with canopy species (Herrando and Brotons 2002) At Sandy Hollow, edge and shape of longleaf savannah and occurrence of longleaf pine had the greatest positive

influence on occurrence of Eastern Wood-pewee, followed by Brown-headed Cowbird and Summer Tanager, and the greatest negative influence on occurrence of Common Yellowthroat and Brown Thrasher. Likewise, patch size of longleaf savannah positively influenced Common Yellowthroat, Brown Thrasher, and Carolina Chickadee, and negatively influenced Pine Warbler, Eastern Wood-pewee, and Northern Cardinal, each associated to some extent with the canopy for nesting, foraging, or perching (Hamel 1992).

Study Limitations

We examined associations of bird species with habitat characteristics based on frequency of occurrence, not productivity (i.e. nest success). We cannot assume that productivity was highest in the habitat in which a species most frequently occurred. In a study such as this, measures of productivity for multiple species over multiple study areas would have been impractical (Hagan et al. 1997). However, on a coarse scale, we believe greater frequencies of occurrence correspond to greater habitat suitability. For example, the Yellow-breasted Chat occurred with much greater frequency in young, regenerating stands (≤5 years old) than in thinned stands (13-23 years old) at Ben's Creek. Therefore, we concluded that younger, early-successional stands provided more suitable habitat than older, thinned stands. However, nest success (based on daily mortality rates) was similar for Yellow-breasted Chat in regenerating (3-6 years old) and thinned stands (17-23 years old) in Arkansas (Barber et al. 2001). Thus, differences in frequency of occurrence can be misleading with respect to habitat and must be interpreted with caution (Hagan et al. 1997).

Management Implications

Our analyses have identified associations between birds and habitat features at microhabitat and stand scales in a bottomland hardwood forest managed with selection cuttings, at microhabitat and landscape scales in a longleaf pine-savannah maintained with prescribed fire,

and at microhabitat, stand, and landscape scales with respect to stand age in an even-aged, loblolly pine plantation. However, specific habitat management recommendations for conservation of bird diversity are difficult to make, because habitat characteristics that appeared to be advantageous for some species are disadvantageous, or potentially detrimental, for other species (Hagan et al. 1997). For example, 2 Neotropical migrant species of conservation concern, Painting Bunting and Northern Parula, have very different habitat requirements, and managing for one may negatively impact the other. We detected Neotropical migrants in all Stand Ages and across all stand types, but the importance of age or stage of succession was species-specific, and sometimes differed for a single species across study areas. Thus, efforts to combine management of timber with conservation of songbirds must take into consideration both the complexity of avian habitat requirements and the landscape context in which these requirements occur (Gram et al. 2003).

Nevertheless, previous studies have determined that negative impacts resulting from regeneration cuts to populations of many species are short-lived, and that a diversity of seral stages, including some of all age and structural classes, ultimately leads to greater species diversity (Wigley and Roberts 1994). At the bottomland hardwood swamp in our study (Sherburne), Prothonotary and Hooded Warblers, 2 additional Neotropical migrant species of concern, occurred more frequently in older age classes, but also were detected in younger stands. This supports the above conclusion and suggests that frequency of occurrence in these stands will increase over time. Regeneration timber harvesting that creates a mosaic of small-scale disturbances, such as individual-selection and group-selection cutting, would promote diverse habitat structure and complexity, which would support both early successional species and species that require a developed understory beneath a closed canopy. Such harvest practices also would maintain a portion of the landscape intact for species that require habitat attributes

associated with mature forest. Ultimately, management should focus on habitat features that have the greatest influence on fitness components (i.e. reproduction and survival), including both nesting and foraging resources, which can differ from each other, but are equally needed to sustain populations of songbirds during the breeding season.

LITERATURE CITED

Adams, J. P., M. J. Lacki, and M. D. Baker. 1996. Response of herpetofauna to silvicultural prescriptions in the Daniel Boone National Forest, Kentucky. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 50:312-320.

Allen, A. W., Y. K. Bernal, and R. J. Moulton. 1996. Pine plantations and wildlife in the southeastern United States: an assessment of impacts and opportunities. U. S. Department of the Interior, Information and Technical Report, National Biological Service, Washington, D.C.

Anderson, S. H. 2003. The relative importance of birds and insects as pollinators of the New Zealand flora. New Zealand Journal of Ecology 27:83-94.

Annand, E. M. and F. R. Thompson, III. 1997. Forest bird response to regeneration practices in central hardwood forests. Journal of Wildlife Management 61:159-171.

Aubry, K. B. 2000. Amphibians in managed, second-growth Douglas-fir forests. Journal of Wildlife Management 64:1041-1052.

Babbitt, K. J. and G. W. Tanner. 1997. Effects of cover and predator identity on predation of *Hyla squirrella* tadpoles. Journal of Herpetology 31:128-130.

Babbitt, K. J. and G. W. Tanner. 2000. Use of temporary wetlands by anurans in a hydrologically modified landscape. Wetlands 20:313-322.

Barber, D. R., T. E. Martin, M. A. Melchiors, R. E. Thill, and T. B. Wigley. 2001. Nesting success of birds of different silvicultural treatments in southeastern U. S. pine forests. Conservation Biology 15:196-207.

Bentley, J. W., M. Howell, and T. G. Johnson. 2002. Louisiana's timber industry: an assessment of timber product output and use, 1999. USDA Forest Service, Southern Research Station. Resource Bulletin SRS-081. 39 pp.

Berry, M. E. and C. E. Bock. 1998. Effects of habitat and landscape characteristics on avian breeding distributions in Colorado foothills shrub. The Southwestern Naturalist 43:453-461.

Block, W. M. and M. L. Morrison. 1998. Habitat relationships of amphibians and reptiles in California oak woodlands. Journal of Herpetology 32:51-60.

Blouin, D.C., and A. M. Saxton. 1990. General Linear Mixed Models (GLMM) User's Manual. Louisiana State University, Baton Rouge, USA.

Bohning-Gaese, K. 1997. Determinants of avian species richness at different spatial scales. Journal of Biogeography 24:49-60.

Boyer, W. D. 2000. Long-term effects of biennial prescribed fires on the growth of longleaf pine. Pages 18-21 *in* W. K. Moser and C. E. Moser, editors. Fire and forest ecology: innovative silviculture and vegetation management. Tall Timbers Fire Ecology Proceedings, No. 21. Tall Timbers Research Station, Tallahassee, FL.

Brotons, L. and S. Herrando. 2001. Factors affecting bird communities in fragments of secondary pine forests in the north-western Mediterranean basin. Acta Oecologica 22:1-11.

Bull, E. L. and B. C. Wales. 2001. Effects of disturbance on amphibians of conservation concern in eastern Oregon and Washington. Northwest Science 75: 174-179.

Bury, R. B. 2004. Wildfire, fuel reduction, and herpetofauna across diverse landscape mosaics in northwestern forests. Conservation Biology 18:968-975.

Canterbury, G. E., T. E. Martin, D. R. Petit, L. J. Petit, and D. F. Bradford. 2000. Bird communities and habitat as ecological indicators of forest condition in regional monitoring. Conservation Biology 14:544-558.

Carter, M. C. and C. D. Foster. 2004. Prescribed burning and productivity in southern pine forests: a review. Forest Ecology and Management 191:93-109.

Chambers, C. L., W. C. McComb, and J. C. Tappeiner, II. 1999. Breeding bird response to three silvicultural treatments in the Oregon coast range. Ecological Applications 9:171-185.

Clout, M. N. and J. R. Hay. 1989. The importance of birds as browsers, pollinators and seed dispersers in New Zealand forests. New Zealand Journal of Ecology 12:27-33.

Conant, R. and J. T. Collins. 1998. A field guide to the reptiles and amphibians of eastern and central North America, 3rd ed. Houghton Mifflin Company, New York, NY. 616 pp.

Conner, R. N., S. D. Jones, and G. D. Jones. 1994. Snag condition and woodpecker foraging ecology in a bottomland hardwood forest. Wilson Bulletin 106:242-257.

Conner, R. N., C. E. Shackelford, R. R. Schaefer, D. Saenz, and D. C. Rudolph. 2002. Avian community response to southern pine ecosystem restoration for red-cockaded woodpeckers. Wilson Bulletin 14:324-332.

Conroy, S. 1999. Lizard assemblage response to a forest ecotone in northeastern Australia: a synecological approach. Journal of Herpetology 33:409-419.

Cromer, R. B., J. D. Lanham, and H. H. Hanlin. 2002. Herpetofaunal response to gap and skidder-rut wetland creation in a southern bottomland hardwood forest. Forest Science 48:407-413.

Crump, M. L. and N. J. Scott, Jr. 1994. Visual encounter surveys. Pages 84-92 *in* W. R. Heyer,

M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster, editors. Measuring and monitoring biological diversity: standard methods for amphibians. Smithsonian Institution Press, Washington, D.C. 364 pp.

Dahl, Thomas E. 1990. Wetlands losses in the United States 1780's to 1980's. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. 13 pp.

Daubenmire, R. K. 1959. A canopy coverage method of vegetational analysis. Northeast Science 43:43-64.

Deng, W. and G. Zheng. 2004. Landscape and habitat factors affecting Cabot's tragopan *Tragopan caboti* occurrence in habitat fragments. Biological Conservation 117:25-32.

Dettmers, R., D. A. Buehler, J. G. Bartlett, and N. A. Klaus. 1999. Influence of point count length and repeated visits on habitat model performance. Journal of Wildlife Management 63:815-823.

DeMaynadier, P.G. and M. L. Hunter, Jr. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Reviews 3:230-261.

DeMaynadier, P. G. and M. L. Hunter. 1998. Effects of silvicultural edges on the distribution and abundance of amphibians in Maine. Conservation Biology 12:340-352.

DeStefano, S. and R. G. Haight. 2002. Forest wildlife-habitat relationships. Forest Science 48:177-178.

Dickson, J. G., F. R. Thompson, III, R. N. Conner, and K. E. Franzreb. 1993. Effects of silviculture on Neotropical migratory birds in central and southeastern oak pine forests. Pages 374-385 *in* D. M. Finch and P. W. Stangel, editors. Status and management of Neotropical migratory birds. U. S. Forest Service, General Technical Report RM-229.

Dodd, C. K. and B. S. Cade. 1998. Movement patterns and the conservation of amphibians breeding in small, temporary wetlands. Conservation Biology 12:331-339.

Dundee, H. A. and D. A. Rossman. 1989. The Amphibians and Reptiles of Louisiana. Louisiana State University Press, Baton Rouge, LA. 300 pp.

Dodd, C. K. and B. S. Cade. 1998. Movement patterns and the conservation of amphibians breeding in small, temporary wetlands. Conservation Biology 12:331-339.

Donnelly, M. A., C. Guyer, J. E. Juterbock, and R. A. Alford. 1994. Techniques for marking amphibians. Pages 277-284 *in* W. R. Heyer, M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster, editors. Measuring and monitoring biological diversity: standard methods for amphibians. Smithsonian Institution Press, Washington, D.C. 364 pp.

Drapeau, P., A. Leduc, J. Giroux, J. L. Savard, Y. Bergeron, and W. L. Vickery. 2000. Landscape-scale disturbances and changes in bird communities of boreal mixed-wood forests. Ecological Monographs 70:423-444.

Duguay, J. P. and P. B. Wood. 2002. Salamander abundance in regenerating forest stands on the Monongahela National Forest, West Virginia. Forest Science 48:331-335.

Dupuis, L. A., J. N. Smith, and F. Bunnell. 1995. Relation of terrestrial-breeding amphibian abundance to tree-stand age. Conservation Biology 9:645-653.

Elkie, P. C., R. S. Rempel, and A. P. Carr. 1999. Patch analyst user's manual: a tool for quantifying landscape structure. Ontario Ministry of Natural Resources. Northwest Science and Technology. Thunder Bay, Ontario. TM-002. 16 pp + Appendix.

Enge, Kevin M. 1998. Herpetofaunal drift-fence survey of steephead ravines in 2 river drainages. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 52:336-348.

Enge, Kevin M. 2001. The pitfalls of pitfall traps. Journal of Herpetology 35:467-478.

Engstrom, R. T., D. B. McNair, L. A. Brennan, C. L. Hardy, and L. W. Burger. 1996. Influence on birds of dormant versus lightning-season prescribed fire in longleaf pine forests: experimental design and preliminary results. Transactions of North American Wildlife and Natural Resource Conference 61:200-207.

Faccio, S. D. 2003. Postbreeding emigration and habitat use by Jefferson and spotted salamanders in Vermont. Journal of Herpetology 37:479-489.

Farnesworth, G. L., K. H. Polluck, J. D. Nichols, T. R. Simons, J. E. Hines, and J. R. Sauer. 2002. A removal model for estimating detection probabilities from point-count surveys. The Auk 119:414-425.

Fredericksen, N. J., and T. S. Fredericksen. 2004. Impacts of selective logging on amphibians in a Bolivian tropical humid forest. Forest Ecology and Management 191:275-282.

Freemark, K. and B. Collins. 1992. Landscape ecology of birds breeding in temperate forest fragments. Pages 443-454 *in* J. M. Hagan, III and D. W. Johnston, editors. Ecology and conservation of Neotropical migrant landbirds. Smithsonian Institution Press, Washington, D.C.

Garson, J., A. Aggarwal, and S. Sarker. 2002. Birds as surrogates for biodiversity: an analysis of a data set from southern Quebec. Journal of Bioscience 27:347-360.

Gibbs, J. P. 1998. Amphibian movements in response to forest edges, roads, and streambeds in southern New England. Journal of Wildlife Management 62:584-589.

Gram, W. K., P. A. Porneluzi, R. L. Clawson, J. Faaborg, and S. C. Richter. 2003. Effects of experimental forest management on density and nesting success of bird species in

Missouri Ozark forests. Conservation Biology 17:1324-1337.

Grant, B.W., K.L. Brown, G.W. Ferguson, and J.W. Gibbons. 1994. Changes in amphibian biodiversity associated with 25 years of pine forest regeneration: implications for biodiversity management. Pages 355-367 i*n* S.K. Majumdar, F.J. Brenner, J.E. Lovich, J.F. Schalles, and E.W. Miller, editors. Biological Diversity: Problems and Challenges. The Pennsylvania Academy of Science.

Greenberg, C. H., D. G. Neary, and L. D. Harris. 1994. A comparison of herpetofaunal sampling effectiveness of pitfall, single-ended, and double-ended funnel traps used with drift fences. Journal of Herpetology 28:319-324.

Gresham, C. A. 2002. Sustainability of intensive loblolly pine plantation management in the South Carolina Coastal Plain, USA. Forest Ecology and Management 155: 69-80.

Grialou, J. A., S. D. West, and R. N. Wilkins. 2000. The effects of forest clearcut harvesting and thinning on terrestrial salamanders. Journal of Wildlife Management 64:105-113.

Guerry A. D. and M. L. Hunter, Jr. 2002. Amphibian distributions in a landscape of forests and agriculture: an examination of landscape composition and configuration. Conservation Biology 16:745-754.

Hagan, J. M. and D. W. Johnston. 1992. Ecology and Conservation of Neotropical migrant landbirds. Smithsonian Institution Press, Washington, D.C. 610 pp.

Hagan, J. M., P. S. McKinley, A. L. Meehan, S. L. Grove. 1997. Diversity and abundance of landbirds in a northeastern industrial forest. Journal of Wildlife Management 61:718-735.

Hagan, J. M. and A. L. Meehan. 2002. The effectiveness of stand-level and landscape-level variables for explaining bird occurrence in an industrial forest. Forest Science 48: 231-241.

Hagar, J., S. Howlin, and L. Ganio. 2004. Short-term response of songbirds to experimental thinning of young Douglas-fir forests in the Oregon Cascades. Forest Ecology and Management 199:333-347.

Hager, S. B. 2001. Microhabitat use and activity patterns of *Holbrookia maculata* and *Sceloporus undulates* at White Sands National Monument, New Mexico. Journal of Herpetology 35:326-330.

Hall, L. S. and R. W. Mannan. 1999. Multiscaled habitat selection by elegant trogans in southeastern Arizona. Journal of Wildlife Management 63:451-461.

Hamel, P. B. 1992. Land manager's guide to birds of the south. The Nature Conservancy, Southeast Region, Chapel Hill, NC. 437 pp.

Hamel, P. B., W. P. Smith, D. J. Twedt, J. R. Woehr, E. Morris, R. B. Hamilton, and R. J. Cooper. 1996. A land manager's guide to point counts of birds in the southeast. United States Department of Agricultural Forest Service. Southern Research Station. General Technical Report SO-120. 39 pp.

Hardin, J. W., D. J. Leopold, and F. M. White. 2001. Harlow and Harrar's Textbook of Dendrology, 9th Ed. McGraw-Hill, New York, NY. 533 pp.

Hargis, C. D., J. A. Bissonette, and J. L. David. 1998. The behavior of landscape metrics commonly used in the study of habitat fragmentation. Landscape Ecology 13:167-186.

Harpole, D. N. and C. A. Haas. 1999. Effects of seven silvicultural treatments on terrestrial salamanders. Forest Ecology and Management 114:349-356.

Harris, M., B. Kuhn, G. Price, P. Smith, W. Xu, J. Stevens, and K. Addy. 2002. Responses to the southern forest resource assessment: ecological effects of pine plantation management. Journal of Forestry 100:58-59.

Haywood, J. D., F. L. Harris, and H. E. Grelen. 2001. Vegetative response to 37 years of seasonal burning on a Louisiana longleaf pine site. Southern Journal of Applied Forestry 25:122-130.

Hecnar, S. T. and R. T. M'Closkey. 1998. Species richness patterns of amphibians in southwestern Ontario ponds. Journal of Biogeography 25:763-772.

Hennings, L. A. and W. D. Edge. 2003. Riparian bird community structure in Portland, Oregon: habitat, urbanization, and spatial scale patterns. The Condor 105:288-302.

Hermann, S. M., T. V. Hook, R. W. Flowers, L. A. Brennan, J. S. Glitzenstein, D. R. Streng, J. L. Walker, and R. L. Myers. 1998. Fire and biodiversity: studies of vegetation and arthropods. North American Wildlife and Natural Resources Conference 63:384-401.

Herrando, S. and L. Brotons. 2002. Forest bird diversity in Mediterranean areas affected by wildfires: a multi-scale approach. Ecography 25:161-172.

Hestir, J. L and M. D. Cain. 1999. Bird diversity and composition in even-aged loblolly pine stands relative to emergence of 13-year periodical cicadas and vegetation structure. USDA, Southern Research Station Pages. Pages 243-248 *in* Tenth Biennial Southern Silvicultural Research Conference, Shreveport, LA.

Heyer, W. R., M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster. 1994. Measuring and monitoring biological diversity: standard methods for amphibians. Smithsonian Institution Press, Washington, D.C. 364 pp.

Hiers, J. K., R. Wyatt, and R. J. Mitchell. 2000. The effects of fire regime on legume reproduction in long pine savannahs: is a season selective? Oecologia 125:521-530.

Hokit, D. G., B. M. Smith, and L. C. Branch. 1999. Effects of landscape structure in Florida scrub: a population perspective. Ecological Applications 9:124-134.

Hosmer, D. W. and S. Lemeshow. 2000. Applied Logistic Regression. Wiley & Sons, NY. 373 pp.

Jackson, D. A. 1993. Stopping rules in principal components analysis: a comparison of heuristical and statistical approaches. Ecology 74:2202-2214.

Janzen, F. J. 1994. Climate change and temperature-dependent sex determination in reptiles. Proceedings of the National Academy of Science 91:7487-7490.

Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65-71.

Kellison, R. C. and M. J. Young. 1997. The bottomland hardwood forest of the southern United States. Forest Ecology and Management 90:101-115.

Knutson, M. G., W. B. Richardson, D. M. Reineke, B. R. Gray, J. R. Parmelee, and S. E. Weick. 2004. Agricultural ponds support amphibian populations. Ecological Applications 14:669-684.

Kolozsvary, M. B. and R. K. Swihart. 1999. Habitat fragmentation and the distribution of amphibians: patch and landscape correlates to farmland. Canadian Journal of Zoology 77:1288-1299.

Krefting, L. W. and E. I. Roe. 1949. The role of some birds and mammals in seed germination. Ecological Monographs 19:269-286.

Krementz, D. G. and J. S. Christie. 1999. Scrub-successional bird community dynamics in young and mature longleaf pine-wiregrass savannahs. Journal of Wildlife Management 63:803-814.

Laiolo, P., F. Dondero, E. Ciliento, and A. Rolando. 2004. Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. Journal of Applied Ecology 41:294-304.

Lemmon, P. E. 1956. A spherical densiometer for estimation of forest overstory density. Forest Science 2:314-320.

Lent, R. A. and D. E. Capen. 1995. Effects of small-scale habitat disturbance on the ecology of breeding birds in a Vermont (USA) hardwood forest. Ecography 18:97-108.

Lewis, S. D., R. R. Fleet, and F. L. Rainwater. 2000. Herpetofaunal assemblages of four forest types in the Big Sandy Creek Unit of the Big Thicket National Preserve. Texas Journal of Science 52:139-150.

Littell, R. C., G. A. Milliken, W. W. Stroup, and R. D. Wolfinger. 2002. SAS system for mixed models. SAS Institute Inc., Cary, NC. 633 pp.

Litwin, T. S. and C. R. Smith. 1992. Factors influencing the decline of Neotropical migrants in a northeastern forest fragment: isolation, fragmentation, or mosaic effects? Pages 483-496 *in* J. M. Hagan, III and D. W. Johnston, editors. Ecology and conservation of Neotropical migrant landbirds. Smithsonian Institution Press, Washington, D.C.

Liu, J., J. B. Dunning, Jr., and H. R. Pulliam. 1995. Potential effects of a forest management plan on Bachman's sparrows (Aimophila aestivalis): linking a spatially explicit model with GIS. Conservation Biology 9:62-75.

Lockwood, C. C. 1981. Atchafalaya: America's largest river basin swamp. Claitor's Publishing Division, Baton Rouge, LA. 105 pp.

Louisiana Amphibian Monitoring Program (LAMP). 1996. Forms. www.wlf.state.la.us

Louisiana Department of Wildlife and Fisheries (LDWF). 2002. Wildlife management areas. www.wlf.state.la.us

Louisiana's Natural Heritage Program (LNHP). 2005. Louisiana's rare animals of conservation concern. Louisiana Department of Wildlife and Fisheries http://www.wlf.state.la.us/apps/netgear/clientFiles/lawlf/files/1103556439.pdf.

MacFaden, S. W. and D. E. Capen. 2002. Avian habitat relationships at multiple scales in a New England forest. Forest Science 48:243-253.

Martin, T.E. 1992. Breeding productivity considerations: What are the appropriate habitat features for management?: Pages 455–473 *in* J.M. Hagan III, D.W. Johnston, editors. Ecology and conservation of Neotropical migrant birds.. Smithsonian Institution Press, Washington, D.C.

Maurer, B. A. 1993. Biological diversity, ecological integrity, and Neotropical migrants: perspectives for wildlife management. Pages 24-31 *in* D. M. Finch and P. W. Stangel, editors. Status and management of Neotropical migratory birds. U.S. Forest Service, General Technical Report RM-229.

McLeod, R. F., and J. E. Gates. 1998. Response of herpetofaunal communities to forest cutting and burning at Chesapeake Farms, Maryland. American Midland Naturalist 139:164-177.

Messere, M. and P. K. Ducey. 1998. Forest floor distribution of northern redback salamanders, *Plethodon cinereus*, in relation to canopy gaps first year following selective logging. Forest Ecology and Management 107:319-324.

Miller, J. H. and K. V. Miller. 1999. Forest plants of the southeast and their wildlife uses. Southern Weed Science Society, Auburn, AL. 454 pp.

Mitchell, J. C., S. C. Rinehart, J. F. Pagels, K. A. Buhlmann, and C. A. Pague. 1997. Factors influencing amphibian and small mammal assemblages in central Appalachian forests. Forest Ecology and Management 96:65-76.

Moorman. C. E., K. R. Russell, G. R. Sabin, and D. C. Guynn. 1999. Snag dynamics and cavity occurrence in the South Carolina Piedmont. Forest Ecology and Management 118:37-48.

Moreira, F., A. Delgado, S. Ferreira, R. Borralho, N. Oliveira, M. Inacio, J. S. Silva, and F. Rego. 2003. Effects of prescribed fire on vegetation structure and breeding birds in young *Pinus pinaster* stands of northern Portugal. Forest Ecology and Management 184:225-237.

Morneault, A. E., B. J. Naylor, L. S. Schaeffer, and D. C. Othmer. 2004. The effect of shelterwood harvesting and site preparation on eastern salamanders in white pine stands. Forest Ecology and Management 199:1-10.

Morrison, M. L., W. M. Block, L. S. Hall, and H. S. Stone. 1995. Habitat characteristics and monitoring of amphibians and reptiles in the Huachua Mountains, Arizona. The Southwestern Naturalist 40:185-192.

National Audubon Society. 1999. http://www.audubon.org/campaign/wetland/atcha.html

National Audubon Society. 2002. http://audubon2.org/webapp/watchlist/viewWatchlist.jsp

Nuzzo, V. A. and K. S. Mierzwa. 2000. The effect of forest structure on amphibian abundance and diversity in the Chicago region. Citizens for Conservation. U. S. Environmental Protection Agency, Great Lakes National Program Office. 29 pp.

Ott, R. L. and M. Longnecker. 2001. An introduction to statistical methods and data analysis, 5th ed. Duxbury Thomson Learning, Pacific Grove, CA. 1152 pp.

Parris, K. M. and D. B. Lindenmayer. 2004. Evidence that creation of a *Pinus radiata* plantation in south-eastern Australia has reduced habitat for frogs. Acta Oecologica 25:93-101.

Perkins, M. W., R. J. Johnson, and E. E. Blankenship. 2003. Response of riparian avifauna to percentage and pattern of woody cover in an agricultural landscape. Wildlife Society Bulletin 31:642-660.

Petranka, J. W., M. P. Brannon, M. E. Hopey, and C. K. Smith. 1994. Effects of timber harvesting on low elevation populations of southern Appalachian salamanders. Forest Ecology and Management 67:135-147.

Petranka, J. W., M. E. Eldridge, and K. E. Haley. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology 7:363-370.

Petranka, J. W., C. A. Kennedy, and S. S. Murray. 2003. Response of amphibians to restoration

of a southern Appalachian wetland: a long-term analysis of community dynamics. Wetlands 23:1030-1042.

Platt, W. J., G. W. Evans, and S. J. Rathburn. 1988. The population dynamics of a long-lived conifer (*Pinus palustris*). American Naturalist 131:491-525.

Pough, F. H. 1980. The advantages of ectothermy for tetrapods. The American Naturalist 115:92-112.

Pough, F. H., E. M. Smith, D. H. Rhodes, and A. Collazo. 1987. The abundance of salamanders in forest stands with different histories of disturbance. Forest Ecology and Management 20:1-9.

Pough, F. H., R. M. Andrews, J. E. Cadle, M. L. Crump, A. H. Savitzky, and K. D. Wells. 1998. Herpetology. Prentice Hall, Upper Saddle River, NY. 577 pp.

Provencher, L., N. M. Gobris, L. A. Brennan, D. R. Gordon, and J. L. Hardesty. 2002. Breeding bird response to midstory hardwood reduction in Florida sandhill longleaf pine forests. Journal of Wildlife Management 66:641-661.

Ralph, J. C. 1993. Designing and implementing a monitoring program and the standards for conducting point counts. Pages 204-207 *in* D. M. Finch and P. W. Stangel, editors. Status and management of Neotropical migratory birds. U.S. Forest Service, General Technical Report RM-229.

Renken, R. O., W. K. Gram, D. K. Fantz, S. C. Richter, T. J. Miller, K. B. Ricke, B. Russell, and X. Wang. 2004. Effects of forest management on amphibians and reptiles in Missouri Ozark forests. Conservation Biology 18:174-188.

Reuss, M. 1988. Challenge of a vital resource: Along the Atchafalaya. Environment. 30:8-44.

Ribig, C. A. and D. W. Sample. 2001. Associations of grassland birds with landscape factors in southern Wisconsin. American Midland Naturalist 146:105-121.

Richter, S. C., J. E. Young, R. A. Seigel, and G. N. Johnson. 2001. Postbreeding movements of the dark gopher frog, *Rana servosa*: implications for conservation and management. Journal of Herpetology 35:316-321.

Robbins, C. S., J. R. Sauer, and B. G. Peterjohn. 1993. Population trends and management opportunities for Neotropical migrants. Pages 17-23 *in* D. M. Finch and P. W. Stangel, editors. Status and management of Neotropical migratory birds. U. S. Forest Service, General Technical Report RM-229.

Rodewald, P. G. and K. G. Smith. 1998. Short-term effects of understory and overstory management on breeding birds in Arkansas oak-hickory forests. Journal of Wildlife Management 62:1411-1417.

Rosson, J. F. 1995. Forest resources of Louisiana, 1991. USDA Forest Service, Southern Forest Experiment Station. Resource Bulletin SO-192. 71 pp.

Rudis, V. A. 1988. Nontimber values of Louisiana's Timberland. USDA Forest Service, Southern Forest Research Station, New Orleans, LA. Resource Bulletin SO-132. 27 pp.

Russell, K. R., H. G. Hanlin, T. B. Wigley, and D. C. Guynn. 2002. Responses of isolated wetland herpetofauna to upland forest management. Journal of Wildlife Management 66:603-617.

Ryan, T. J., T. Philippi, Y. A. Leiden, M. E. Dorcas, T. B. Wigley, and J. W. Gibbons. 2002. Monitoring herpetofauna in a managed forest landscape: effects of habitat types and census techniques. Forest Ecology and Management 167:83-90.

Saab, V. 1999. Importance of spatial scale to habitat use by breeding birds in riparian forests: a hierarchical analysis. Ecological Applications 9:135-151.

Saveraid, E. H., D. M. Debinski, K. Kindscher, and M. E. Jakubauskas. 2001. A comparison of satellite data and landscape variables in predicting bird species occurrence in the Greater Yellowstone Ecosystem, USA. Landscape Ecology 16:71-83.

Schulte, L. A. and G. J. Niemi. 1998. Bird communities of early-successional burned and logged forest. Journal of Wildlife Management 62:1418-1429.

Schurbon, J. M. and J. E. Fauth. 2003. Effects of prescribed burning on amphibian diversity in a southeastern U. S. national forest. Conservation Biology 17:1338-1349.

Seebacher, F. and R. A. Alford. 1999. Movement and microhabitat use of a terrestrial amphibian (*Bufo marinus*) on a tropical island: season variation and environmental correlates. Journal of Herpetology 33:208-214.

Seoane, J., J. Bustamante, and R. Diaz-Delgado. 2004. Competing roles for landscape, vegetation, topography and climate in predictive models of bird distribution. Ecological Modeling 171:209-222.

Sheffield, R. M. and J. G. Dickson. 1998. The South's forestland: on the hot seat to provide more. Transactions of the 63rd North American Wildlife and Natural Resources conference. SRS: 316-331.

Sibley, D. A. 2000. National Audubon Society: The Sibley Guide to Birds. Chanticleer Press, New York, NY. 545 pp.

Stanturf, J. A. 1994. Abstracts workshop on harvesting impacts on bottomland hardwood forest ecosystems. General Technical Report SO-103. United States Department of Agriculture Forest Service. Southern Forest Experiment Station. 23 pp.

Steele, C. A., E. D. Brodie, and MacCracken, J. G. 2003. Relationships between abundance of

cascade torrent salamanders and forest age. Journal of Wildlife Management 67:447-453.

Stevens, C. E., A. W. Diamond, and T. S. Gabor. 2002. Anuran call surveys on small wetlands in Prince Edward Island, Canada, restored by dredging of sediments. Wetlands 22:90-99.

Strijbosch, H. 1980. Habitat selection by amphibians during their terrestrial phase. British Journal of Herpetology 6:93-98.

Sturges, F. W., R. T. Holmes, G. E. Likens. 1974. The role of birds in nutrient cycling in a northern hardwoods ecosystem. Ecology 55:149-155.

Thill, R. 1990. Managing southern pines for wildlife. Pages 58-68 *in*: Proceedings XIXth IUFRO World Congress. Canadian International Union of Forestry Research Organizations. Vol 1. Montreal, Canada.

Thompson, F. R., III, J. R. Probst, and M. G. Raphael. 1993. Silvicultural options for Neotropical migratory birds. Pages 353-362 *in* D. M. Finch and P. W. Stangel, editors. Status and management of Neotropical migratory birds. U. S. Forest Service, General Technical Report RM-229.

Trani, M. K. 2002. Terrestrial ecosystems. Pages 3-45 *in* D. N. Wear and J. G. Greis, editors. Southern Forest Resources Assessment. U. S. D. A. Forest Service, Southern Research Station. General Technical Report SRS-53. 635 pp.

Turner, J. C., J. A. Gerwin, and R. A. Lancia. 2002. Influences of hardwood stand area and adjacency on breeding birds in an intensively managed pine landscape. Forest Science 48:323-330.

United States Fish and Wildlife Service (USFWS). 1985. Red-cockaded woodpecker recovery plan. Atlanta; USFWS.

United States Fish and Wildlife Service (USFWS). 1998. The longleaf pine/wiregrass ecosystem. http://www.fws.gov/carolinasandhills/longleaf.html.

USGS Patuxent Wildlife Research Center Frogwatch USA Program. 2002. www.nwf.org/keepthewildalive/frogwatch.

Vesley, D. G. and W. C. McComb. 2002. Salamander abundance and amphibian species richness in riparian buffer strips in the Oregon coast range. Forest Science 48:291-297.

Vissage, J. S., P. E. Miller, and A. J. Hartsell. 1992. Forest statistics for Louisiana Parishes – 1991. USDA Forest Service. Southern Forest Experiment Station, New Orleans, LA. Resource Bulletin SO-168, 65 pp.

Weakland, C. A. and P. B. Wood. 2005. Cerulean warbler (*Dendroica cerulea*) in microhabitat and landscape-level habitat characteristics in southern West Virginia. The Auk 122:497-

508.

Wiedenfeld, D. A. and M. Mark Swan. 2000. Louisiana Breeding Bird Atlas, Internet Version. Louisiana Sea Grant College Program, Louisiana State University, Baton Rouge, 78 pp.

Wilson, C. W., R. E. Masters, and G. A. Bukenhofer. 1995. Breeding bird response to pine-grassland community restoration for red-cockaded woodpeckers. Journal of Wildlife Management 59:56-67.

Wigley, T. B., and T. H. Roberts. 1994. A review of the wildlife changes in southern bottomland hardwoods due to forest management practices. Wetlands 14:41-48.

Woinarski, J. C. Z., C. Brock, A. Fisher, D. Milne, and B. Oliver. 1999. Response of birds and reptiles to fire regimes on pastoral land in the Victoria River District, Northern Territory. Rangeland Journal 21:24-38.

Wood, D. R., L. W. Burger, Jr., J. L. Bowman, and C. L. Hardy. 2004. Avian community response to pine-grassland restoration. Wildlife Society Bulletin 32:819-829.

Wyman, R. 1990. What's happening to the amphibians? Conservation Biology 4:350-352.

Yahner, R. H. 2000. Long-term effects of even-aged management on bird communities in central Pennsylvania. Wildlife Society Bulletin 28:1102-1110.

Yahner, R. H. 2003. Responses of bird communities to early successional habitat in a managed landscape. Wilson Bulletin 115:292-298.

Yin, R., and R. A. Sedjo. 2001. Is this the age of intensive management: a study of loblolly pine on Georgia's piedmont. Journal of Forestry 99:10-17.

Zimmerman, B. 1994. Audio strip transects. Pages 92-97 *in* W. R. Heyer, M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster, editors. Measuring and monitoring biological diversity: standard methods for amphibians. Smithsonian Institution Press, Washington, D.C. 364 pp.

APPENDIX 1. WOODY SPECIES RECORDED AT DRIFT FENCE ARRAYS AND AVIAN POINT COUNT STATIONS AT SHERBURNE WMA, LA

Common Name	Scientific Name
Boxelder	*Acer negundo*
Drummond red maple	*Acer rubrum* var. *drummondii*
Devil's-walkingstick	*Aralia spinosa*
French mulberry	*Callicarpa americana*
Water hickory	*Carya aquatica*
Bitternut hickory	*Carya cordiformis*
Sugarberry	*Celtis laevigata*
Buttonbush	*Cephalanthus occidentalis*
Roughleaf dogwood	*Cornus drummondii*
Hawthorn	*Crataegus* spp.
Persimmon	*Diospyros virginiana*
Swamp privet	*Forestiera acuminata*
Green ash	*Fraxinus pennsylvanica*
Pumpkin ash	*Fraxinus profunda*
Honeylocust	*Gleditsia triacanthos*
Deciduous holly	*Ilex decidua*
American holly	*Ilex opaca*
Virginiawillow	*Itea virginica*
Chinese privet	*Ligustrum sinense*
Sweetgum	*Liquidambar styraciflua*
Red mulberry	*Morus rubra*
Waxmyrtle	*Myrica cerifera*
Water tupelo	*Nyssa aquatica*
Black tupelo	*Nyssa sylvatica*
American sycamore	*Platanus occidentalis*
Eastern cottonwood	*Populus deltoides*
Black cherry	*Prunus serotina*
Swamp laurel oak, Diamond leaf oak	*Quercus laurifolia*
Overcup oak	*Quercus lyrata*
Water oak	*Quercus nigra*
Willow oak	*Quercus phellos*
Post oak	*Quercus stellata*
Live oak	*Quercus virginiana*
Black willow	*Salix nigra*
Elderberry	*Sambucus canadensis*
Chinese tallowtree	*Sapium sebiferum*
Sweetleaf	*Symplocos tinctoria*
Baldcypress	*Taxodium distichum*
Winged elm	*Ulmus alata*
American elm	*Ulmus americana*
Hercules club	*Zanthoxylum clava-herculis*

Appendix 1. Scientific names based on Hardin et al. (2001) and Miller and Miller (1999).

APPENDIX 2. WOODY SPECIES RECORDED AT DRIFT FENCE ARRAYS AND AVIAN POINT COUNT STATIONS AT BEN'S CREEK WMA, LA

Common Name	Scientific Name
Red maple	*Acer rubrum* var. *rubrum*
Red buckeye	*Aesculus pavia*
Devil's-walkingstick	*Aralia spinosa*
Pawpaw	*Asimina triloba*
Dwarf pawpaw	*A. parvifolia*
Eastern baccharis	*Baccharis halimifolia*
French mulberry	*Callicarpa americana*
Hickory	*Carya* spp.
Chinkapin	*Castanea pumila*
Flowering dogwood	*Cornis florida*
Hawthorn	*Crataegus* spp.
Persimmon	*Diospyros virginiana*
American beech	*Fagus grandifolia*
Huckleberry	*Gaylussacia* spp.
Witch-hazel	*Hamamelis virginiana*
St. John's-wort	*Hypericum* spp.
Large gallberry	*Ilex coriacea*
Gallberry	*I. glabra*
American holly	*I. opaca*
Yaupon	*I. vomitoria*
Chinese privet	*Ligustrum sinense*
Sweetgum	*Liquidambar styraciflua*
Yellow-poplar	*Liriodendron tulipifera*
Japanese honeysuckle	*Lonicera japonica*
Southern magnolia	*Magnolia grandiflora*
Bigleaf magnolia	*M. macrophylla*
Sweetbay magnolia	*M. virginiana*
Southern crabapple	*Malus angustifolia*
Wax myrtle	*Myrica cerifera*
Black tupelo	*Nyssa sylvatica*
Eastern hophornbeam	*Ostrya virginiana*
Sourwood	*Oxydendrum arboreum*
Loblolly pine	*Pinus taeda*
Black cherry	*Prunus serotina*
White oak	*Quercus alba*
Southern red oak	*Q. falcata*
Bluejack oak	*Q. incana*
Blackjack oak	*Q. marilandica*
Swamp chestnut oak	*Q. michauxii*
Water oak	*Q. nigra*
Nutall oak	*Q. texana*
Post oak	*Q. stellata*

Appendix 2. Scientific names based on Hardin et al. (2001), Miller and Miller (1999).

APPENDIX 2. continued.

Common Name	Scientific Name
Winged sumac	*Rhus copallina*
Chinese tallowtree	*Sapium sebiferum*
Sassafras	*Sassafras albidum*
Sweetleaf	*Symplocos tinctoria*
Tree-huckleberry	*Vaccinium arboreum*
Elliott blueberry	*V. elliottii*
Deerberry	*V. stamineum*
Arrow-wood	*Viburnum dentatum*

APPENDIX 3. WOODY SPECIES RECORDED AT DRIFT FENCE ARRAYS AND AVIAN POINT COUNT STATIONS AT SANDY HOLLOW WMA, LA

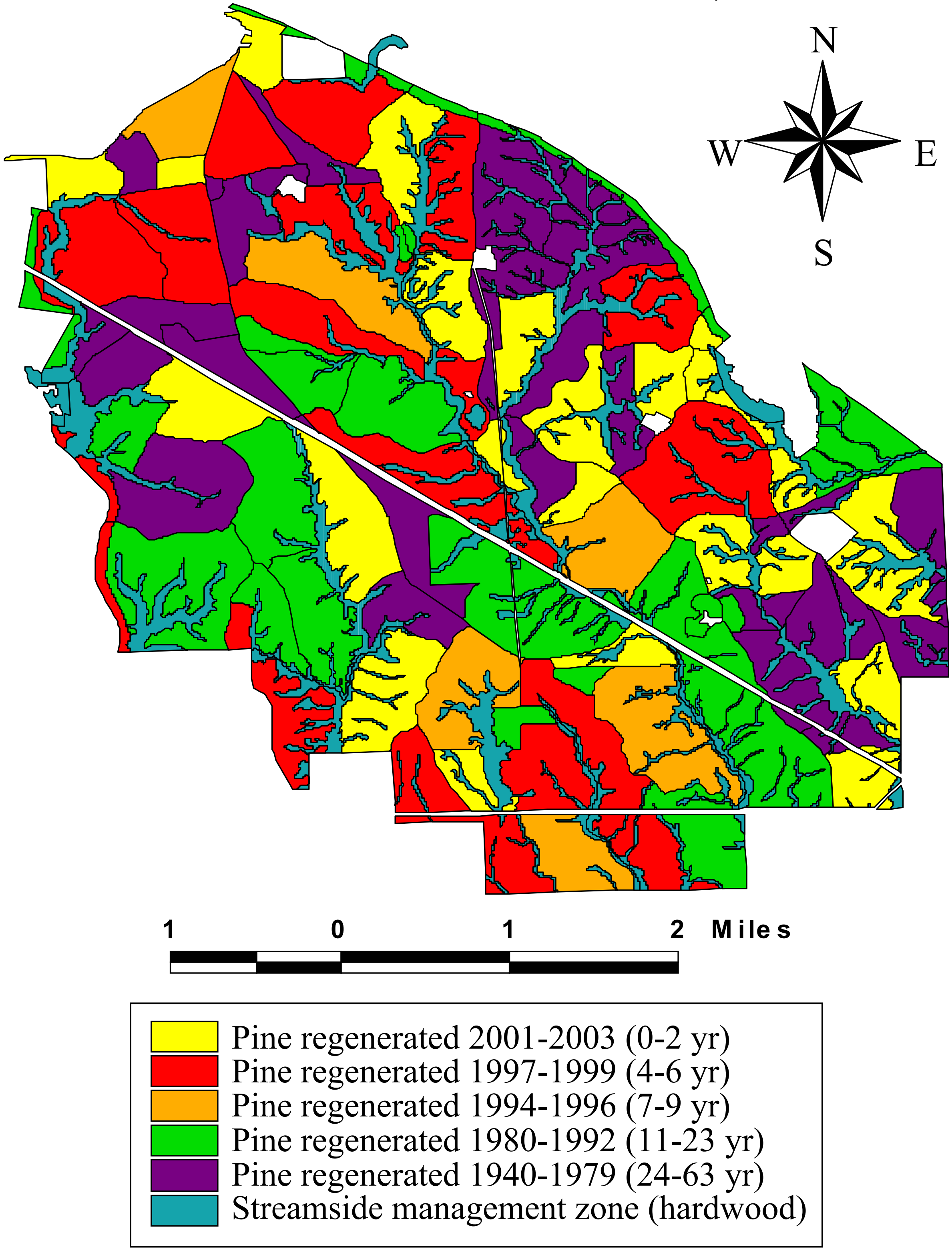
Common Name	Scientific Name
Red maple	*Acer rubrum*
Devil's-walkingstick	*Aralia spinosa*
Eastern baccharis	*Baccharis halimifolia*
French mulberry	*Callicarpa americana*
Bitternut hickory	*Carya cordiformis*
Chinkapin	*Castanea pumila*
Flowering dogwood	*Cornus florida*
Hawthorn	*Crataegus* spp.
Persimmon	*Diospyros virginiana*
American beech	*Fagus grandifolia*
Huckleberry	*Gaylussacia* spp.
Honeylocust	*Gleditsia triacanthos*
Witch-hazel	*Hamamelis virginiana*
St. John's-wort	*Hypericum* spp.
Large gallberry	*Ilex coriacea*
Gallberry	*I. glabra*
American holly	*I. opaca*
Yaupon	*I. vomitoria*
Chinese privet	*Ligustrum sinense*
Sweetgum	*Liquidambar styraciflua*
Southern magnolia	*Magnolia grandiflora*
Sweetbay magnolia	*M. virginiana*
Southern crabapple	*Malus angustifolia*
Waxmyrtle	*Myrica cerifera*
Black tupelo	*Nyssa sylvatica*
Shortleaf pine	*Pinus echinata*
Longleaf pine	*P. palustris*
Loblolly pine	*P. taeda*
Virginia pine	*P. virginiana*
Black cherry	*Prunus serotina*
White oak	*Quercus alba*
Southern red oak	*Q. falcata*
Laurel oak	*Q. hemispherica*
Bluejack oak	*Q. incana*
Blackjack oak	*Q. marilandica*
Swamp chestnut oak	*Q. michauxii*
Water oak	*Q. nigra*
Cherry bark oak	*Q. pagoda*
Willow oak	*Q. phellos*
Post oak	*Q. stellata*
Winged sumac	*Rhus copallina*
Chinese tallowtree	*Sapium sebiferum*

Appendix 3. Scientific names based on Hardin et al. (2001) and Miller and Miller (1999).

APPENDIX 3 continued.

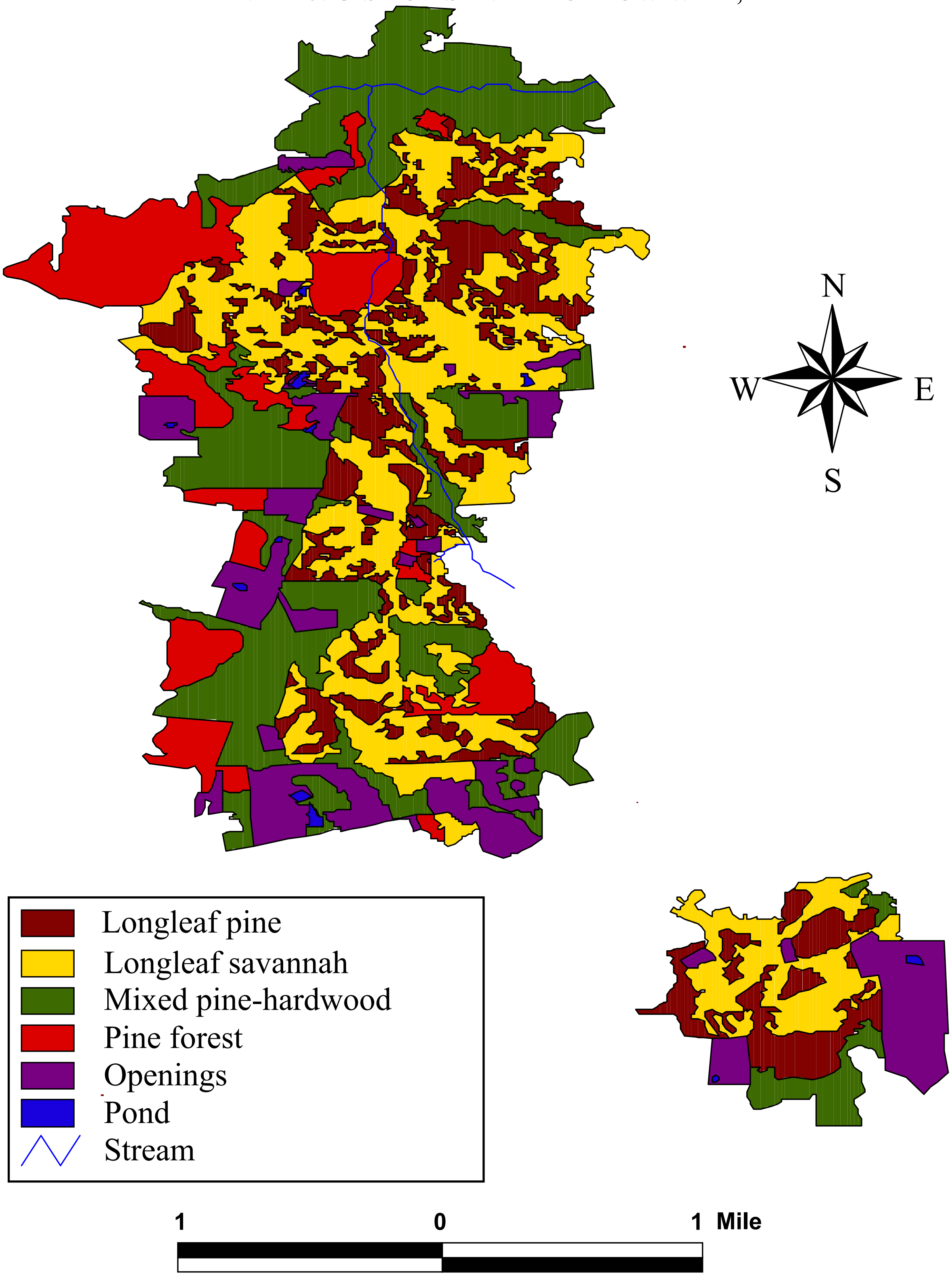
Common Name	Scientific Name
Sassafras	*Sassafras albidum*
Poison oak	*Toxicodendron pubescens*
Winged elm	*Ulmus alata*
Huckleberry, blueberry	*Vaccinium* spp.
Arrow-wood	*Viburnum dentatum*

APPENDIX 4. GIS FOR BEN'S CREEK WMA, LA



Appendix 4. GIS landcover delineation by habitat type and regeneration age of loblolly pine (*Pinus taeda*) stands at Ben's Creek WMA, LA, 2003.

APPENDIX 5. GIS FOR SANDY HOLLOW WMA, LA



Appendix 5. GIS landcover delineation by habitat type at Sandy Hollow WMA, LA, based on 1998 DOQQs.

VITA

Holly LeGrand was born in Wilmington, North Carolina, on December 5, 1974, to Jacqueline and Vernon Hoyle. At the age of 4, she moved with her family to Douglas, Georgia, where she resided until graduation from high school in 1993. After graduation, Holly moved to Winston-Salem, North Carolina, where she attended Salem College between August 1993 and May 1997, earning a Bachelor of Science degree in biology, with a minor in mathematics. While earning her degree, Holly also studied primates, tracked wolves in Minnesota and Wisconsin, and collected plant specimens in a Costa Rican rainforest.

After graduating from Salem College, Holly was employed as a field research technician on a variety of research projects that examined effects of silviculture on wildlife. Across the southeast, she worked with bats in bottomland hardwood swamps, small mammals, amphibians, and reptiles in upland pine forests, and breeding birds in the southern region of the Appalachian Mountains. In August 2002, Holly began graduate work with the School of Renewable Natural Resources at Louisiana State University, Baton Rouge. Her research involved monitoring communities of amphibians, reptiles, and breeding birds in managed forests, including a bottomland hardwood swamp, longleaf pine-savannah, and loblolly pine plantation, in central and northeast Louisiana. Holly will be awarded the Master of Science in wildlife in December 2005.