RESPONSE OF AVIAN AND ARTHROPOD COMMUNITIES TO NATIVE GRASS RESTORATION IN CENTRAL GEORGIA, USA

by

ANGELA BARBARA MCMELLEN

(Under the Direction of Sara H. Schweitzer)

ABSTRACT

North American grassland birds are declining more rapidly than any other group of birds. Reasons for decline include habitat fragmentation, alteration, and destruction. The southeastern United States provides both breeding and winter habitat for grassland birds. I investigated responses of bird communities during breeding and winter seasons, and summer arthropod communities to current land management practices (annual mowing and periodic burning), and experimental native grass restoration. Study plots encompassed the spectrum of available habitat, from open agricultural fields to pine stands with low basal area and grassy understory.

Native warm season grasses were established successfully in an open, agricultural landscape and 1.2–4 ha forest openings. Restored plots had tall, distinct bunches of native grass and little shrub cover. No differences were detected between breeding bird communities of restored and control plots. Winter bird abundance was highly variable between years, but consistently greater in restored plots.

Native warm season grass plots supported substantial numbers of shrub-scrub and forest-disturbance birds during breeding seasons, and grass-herb and shrub-scrub birds during winters. Shrub-scrub birds were negatively correlated with forest cover in a 1-km buffer around plots during breeding and winter seasons. Grass-herb birds were positively correlated with grass cover during breeding and winter seasons. No relationship was detected between grass-herb birds and forest cover during breeding seasons; a negative relationship was detected during winter. Open woodlands had the highest total conservation value during summer and winter, but differences were not significant, suggesting all plots contributed conservation value to the system.

Managing for native grass and forb cover may increase arthropod abundance. Arthropod abundance and family richness were greatest in plots with little canopy cover and much grass and forb cover. Contrary to expectations, fire ants (*Solenopsis invicta*) did not negatively impact arthropod abundance or richness. Orthopteran abundance was lowest in mowed treatments. Araneae were abundant in all but the densest forest treatment.

One management prescription will not maintain a diverse community of early successional birds in the Southeast. A mixture of grassland types, including native grasslands, is desirable across the landscape.

INDEX WORDS: Georgia, grassland birds, native warm season grass restoration, short distance migrants, arthropods, winter bird communities

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DEDICATION

To my tripod,

Mom, Dawn, and Ted,

for all the love and support

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

STATUS OF EARLY SUCCESSIONAL BIRDS

Since the 1960s, grassland birds have underdone steeper and more consistent population declines than any other group of birds (Knopf 1994). Forty percent of North American birds dependent on a frequently disturbed ecosystem experienced a significant decline between 1969 and 1998 (Brawn et al. 2001). Breeding Bird Survey data from the eastern U.S. revealed significant population declines for 13 of 14 grasslands birds while only 2 of 40 forest birds had declining trend (Peterjohn and Sauer 1999, Askins 2000). Grassland birds had the smallest proportion of increasing birds of any bird group (Peterjohn and Sauer 1999). Fire suppression, declining beaver populations, urbanization, aforestation, short rotation pine plantations, intensive agriculture, and introduction of exotic pasture grasses have all been implicated in the decline of eastern disturbance dependent birds (Askins 2000). In the late nineteenth century, 55-60% of the eastern United States was young, open forest, but that percentage has dropped to less than 20% (Lorimer 2001).

There has been a concerted research effort to stem the decline of grassland birds in the Great Plains states. The utility of the farmland landscape for grassland bird conservation has been of particular interest. The Natural Resources Conservation Service (NRCS) has numerous programs that give farmers subsidies, encouraging them to take marginally productive and erodible land out of production. The original intent of the program was to conserve soil and water resources, but wildlife biologists quickly realized that the land could prove beneficial to wildlife (Hays and Farmer 1990, Dunn et al. 1993, Hall and Willig 1994). Benefits to breeding grassland birds have been found in Texas (Berthelsen and Smith 1995), Kansas (Granfors et al. 1996), and Iowa (Patterson and Best 1996). More recent literature has moved beyond the farm to look at the landscape surrounding early successional habitat and the impacts of grassland fragmentation on breeding grassland birds (Bajema and Lima 2001, Coppedge et al. 2001a, Coppedge et al. 2001b, Robinson et al. 2001, Bakker et al. 2002, Pons et al. 2003, Shriver et al. 2004, Brotons et al. 2005).

Research in the eastern United States has lagged behind, hindered by questions of the validity of preserving and managing for a habitat type considered unnatural by some. There is an impression that the eastern United States was contiguous forest from the eastern seaboard to the Mississippi River at the time of European contact (Denevan 1992) and that all early successional habitat in the East resulted from colonial actions. Historic disturbance regimes are difficult to establish for some parts of the East, but archaeological evidence suggests that fire was common in some regions (Lorimer 2001). In the east, fires were likely started by lightening as well as Native Americans. Archeological evidence suggests that fire and disturbance near extensive Native American settlements provided large patches of early successional habitat (Doolittle 1992).

Several species probably expanded their range into the eastern United States after extensive forest clearing by European colonists (Askins 2000), but early successional habitat has been a part of the eastern landscape for thousands of years.

The presence of three distinct sub-species of grassland birds in the East would argue against a recent human origin of early successional habitat (Askins 1999). Savannah sparrows (*Passerculus sandwichensis*), Henslow's sparrows (*Ammodramus henslowii*), and greater-prairie chickens (*Tympanuchus cupido*) each have distinct eastern subspecies. The eastern subspecies of the greater-prairie chicken has gone extinct probably due to habitat loss (Askins 2000).

EARLY SUCCESSIONAL HABITAT IN THE SOUTHEAST

Early successional habitat has been a major component of the southeastern landscape until recently. Longleaf pine (*Pinus palustris*) woodlands once covered over 30 million ha from southern Virginia to eastern Texas (Van Lear et al. 2005). Longleaf pine woodlands are a fire dependent ecosystem of widely spaced pine trees and a botanically diverse understory of plants from tall-grass prairies and Appalachian balds (Vogl 1973). In his travels through the Southeast in the 1700s, William Bartram described pine parklands, grassy openings, and large cane (*Arundinaria* spp.) breaks (Harper 1998). These areas provided abundant habitat for many grassland birds in the southeastern United States. Southeastern grasslands were not pristine when the first Europeans arrived in the fifteenth century. Native Americans had modified the landscape through centuries of fire use, cultivation, and other activities (Denevan 1992).

Native southeastern grasslands did not evolve with disturbance from intensive grazing. Livestock and intense grazing pressure arrived with the Europeans. By the mid-1700s, large herds of free-ranging cattle began to damage the wire grass (*Aristida stricta*) community within pine savannas and by the 1840s, range conditions were degraded by overgrazing (Gray 1933). Southeastern grasslands have undergone centuries of modification, including exclusion of fire, intensive grazing, and the introduction of cultivated, exotic, sod-forming grasses, that have resulted in an increase in hardwood trees and shrubs, changes in herbaceous species composition, and loss of native grasses and the near extirpation of native warm-season species such as switch grass (*Panicum virgatum*), big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), Indian grass (*Sorghastrum nutans*), and eastern gamma grass (*Tripsacum dactyloides*) (Rasnake 1992). Currently, most southeastern pastures are planted in introduced cool and warm-season grass species such as fescue (*Festuca arundinacea*), bermuda (*Cynodon dactylon*), and bahia grass (*Paspalum notatum*). Only about 1.2 million ha of longleaf pine forest remains in scattered patches in the Southeast (Van Lear et al. 2005).

Exotic grass pastures may be good for widespread species, but grassland generalists are rarely found within these pastures (DeVault et al. 2002). Several songbird species are closely associated with the structure of native bunchgrass-forb communities that formed the understory of the extensive longleaf pine and slash pine (*Pinus elliotti*) forests of the past. Partners in Flight identified the Henslow's sparrow (*Ammodramus henslowii*), Bachman's sparrow (*Aimophila aestivalis*), red-cockaded woodpecker (*Picoides borealis*), and prairie warbler (*Dendroica discolor*) as priority species in the pine savanna and grassland ecosystems of the Southeast (Cooper 1999). Other species warranting attention include the loggerhead shrike (*Lanius ludovicianus*), northern bobwhite (*Colinus virginianus*), eastern meadowlark (*Sturnella magna*), and savannah sparrow (*Passerculus sandwichensis*).

MANAGEMENT OF A DISTURBANCE SYSTEM

Grasslands are disturbance dependent ecosystems. Without some type of disturbance, succession will transform a grassland into a shrubland and eventually forested habitat. The fragmented patchwork of grassland habitat in the landscape means it is no longer possible to let natural disturbance maintain the system. Land managers must actively manage open habitat. Fire, grazing, and mowing are the most common types of grassland disturbance.

The literature suggests that avian response to disturbance management is species specific and varies with intensity of the disturbance. Vesper and savannah sparrows responded positively to fire disturbance in Arizona (Gordon 2000a), but eastern meadowlarks are not found in fields until 1-2 years after burning. Richness of grassland birds (Jansen et al. 1999) and abundance of ground-feeding insectivorous birds (Soderstrom et al. 2001) are negatively correlated with grazing intensity. Timing and intensity of mowing can impact breeding bird success (Dale et al. 1997), occupancy (Horn and Koford 2000), and survival (Broyer 2003).

Prescribed burning is the most common management technique for creating and maintaining early successional habitat in the southeastern United States and has been used and researched extensively since the 1960s when the first fire ecology conference was held at Tall Timbers Research Station (Komarek 1962). Even with 40 years of fire management in the Southeast, declines of eastern early successional birds continue. Three broad themes have emerged in the literature to explain the continuing decline of early successional birds: food resource availability, habitat structure and surrounding landscape, and the importance of wintering habitat.

AVIAN FOOD RESOURCE AVAILABILITY IN GRASSLANDS

Invertebrate abundance and diversity should be considered a top priority in avian management (Robel et al. 1995). Arthropods are a key component of avian diets during the spring and summer. Both chicks and older birds rely on this protein rich food source to fuel energetic demands of rapid growth and reproduction, respectively. If food resources are limited, adult birds may face a trade-off between annual survival and reproductive success (Martin 1995). Availability of arthropods to provision nestlings may directly limit nestling survival (Moreby 2004).

Birds can use their mobility to track fluctuating resources and distribute themselves according to resource abundance (Hurlbert and Haskell 2003). Breeding bird density has been linked to arthropod abundance in early succesional ecosystems (Zimmerman 1992, Howe et al. 2000, Shochat et al. 2005a). Grassland birds in an Oklahoma study nested in higher densities in plots with high arthropod abundance (Shochat et al. 2005a). The link between birds and arthropod abundance in early successional habitat has been the focus of much research in Europe. A review of 27 years of insect and bird data in Scotland found a link between the decline of insect species and bird species (Benton et al. 2002). The declines in birds and arthropods were the result of changing land management practices. Over the last 40 years, arthropod abundance and the birds that rely on arthropods during the breeding season have declined in Britain with modification and intensification of land management practices (reviewed in Barker 2004). Barker (2004) suggests that the declines of arthropods are an important cause of the decline of farmland birds in the British Isles.

Arthropod communities can reflect differences in landscape level factors as well as plant communities (Jonas et al. 2002). Different management strategies can impact the availability, abundance, and size of arthropods (McCraken et al. 2004). Changing the structure of vegetation modifies avian foraging behavior through changes in prey detectability, accessibility, and bird mobility (Butler and Gillings 2004). Plant species composition may be more important than structure in shaping arthropod communities in early successional habitat. Diversity and richness of arthropods in Kansas were positively correlated to plant diversity and richness (Jonas et al. 2002). A recent study of Texas rangelands found higher arthropod abundance in areas with native grasses (Flanders et al. 2006). Native prairie had the highest arthropod abundance when compared to native and non-native conservation reserve program (CRP) fields (McIntyre and Thompson 2003).

Changes in vegetation or disturbance regime can change the abundance and type of arthropods available to birds. The abundance (Anderson et al. 1989) and biomass (Shochat et al. 2005b) of arthropods are greater immediately after fire. Arthropod abundance in Oklahoma was higher on plots that were burned or grazed than on plots that received no management (Shochat et al. 2005a).

Land management practices can also negatively impact arthropods. Intensively managed grasslands had more widespread, less specialist arthropod species (DiGiulio et al. 2001). Abundance and diversity of invertebrates on British farmland declined on farms with management practices that decreased the number and diversity of forbs (Atkinson et al. 2004).

Fire ants are a serious problem in the southeastern United States but their impacts on invertebrate and vertebrate communities are still under investigation (Porter and Savignano 1990, Jusino-Atresino and Phillips 1994, Wojcik 1994, Wojcik et al. 2001, Vogt et al. 2002). Fire ants negatively impact invertebrates through direct mortality and competition for food and habitat (Wojcik et al. 2001). In central Texas, areas infested with fire ants had 40% fewer arthropod species and 75% fewer individuals than areas without fire ants (Porter and Savignano 1990). It is unclear what impact fire ants are having on arthropod food resources for birds.

IMPACT OF GRASSLAND STRUCTURE AND LANDSCAPE

The structure of grasslands can have an impact on avian occupancy (Pons et al. 2003), reproductive success (Dale et al. 1997, Shochat et al. 2005a), and food resource availability (Bock et al. 1996, McCraken et al. 2004). Titeux et al (2004) suggest that local environmental factors drive species assemblages across a wide-range of habitats, but other studies have found species specific response to scale (Morimoto and Wasserman 1991, Robinson et al. 2004). The bird community of a pitch pine-scrub oak forest, similar in physical structure to the pine-oak savannahs of the Southeast, was determined by environmental variables at the plot, patch, and landscape scale (Grand and Cushman 2003). Shrub-scrub bird occupancy was influenced by landscape level features and grassland bird occupancy by grass cover in Idaho (Knick and Rotenberry 1995).

Landscape context of early successional habitat can determine its utility (Coppedge et al 2001a, Shochat et al 2005a). Grasshopper sparrows (Kobal et al. 1999, Bakker et al. 2002), savannah sparrows (Herkert 1994, Kobal et al. 1999) and eastern meadowlarks (Granfors et al. 1996, Walk and Warner 2000) are considered area sensitive in the Midwest during the breeding season. These species tend to avoid breeding on small, isolated grassland patches.

Distribution of patches of grassland habitat within the landscape may be just as important as the quantity of grassland habitat within the landscape (Gordon 2000b). Grassland birds respond to a fine scale (0.9-16.5 ha) mosaic of grassland types (Pons et al. 2003). Increasing heterogeneity in the farmland landscape through edges, fallow areas, and different types of fields can make the landscape more hospitable to breeding (Benton et al. 2002) and wintering (Smith et al. 2005) early successional birds. Habitat preferences of individual species influence the spatial structure of grassland communities (Mikami and Kawata 2002, Pons et al. 2003). Grassland birds select different types and ages of grasslands. Eastern meadowlarks require older grasslands without recent severe disturbance (Wiens 1969, Granfors et al. 1996, Kobal et al. 1999). Grasshopper sparrows prefer sparse vegetation with bare earth patches (Arguedas-Negrini 2001) and intermediate litter layer (Powell 2006).

Increasing cover of forest within the landscape negatively impacts the abundance and occupancy of grassland birds in some areas (King et al. 2001, Ribic and Sample 2001, Berg 2002). An increase of 10-20% forest cover in Britain changed the avian community composition (Berg 2002). Shrub-scrub birds in Idaho also responded negatively to forest cover in the landscape during the summer (Knick and Rotenberry 2000). Increasing woody plant cover in grasslands decreases the suitability of habitat for grassland birds (O'Leary and Nyberg 2000, Coppedge et al. 2001a). Reducing woody ground cover within southern pine parklands was beneficial to grassland birds of concern (Conner et al. 2002).

It is unknown whether the same negative relationships between woody habitat and occupancy of early successional species that has been found in the Midwest will be found in the Southeast. Early successional habitat in the Southeast exists in a spectrum from open, agricultural land to grassy understories of open-canopy woodland. Many eastern birds may be defined as disturbance-dependent rather than early successional or grassland birds because they are not restricted to a single habitat type, but exist across a spectrum ranging from grasslands to gaps in mature forests (Hunter et al. 2001).

WINTER AVIAN ECOLOGY

Winter ecology and habitat use are research priorities for conservation of early successional birds (Vickery et al. 2000, Vickery and Herkert 2001). Winter is an often neglected part of the annual cycle of birds. Little research has focused on wintering birds within the United States (but see Grzybowski 1982, Gordon 2000, Tucker 2003).

Research suggests that avian community indices are more closely tied to vegetation in the winter than during the breeding season (Anderson et al. 1983), but research defining the habitat needs of winter birds is lacking. Birds that are grassland specialists during the breeding season exhibit this habitat specificity on the winter grounds birds as well (Igl and Ballard 1999). The highest density of over-wintering sparrows in Southern Texas was found in shrub-grasslands (Igl and Ballard 1999). In Oregon, sparrow diversity was highest in areas with low vegetation complexity and recent disturbance (Patterson 2002).

Abundance of winter birds may be highly variable temporally and spatially (Gordon 2000a, 2000b). Wintering sparrows are short distance migrants (Root 1988) moving only as far as necessary to acquire necessary food resources and escape inclement weather. Wintering grassland birds select sites that have high seed density (Robinson et al. 2004) to fuel the high energetic demands of winter. Wintering bird distribution may also be driven by weather (French and Picozzi 2002).

The Southeast provides both breeding and wintering habitat to early successional birds. Many early successional birds are short distance migrants which winter in the southern United States and northern Mexico (Root 1988). In the Southeast, the winter habitat needs of the declining Henslow's sparrow have received attention (Plentovich et al. 1998, 1999; Tucker and Robinson 2003), but there are still large gaps in our knowledge.

RESTORATION AS A POSSIBLE SOLUTION

Eastern grasslands are fire dependent systems and most management has focused on re-introducing fire. Reintroduction of the necessary disturbance may not be enough to restore native vegetation if the local seed bank is impoverished and seed dispersal is limited by a fragmented landscape (Kindscher and Tieszen 1998, Bakker and Berendse 1999). Restoration is increasingly important in regions with little quality habitat (Vickery et al. 1999) or where habitat change can not be remedied with less intense management alternatives (Briggs et al. 2005) and is considered a priority for grassland bird research (Brennan and Kuvlesky 2005). The Partners in Flight migratory bird program set habitat objectives for the South Atlantic Coastal Plain region that include restoring or converting cool-season grass (e.g. fescue) pastures to native warm-season grasslands on more than 4 million ha (Hunter et al. 2001b).

The largest grasslands in the eastern United States are reclaimed coal-mines that are providing habitat for large populations of Henslow's sparrows (Bajema et al. 2001). Native warm season grass (NWSG) restoration has had positive impacts on breeding bird communities in Texas (Flanders et al. 2006) and Pennsylvania (Giuliano and Daves 2002). Abundance, richness, and nesting success of birds was greater in the NWSG fields in Pennsylvania (Giuliano and Daves 2002). In Texas, richness did not differ between native and exotic prairies, but breeding birds were more abundant in native grass (Flanders et al. 2006). Information on the impacts of restoration on breeding and wintering bird communities in the Southeast is lacking.

PURPOSE OF THE RESEARCH

Early successional birds in the southeastern United States have received little attention in the literature (but see Plentovich et al. 1998, 1999; Marcus et al. 2000). Research and restoration in the Southeast has focused on habitat needs of the endangered red-cockaded woodpeckers and the northern bobwhite. Management for the red-cockaded woodpecker has had a positive impact on shrub and grassland associated birds (Conner et al. 2002). In a study in Mississippi, many species of management concern were more common in pine-grassland restoration areas than traditional forest management areas in the southeastern United States (Wood et al. 2004), but several grassland associated birds in the Southeast are still declining and warrant further investigation.

The purpose of our research was to fill in some of the gaps in current knowledge about early successional bird species in the southeastern United States. We wanted to look at current management alternatives in the Southeast as well as a novel technique: native grass restoration. We hypothesized that native warm season grasses (NWSG) could be established successfully in old field habitat consisting primarily of exotic pasture grasses. Further, we believed that restored habitat would differ in vegetative community indices and structure from old field habitat. We hypothesized that the differences in vegetation community and structure in restored grasslands would positively impact the bird community by: 1) supporting a more diverse community, 2) supporting more birds, and 3) providing more conservation benefit during both the breeding and winter seasons.

Because breeding birds are heavily dependent on arthropods for food, we also wanted to examine arthropod community responses to the same management practices. We hypothesized that arthropod richness and abundance would be greatest in treatments with the greatest native grass and forb cover. We also hypothesized that fire ant (*Solenopsis invicta*) abundance would negatively impact the richness and abundance of other arthropods.

Early successional habitat in the Southeast exists in a spectrum from open, agricultural land to the grassy understory of open woodlands. Many eastern birds can be better defined as disturbance-dependent rather than grassland birds. These bird species are not restricted to a single habitat type, but exist across a spectrum ranging from open grasslands to gaps in mature forests (Hunter et al. 2001a). The impact of a tree component to early successional habitat on the avian community is not known for the Southeast. To examine this question, we looked at early successional habitat from open fields to the grass understory of a pine forest. Contrary to research in the Midwest, we hypothesized that abundance of breeding populations of southeastern early successional birds would not be negatively impacted by forest cover in the landscape. We hypothesized short-distance migrants that winter in the Southeast would show a negative relationship with forest cover.

LITERATURE CITED

Anderson, B. W., R. D. Ohmart, and J. Rice. 1983. Avian and vegetation community structure and their season relationships in the lower Colorado River valley. Condor 85:392-405.

Anderson, R.C., T. Leahy, and S.S. Dhillion. 1989. Numbers and biomass of selected insect groups on burned and unburned sand prairie. American Midland Naturalist 122:151-162.

Arguedas-Negrini, N. 2001. Distribution, habitat and behavior of grasshopper sparrows, *Ammodramus savannarum* (Passeriformes: Emberizidae) in northwestern Nicaragua. Revista de Biologia Tropical 49:703-708.

Askins, R. A. 1999. History of grassland birds in Eastern North America. Studies in Avian Biology 19:60-71.

Askins, R.A. 2000. Restoring North America's birds: lessons from landscape ecology. Second edition. New Haven: Yale University Press.

Atkinson, P.W., D. Buckingham, and A.J. Morris. 2004. What factors determine where invertebrate-feeding birds forage in dry agricultural grasslands? Ibis 146:99-107.

Bajema, R. A., T. L. DeVault, P. E. Scott, and S. L. Lima. 2001. Reclaimed coal mine grasslands and their significance for Henslow's Sparrows in the American Midwest. The Auk 118:422-431.

Bajema, R. A., and S. L. Lima. 2001. Landscape-level analyses of Henslow's Sparrow (*Ammodramus henslowii*) abundance in reclaimed coal mine grasslands. American Midland Naturalist 145:288-298.

Bakker, J. P., and F. Berendse. 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. Tree 14:63-67.

Bakker, K. K., D. E. Naugle, and K. F. Higgins. 2002. Incorporating landscape attributes into models for migratory grassland bird conservation. Conservation Biology 6:1638-1646.

Barker, A.M. 2004. Insects as food for farmland birds: is there a problem? Pages 37-50 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

Benton, T.G., D.M. Bryant, L. Cole, and H.Q.P. Crick. 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. Journal of Applied Ecology 39:673-687.

Berg, A. 2002. Composition and diverstiy of bird communities in Swedish farmland-forest mosaic landscapes. British Trust for Ornitholgy 49:153-165.

Berthelsen, P. S., and L. M. Smith. 1995. Nongame bird nesting on CRP lands in the Texas Southern High Plains. Journal of Soil and Water Conservation 50:672-675.

Brawn, J.D., S.K. Robinson, and F.R. Thompson, III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32:251-276.

Brennan, L. A., and W. P. Kuvlesky. 2005. North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1-13.

Briggs, J. N., A. K. Knapp, J. M. Blair, J. L. Heisler, G. A. Hoch, M. S. Lett, and J. K. McCarron. 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55:243-254.

Brotons, L., A. Wolff, G. Paulus, and J.L. Martin. 2005. Effect of adjacent agricultural habitat on the distribution of passerines in natural grasslands. Biological Conservation 124:407-414.

Broyer, J. 2003. Unmown refuge areas and their influence on the survival of grassland birds in the Saone valley (France). Biodiversity and Conservation 12:1219-1237.

Butler, S. J., and S. Gillings. 2004. Quantifying the effects of habitat structure on prey detectability and accessibility to farmland birds. Ibis 146:123-130.

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 114:324-352.

Cooper, R. J. 1999. Partners in Flight bird conservation plan for the southern Piedmont (physiographic area 11). The Plains, VA: American Bird Conservancy.

Coppedge, B.R., D.M. Engle, S.D. Fuhlendorf, R. E. Masters, and M. S. Gregory. 2001a. Landscape cover type and pattern dynamics in fragmented southern Great Plains grasslands, USA. Landscape Ecology 16:677-690.

Coppedge, B. R., D.M. Engle, R. E. Masters, and M. S. Gregory. 2001b. Avian response to landscape change in fragmented southern Great Plains grasslands. Ecological Applications 11:47-59.

Dale, B. C., P. A. Martin, and P. S. Taylor. 1997. Effects of hay management on grassland songbirds in Saskatchewan. Wildlife Society Bulletin 25:616-626.

Denevan, W. M. 1992. The pristine myth: the landscape of the Americas in 1492. Annals of the Association of American Geographers 82:369-385.

DeVault, T.L., P.E. Scott, R.A. Bajema, and S.L. Lima. 2002. Breeding bird communities in reclaimed coal-mine grasslands in the American Midwest. Journal of Field Ornithology 73:268-275.

DiGiulio, M., P.J. Edwards, and E. Meister. 2001. Enhancing insect diversity in agricultural grasslands: the roles of management and landscape structure. Journal of Applied Ecology 38:310-319.

Doolittle, W. E. 1992. Agriculture in North America on the eve of contact: a reassessment. Annals of the Association of American Geographers 82:386-401.

Dunn, C. P., F. Stearns, G. R. Guntenspergen, and D. M. Sharpe. 1993. Ecological benefits of the Conservation Reserve Program. Conservation Biology 7:132-139.

Flanders, A.A., W.P. Kuvlesky, D.C. Ruthven, R.E. Zaiglin, R.L. Bingham, T.E. Fulbright, F. Hernandez, and L.A. Brennan. 2006. Effects of invasive exotic grasses on South Texas rangeland birds. The Auk 123:171-182.

French, D. D., and N. Picozzi. 2002. 'Functional groups' of bird species, biodiversity and landscapes in Scotland. Journal of Biogeography 29:231-259.

Giuliano, W. M., and S. E. Daves. 2002. Avian response to warm-season grass use in pasture and hayfield management. Biological Conservation 106:1-9.

Gordon, C. E. 2000a. Fire and cattle grazing on wintering sparrows in Arizona grasslands. Journal of Range Management 53:384-389.

Gordon, C.E. 2000b. Movement patterns of wintering grassland sparrows in Arizona. The Auk 117:748-759.

Grand, J. and S. A. Cushman. 2003. A multi-scale analysis of species-environment relationships: breeding birds in a pitch pine-scrub oak (*Pinus rigida-Quercus ilicifolia*) community. Biological Conservation 112:307-317.

Granfors, D. A., K. E. Church, and L. M. Smith. 1996. Eastern Meadowlarks nesting in rangelands and Conservation Reserve Program fields in Kansas. Journal of Field Ornithology 67:222-235.

Gray, L. C. 1933. HIstory of agriculture in the southern United States to 1860. Volume II. Baltimore: Waverly Press, Inc.

Hall, D. L. and M. R. Willig. 1994. Mammalian species composition, diversity, and succession in Conservation Reserve Program grasslands. The Southwestern Naturalist 39:1-10.

Hays, R. L. and A. H. Farmer. 1990. Effects of the CRP on wildlife habitat: emergency haying in the Midwest and pine plantings in the Southeast. Transactions of the North American Wildlife and Natural Resources Conference 55:30-39.

Herkert, J. R. 1994. The effects of habitat fragmentation on midwestern grassland bird communities. Ecological Applications 4:461-471.

Horn, D. J. and R. R. Koford. 2000. Relation of grassland bird abundance to mowing of Conservation Reserve Program fields in North Dakota. Wildlife Society Bulletin 28:653-659.

Howe, F.P., R.L. Knight, L.C. McEwen, and T.L. George. 2000. Diet switching and food delivery by shrubsteppe passerines in response to an experimental reduction in food. Western North American Naturalist 60:139-154.

Hunter, W.C., D.A. Buehler, R.A. Canterbury, J.L. Confer, and P.B. Hamel. 2001a. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin 29:440-455.

Hunter, W. C., L. H. Peoples, and J. A. Collazo. 2001. Partners in Flight conservation plan for the South Atlantic Coastal Plain. U.S. Fish and Wildlife Service.

Hurlbert, A. H., and J. P. Haskell. 2003. The effect of energy and seasonality on avian species richness and community composition. The American Naturalist 161:83-97.

Igl, L. D., and B. M. Ballard. 1999. Habitat associations of migrating and overwintering grassland birds in Southern Texas. The Condor 101:771-782.

Jansen, R., R. M. Little, and T. M. Crowe. 1999. Implications of grazing and burning of grasslands on the sustainable use of francolins (*Francolinus* spp.) and on overall bird conservation in the highlands of Mpumalanga province, South Africa. Biodiversity and Conservation 8:587-602.

Jonas, J.L., M.R. Whiles, and R.E. Charlton. 2002. Aboveground invertebrate responses to land management differences in a central Kansas grassland. Environmental Entomology. 31:1142-1152.

Jusino-Atresino, R. and S.A. Philips, Jr. 1994 Impact of red imported fire ants on the ant fauna of central Texas. Pages 258-269 In D.F. Williams, editor. Exotic ants: biology, impact, and control of introduced species. Boulder, CO: Westview Press.

Kindscher, K., and L. L. Tieszen. 1998. Floristic and soil organic matter changes after five and thirty-five years of native tallgrass prairie restoration. Restoration Ecology 6:181-196.

King, D. I., R. M. Degraaf, and C. R. Griffin. 2001. Productivity of early successional shrubland birds in clearcuts and groupcuts in an eastern deciduous forest. Journal of Wildlife Management 65:345-350.

Knick, S. T., and J. T. Rotenberry. 1995. Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. Conservation Biology 9:1059-1071.

Knick, S. T., and J. T. Rotenberry. 2000. Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. Ecology 8:220-227.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247-257.

Kobal, S. N., N. F. Payne, and D. R. Ludwig. 1999. Habitat/area relationships, abundance, and composition of bird communities in 3 grassland types. Transactions of the Illinois State Academy of Science 92:109-131.

Komarek, E. V., Sr., editor. 1962. Proceedings of the First Tall Timbers Fire Ecology Conference. Tall Timbers Research Station, Tallahassee, FL.

Lorimer, C. G. 2001. Historical and ecological roles of disturbance in Eastern North American forests: 9,000 years of change. Wildlife Society Bulletin 29:425-439.

Marcus, J. F., W. E. Palmer, and P. T. Bromley. 2000. The effects of farm field borders on overwintering sparrow densities. Wilson Bulletin 112:517-523.

Martin, T. E. 1995. Avian life history evolution in relation to nest sites, nest predation, and food. Ecological Monographs 65:101-127.

McCraken, D.I., E.M. Bigmal, S. Blake, and G.N. Foster. 2004. Productivity and profitability: the effects of farming practices on the prey of insectivorous birds. Pages 75-87 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

McIntyre, N.E. and T.R. Thompson. 2003. A comparison of conservation reserve program habitat plantings with respect to arthropod prey for grassland birds. American Midland Naturalist 150:291-301.

Mikami, O. K. and M. Kawata. 2002. The effects of individual interactions and habitat preferences on spatial structure in a grassland bird community. Ecography 25:200-214.

Moreby, S.J. 2004. Birds of lowland arable farmland: the importance and indentification of invertebrate diversity in the diet of chicks. Pages 21-35 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Pubishers.

Morimoto, D. C. and F. E. Wasserman. 1991. Dispersion pattersn and habitat associations of rufous-sided towhees, common yellowthroats, and prairie warblers in the southeastern Massachusetts pine barres. The Auk 108:264-276.

O'Leary, C. H., and D. W. Nyberg. 2000. Treelines between fields reduce the density of grassland birds. Natural Areas Journal 20:243-249.

Patterson, M. 2002. Analysis of bird point count data for the Mill Pond Site, Neawanna Ecological Observatory, Seaside, Clatsop Co., Oregon. Oregon Birds 28:168-171.

Patterson, M. P. and L. B. Best. 1996. Bird abundance and nesting success in Iowa CRP fields: the importance of vegetaion structure and composition. The American Midland Naturalist 135:153-167.

Peterjohn, B. G., and J. R. Sauer. 1999. Population status of North American grassland birds from the North American Breeding Bird Survey, 1966-1996. Studies in Avian Biology 19:27-44.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1998. Site fidelity of wintering Henslow's Sparrows. Journal of Field Ornithology 69:486-490.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1999. Habitat requirements of Henslow's Sparrows wintering in silvicultural lands of the Gulf Coastal Plain. The Auk 116:109-115.

Pons, P., B. Lambert, E. Rigolot, and R. Prodon. 2003. The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape. Biodiversity and Conservation 12:1843-1860.

Porter, S.D. and D.A. Savignano. 1990. Invasion of polygyne fire ants decimates native ants and disrupts arthropod community. Ecology 71:2095-2106.

Rasnake, M. 1992. Management of warm season bunch grasses. Proceedings of the Southern Pasture and Forage Crop Improvement Conference 48:38-39.

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

Robel, R.J., B.M. Press, B.L. Henning, K.W. Johnson, H.D. Blocker, and K.E. Kemp. 1995. Nutrient and energetic characteristics of sweepnet collected invertebrates. Journal of Field Ornithology 66:44-53.

Robinson, R. A., J. D. Hart, J. M. Holland, and D. Parrott. 2004. Habitat use by seed-eating birds: a scale dependent approach. Ibis 146:87-98.

Robinson, R. A., J. D. Wilson, and H.Q.P. Crick. 2001. The importance of arable habitat for farmland birds in grassland landscapes. Journal of Applied Ecology 38:1059-1069.

Root, T. 1988. Atlas of wintering North American birds: an analysis of Christmas Bird Count data. Chicago: The University of Chicago Press.

Shochat, E., M.A. Patten, D.W. Morris, D.L. Reinking, D.H. Wolfe, and S.K. Sherrod. 2005a. Ecological traps in isodars: effects of tallgrass prairie management on bird nest success. Oikos 111:159-169.

Shochat, E., D.H. Wolfe, M.A. Patten, D.L. Reinking, and S.K. Sherrod. 2005b. Tallgrass prairie management and bird nest success along roadsides. Biological Conservation 121:399-407.

Shriver, W. G., T. P. Hodgman, J. P. Gibbs, and P. D. Vickery. 2004. Landscape context influences salt marsh bird diversity and area requirements in New England. Biological Conservation 119:545-553.

Smith, M. D., P. J. Barbour, W. Burger, Jr., and S. J. Dinsmore. 2005. Density and diversity of overwintering birds in managed field borders in Mississippi. Wilson Bulletin 117:258-269.

Soderstrom, B., T. Part, and E. Linnarsson. 2001. Grazing effects on between-year variation of farmland bird communities. Ecological Applications 11:1141-1150.

Titeux, N., M. Dufrene, J.P. Jacob, M. Paquay, and P. Defourny. 2004. Multivariate analysis of a fine-scale breeding bird atlas using a geographical information system and partial canonical correspondence analysis: environmental and spatial effects. Journal of Biogeography 31:1841-1856.

Tucker, J. W., Jr., and W. D. Robinson. 2003. Influence of season and frequency of fire on Henslow's sparrows (Ammodramus Henslowii) wintering on Gulf Coast pitcher plant bogs. The Auk 120:96-106.

Van Lear, D. H., W. D. Carroll, P. R. Kapeluck, and R. Johnson. 2005. History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk. Forest Ecology and Management 211:150-165.

Vickery, P. D., J. Herkert, R., F. L. Knopf, J. Ruth, and C. E. Keller. 2000. Grassland birds: an overview of threats and recommended management strategies. Pages 74–77 in R. Bonney, D. N. Pashley, R. J. Cooper, and L. Niles, editors. Strategies for bird conservation: the Partners in Flight planning process. Proceedings of the 3rd Partners in Flight workshop (1995). Proceedings RMRS-P-16. U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah.

Vickery, P. D., and J. R. Herkert. 2001. Recent advances in grassland bird research: where do we go from here? The Auk 118:11-15.

Vickery, P. D., P. L. Tubaro, J. M. Cardoso da Silva, B. G. Peterjohn, J.R. Herkert, and R. B. Cavalcanti. 1999. Conservation of grassland birds in the western hemisphere. Studies in Avian Biology 19:2-26.

Vogl, R.J. 1973. Fire in the Southeastern grasslands. Proceedings of the Tall Timbers Fire Ecology Conference 12:178-193.

Vogt, J.T., R.A. Grantham, E. Corbett, S.A. Rice, and R.E. Wright. 2002. Dietary habitats of *Solenopsis invicta* (Hymenoptera: Formicidae) in four Oklahoma habitats. Environmental Entomology 31:47-53.

Walk, J. W. and R. E. Warner. 2000. Grassland management for the conservation of songbirds in the Midwestern USA. Biological Conservation 94:165-172.

Wiens, J. A. 1969. An approach to the study of ecological relationships among grassland birds. Ornithological Monographs 8:1-93.

Wojcik, D.P. 1994. Impact of the red imported fire ants on native ant species in Florida. Pages 269-281 *in* D.F. Williams, editor. Exotic ants: biology, impact, and control of introduced species. Boulder, CO: Westview Press.

Wojcik, D.P., C.R. Allen, R.J. Brenner, E.A. Forys, D.P. Jouvenaz, and R.S. Lutz. 2001. Red imported fire ants: impact on biodiversity. American Entomologist 47:16-23.

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

Zimmerman, J.L. 1992. Density-independent factors affecting the avian diversity of the tallgrass prairie community. Wilson Bulletin 104:85-94.

CHAPTER 2

NATIVE GRASS RESTORTION IN GEORGIA:

IS IT A VIABLE MANAGEMENT STRATEGY FOR BREEDING AND WINTERING EARLY SUCCESSIONAL BIRDS?[1]

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[1]McMellen, A.B. and S.H. Schweitzer. To be submitted to *Journal of Wildlife Management.*

ABSTRACT

Southeastern early-successional habitat has experienced large-scale conversion to high-intensity agriculture, pine plantations, and exotic grass pastures. Many native grasses have virtually been eliminated. Birds that depend on grassland communities for breeding and/or wintering habitat have experienced precipitous declines in the eastern United States. We established 12 plots, 1.2–4.0 ha each, in 2 locations within the Piedmont region of central Georgia to evaluate native grass re-establishment and subsequent enhancement of grassland-associated songbird communities. Six plots were planted with combination of switch grass (*Panicum virgatum*), big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), and Indian grass (*Sorghastrum nutans*) during spring 2002 (planted treatment). Six plots were maintained under traditional management of annual mowing and periodic burning (control treatment). Vegetation measurements were taken during spring 2002-2004 and winter 2003-2004 to estimate success of native grass re-establishment and to quantify vegetative differences between experimental and control plots. Breeding bird use of planted and control plots was monitored using constant effort mist netting and point counts during each spring, 2002-2004, and constant effort mist netting during winter 2003-2004. By 2004, planted plots had more native grass cover, taller grass, and less shrub cover than control plots. Estimates of avian abundance, richness, diversity, and total conservation value (TCV) from mist netting and point counts during the breeding season did not differ between planted and control treatments. Point count data provided larger indices than mist net data. Winter avian diversity and richness did not differ between treatments. Planted plots had higher TCV in 2003 and 2004. Capture rates declined between 2003 and 2004 in both treatments. Overall abundance and sparrow abundance was greater in planted plots in 2003, but not in 2004. Landscape-level management is recommended for enhancing habitat for the diverse, declining species in the Southeast. Our data suggest that re-establishment of native warm season grasses will provide positive conservation benefit for over-wintering, early successional birds.

INTRODUCTION

Since the 1960s, grassland birds have underdone steeper and more consistent population declines than any other group of birds (Knopf 1994). Forty percent of North American birds dependent on a frequently disturbed ecosystem experienced a significant decline between 1969 and 1998 (Brawn et al. 2001). Breeding Bird Survey data from the eastern U.S. revealed significant population declines for 13 of 14 grasslands birds while only 2 of 40 forest birds had declining trend (Askins 2000). Grassland birds had the smallest proportion of increasing birds than any group of birds (Peterjohn and Sauer 1999). Fire suppression, declining beaver populations, urbanization, aforestation, short rotation pine plantations, intensive agriculture, and introduction of exotic pasture grasses have all been implicated in the decline of eastern grassland birds (Askins 2000).

In his travels through the Southeast in the 1700s, William Bartram described pine parklands, grassy openings, and large cane breaks (Harper 1998). These areas provided abundant habitat for many grassland birds in the southeastern United States. Southeastern grasslands were not pristine when the first Europeans arrived in the fifteenth century. Native Americans had modified the landscape through centuries of fire use, cultivation, and other activities (Denevan 1992). Vegetation in the Georgia Piedmont was dominated by fire-tolerant species before European settlement (Cowell 1998).

Native southeastern grasslands did not evolve with disturbance from intensive grazing. Livestock and intense grazing pressure arrived with the Europeans. By the mid-1700s, large herds of free-ranging cattle began to damage the wire grass (*Aristida stricta*) community within pine savannas and by the 1840s, range conditions were degraded by overgrazing (Gray 1933). Southeastern grasslands have undergone centuries of modification, including exclusion of fire, intensive grazing, and the introduction of cultivated, exotic, sod-forming grasses, that have resulted in an increase in hardwood trees and shrubs, changes in herbaceous species composition, and loss of native grasses and the near extirpation of native warm-season grass (NWSG) species such as switch grass (*Panicum virgatum*), big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), Indian grass (*Sorghastrum nutans*), and eastern gamma grass (*Tripsacum dactyloides*) (Rasnake 1992). Currently, most southeastern pastures are now planted in introduced cool and warm-season grass species such as fescue (*Festuca arundinacea*), bermuda (*Cynodon dactylon*), and bahia grass (*Paspalum notatum*). Exotic grass pastures may be good for widespread species, but grassland specialists are rarely found within these pastures (DeVault et al. 2002).

Several songbird species are closely associated with the structure of native bunchgrass-forb communities that formed the understory of the extensive longleaf pine (*Pinus palustris*) and slash pine (*P. elliotti*) forests of the past. Partners in Flight identified the Henslow's sparrow (*Ammodramus henslowii*), Bachman's sparrow

(*Aimophila aestivalis*), red-cockaded woodpecker (*Picoides borealis*), and prairie warbler (*Dendroica discolor*) as priority species in the pine savanna and grassland ecosystems of the Southeast (Cooper 1999). Other species warranting attention include the loggerhead shrike (*Lanius ludovicianus*), northern bobwhite (*Colinus virginianus*), eastern meadowlark (*Sturnella magna*), and savannah sparrow (*Passerculus sandwichensis*).

Early successional birds in the Southeastern United States have received little attention in the literature (see Plentovich et al. 1998, 1999; Marcus et al. 2000). Research and restoration in the Southeast has focused on habitat needs of the endangered red-cockaded woodpeckers. Management for the red-cockaded woodpecker has had a positive impact on shrub and grassland associated birds (Conner et al. 2002). Many species of management concern were more common in pine-grassland restoration areas than traditional forest management areas in the southeastern United States (Wood et al. 2004), but several grassland associated birds in the Southeast are still declining and warrant further investigation.

The Southeast provides both breeding and wintering habitat to early successional birds. Many early successional birds are short distance migrants that winter in the southern United States and northern Mexico (Root 1988). Winter is an often neglected part of the annual cycle of birds. Little research has focused on wintering grassland birds within the United States (Grzybowski 1982, 1983; Gordon 2000a, 2000b; Tucker and Robinson 2003). In the Southeast, wintering ecology of the declining Henslow's sparrow has received some attention (Plentovich et al. 1996, 1998;

Tucker and Robinson 2003), but large gaps still exist in our knowledge of wintering ecology.

Priorities for grassland bird research include multiple-species management, replacement of exotic grasses with native grasses, and wintering ecology (Brennan and Kuvlesky 2005). The Partners in Flight migratory bird program set habitat objectives for the South Atlantic Coastal Plain region that include restoring or converting cool-season grass (e.g., fescue) pastures to native warm-season grasslands on more than 10 million acres (4 million hectares; Hunter et al. 2001). Eastern grasslands are fire dependent systems and most management has focused on re-introducing fire. Reintroduction of the necessary disturbance may not be enough to restore native vegetation if the local seed bank is impoverished and seed dispersal is limited by a fragmented landscape (Kindscher and Tieszen 1998, Bakker and Berendse 1999).

The purpose of this study was to examine the vegetative and avian community responses to native grass restoration in Georgia. We hypothesized that native warm season grasses (NWSG) could be established successfully in old field habitat consisting primarily of exotic pasture grasses. Further, we believed that restored habitat would differ in vegetative community indices and structure from old field habitat. We hypothesized that the differences in vegetation community and structure in restored grasslands would positively impact the bird community by 1) supporting a more diverse community, 2) supporting more birds, and 3) providing more conservation benefit during the breeding and winter seasons.

STUDY AREAS

We conducted field work at Piedmont National Wildlife Refuge (NWR) and Joe Kurz Wildlife Management Area (WMA) in central Georgia. Both locations are in the transition zone from Piedmont to Coastal Plain physiographic areas and are within the historic range of the pineland savannah habitat (Vogl 1973). Piedmont NWR is located in Jones and Jasper Counties and Joe Kurz WMA is located in Meriwether County.

Piedmont NWR is approximately 14,000 ha of upland pine habitat managed by the U.S. Fish and Wildlife Service. According to data from the nearest University of Georgia (UGA) climate station (Eatonton, GA), average daily temperature was 16.9 C during the study and average annual precipitation was 1200 mm with 2 peaks, 1 each in March and July. Soils were Davidson clay loam with a 6-25% slope. Davidson clay loam is a deep, well drained soil with moderate permeability (National Resources Conservation Service 2005). The landscape was ≥95% forested consisting of a loblolly pine (*Pinus taeda*) canopy with an understory of flowering dogwood (*Cornus florida*), redbud (*Cercis canadensis*), and various native and non-native grasses. The bottomlands were mixed hardwood dominated by oaks (*Quercus* spp.). The endangered red-cockaded woodpecker was the primary management focus. Large areas of the refuge were burned during winter on a 3-5 year rotation in an effort to create and maintain open pine parklands, the preferred foraging habitat of this woodpecker. Six 1.2-4-ha forest openings at Piedmont NWR were used as study plots. Originally, forest openings were established as food plots for white-tailed deer (*Odocoileus virginianus*), eastern wild turkey (*Meleagris gallopavo*), and other wildlife, but have been fallow since the 1970s and managed as early successional habitat using annual mowing and periodic burning.

Hence, forest openings were old field habitat consisting of grasses, forbs, shrubs, and scattered saplings. Dominant grasses were bahia, broomsedge (*Andropogon virginicus*), Johnson grass (*Sorghum halpense*), and fescue. Blackberry (*Rubus* spp.) was the most common woody plant. Most saplings were loblolly pine and sweetgum (*Liquidambar styraciflua*).

Joe Kurz WMA was a privately held farm until it was acquired by the Georgia Department of Natural Resources in 1995. Soils were Appling loamy sand and Cecil sandy loam and there was a 2-10% slope (National Resources Conservation Service 2005). Both soil types were deep and well drained with a clayey subsoil and moderate permeability. During the study, average annual precipitation was 1289 mm with 2 peaks of rainfall, 1 each in March and July, and the mean daily temperature was 16.0 C according to the nearest UGA climate center (Williamson, GA). About 70-75% of the surrounding landscape was forested with row crop agricultural, pasture, and hay fields accounting for most of the remaining land cover in the area. Joe Kurz WMA included approximately 1,495 ha of old field habitat, upland loblolly pine, and bottomland oaks. Vegetative structure within our 6, 1-2 to 4 ha study plots was old field habitat similar to the plots at Piedmont NWR.

At each site, for our planted study plots, we chose the 3 largest plots previously selected by the management agency for native grass restoration activities. Three control plots were then selected such that they were similar in size, shape, and landscape composition to the planted study plots. Control plots were maintained under the agency's traditional management of annual late-summer mowing and periodic winter burning (3-5 year cycle). None of the study plots was closer than 1 km, thus the study plots were considered independent.

METHODS

Native Grass Establishment

All study plots were mowed during August-September 2001, then vegetation was allowed to grow for 1-2 weeks before Roundup® herbicide (isopropylamine salt of glyphosate) was applied to the plots selected for NWSG planting. All study plots were burned during winter 2001-2002 to suppress woody vegetation and remove dead plant material. NWSG plots were tilled and planted in April 2002. A Truax™ (Truax Co. Inc., Minneapolis, MN) native grass seed drill was used to plant seeds at a 0.6 cm depth. A mixture of switch grass, big bluestem, little bluestem, and Indian grass was planted at a rate of 7.8 kg pure live seed/ha.

In August 2002, we applied a 2% solution of Roundup® with back-pack sprayers in NWSG plots to patches of exotic grass or weeds while carefully avoiding native grasses. We applied Plateau® (imazapic ammonium salt) at a rate of 0.9L/ha to 2 NWSG study plots with large patches of Johnson grass. Plateau® was applied using a broadcast sprayer pulled by a tractor.

Evaluation of Grass Establishment and Vegetation Structure

We determined success of native grass restoration and development of vertical structure through summer and winter vegetation measurements. Summer measurements were taken in July to minimize impact on nesting birds. All vegetation measurements, except shrub cover, were taken in 1-m^2 square, permanent sample plots. Sample plots were distributed through each study plot in a grid-like fashion ≥10m from the study plot edge and ≥40m from other sample plots. Vegetation measurements were recorded from permanent sample plots during July 2002, 2003, and 2004.

We identified and recorded all species of forbs, grasses, and woody plants within each sample plot. Percentage of total grass cover and native grass cover was estimated visually using a modified Braun-Blanquet cover scale (Braun-Blanquet 1932, Table 2.1). Litter depth and grass height were measured in the center of each sample plot. A cover board was used to estimate vertical vegetation density in the first 0.5-m vegetation strata. The cover board was placed in the center of the sample plot and an observer recorded measurements at a distance of 10 m in each cardinal direction. The measurements were averaged to determine a value for each sample plot. In 2004, we estimated shrub cover using the line intercept method on 20 20-m transects in each sample plot. During winters, we only measured vertical vegetation density within permanent sample plots.

Assessment of the Avian Community

Breeding season--We sampled the bird community using mist nets from late April to early July 2002-2004 (UGA AUCP Permit A2002-10095). Every 10-14 days, 12-m long mist nets with 36-mm mesh were erected in each plot. In 2002, 2 nets were used in each plot, but in 2003 and 2004, 4 nets were used. Net locations from 2002 were maintained and 2 additional locations added for the new nets. Nets were arranged in a V formation to maximize capture rates (Martin 1969). We conducted 6 rounds of mist netting in each study plot. Nets were opened at sunrise and closed 6 hours later or when the temperature reached 29 C. Species, weight, wing chord, bill length, tail length, sex, breeding status, and age were recorded for each bird captured. In addition, each bird was banded with a uniquely numbered USGS aluminum leg-band for future identification (BBL Permit #22746).

Breeding birds within study plots were also sampled using fixed-radius point counts (Hutto et al. 1986, Hamel et al. 1996). One point count station was established in the center of each plot and visited 3 times each breeding season for 1 10-min point count. All point counts were conducted between 0.5 hour after sunrise and 0930 hours. Point counts were only done in clear weather with low winds to maximize detection ability. All birds seen or heard within the 50-m radius point count were recorded. Birds that flew over the point count station were recorded, but not used in the analysis; only singing males were used in analyses. Because all study plots were visited equally, total detections for the 3 visits per season were used in analyses as it yields a result comparable to averaging (Nur et al. 1999). Due to problems with data collection methodology, 2002 point count data for Joe Kurz WMA were not used in analyses.

Winter season-Mist netting was also conducted to assess avian community indices in study plots during January-March 2003 and 2004. Every 10-14 days, 4 12-m long mist nets with 36 mm mesh were erected in each plot for a total of 12 nets per treatment. Nets were erected at the same location during each sampling period. We conducted 3 rounds of mist netting in each study plot each year. Nets were opened at sunrise and closed after 6 hours, but were not opened on days with adverse weather such as extreme cold, rain, or high winds. Species, weight, wing chord, bill length, tail length, sex, and age were recorded for each bird captured. In addition, each bird was banded with a uniquely numbered USGS aluminum leg-band for future identification.

Statistical Analysis

Community indices estimated from mist net and point count data were diversity, richness, total conservation value, and abundance. In addition, an index of abundance of sparrows was calculated using the winter mist net data. We used the transformed Shannon diversity index (exp H') because it is better able to detect diversity differences between areas or treatments (Kempton and Taylor 1976). Richness was defined as the number of species detected in the study plot during all sampling periods within a season and was calculated separately for point count and mist net data during the summer.

To assess the overall avian conservation value of our treatments, we calculated total conservation value (TCV, Nuttle et al. 2003) for each study plot each year during both winter and breeding season. Partners in Flight (PIF) priority scores were used to give each bird species a rank from 0-4 (PIF rank) for breeding and winter seasons. Ranks were used instead of the summed PIF priority scores to avoid previously identified statistical problems with meaningful analysis of summed scores (Beissinger et al. 2000, Nuttle et al. 2003). Species PIF rank was multiplied by abundance (birds/100 net hours or birds/ha) for each species in the study plot, then these values were summed to find a final TCV for each study plot.

For breeding season and winter mist net data, we used capture rate (birds/100 net hours) as an index of abundance. Abundance from the point count data was calculated as birds/ha. To detect differences between control and planted study plots, we used a repeated measures analysis of variance with a randomized complete block design. Landscape (Piedmont NWR and Joe Kurz WMA) was the blocking factor. We used SAS (SAS Institute 2005) for all statistical analyses, and a $P \leq 0.05$ as our *a priori* level of significance.

RESULTS

Grass Establishment and Vegetation Structure

Treatment (control or planted) and time (year) effects on species richness of plant groups varied with the plant group under consideration. Forb species richness differed among years (F = 19.98, df = 2, 18, $P = 0.0008$) and there was no interaction between time and treatment (F = 3.17, df = 2, 18, $P = 0.0968$). Forb richness increased from 2002 to 2004 within both treatments (Table 2.2), but there were no treatment effects within year. We detected a time x treatment interaction in the ANOVA of grass species richness (F = 24.97, df = 2, 18, $P = 0.0004$). In 2002, we recorded nearly twice the number of grass species in control plots than in planted plots (F = 12.08, df = 2, 9, $P = 0.007$). In 2003, 30% more grass species were present in planted plots than in control plots (F = 4.70, df = 2, 9, $P = 0.05$), but there was no difference in grass species richness between planted and control plots in 2004 (F = 1.33, df = 2, 9, $P = 0.28$). Species richness of shrubs and trees did not differ between planted and control treatments (time x treatment: F = 2.25, df = 2, 18, $P = 0.17$) or change over time in either planted or control plots (time: F = 3.25, df = 2, 18, $P = 0.09$).

By the end of the study period, plots planted with native grasses had higher native grass cover, grass height, and vertical vegetation density and lower shrub cover. There was an interaction between time and treatment for total grass cover (F = 7.24, df = 2, 18, $P = 0.003$, Table 2.3). The mean grass cover score was greater in control plots (3.9 ± 0.8; 50-75%) than in planted plots (1.8 ± 1.0; 5-25%) in 2002, one growing season following treatment application (F = 15.81, df = 2, 9, *P* = 0.003). During the second and third growing seasons, mean grass cover scores did not different between control and planted study plots (Table 2.3). Native grass cover had a significant time x treatment effect (F = 3.9, df = 2, 18, *P* = 0.04, Table 2.3). Native grass did not differ between control (2.67 ± 1.99) and (2.79 ± 1.23) planted plots in 2002 (F = 0.00, df = 2, 9, *P* = 0.99, Table 2.3), but planted plots had significantly more native grass cover in 2003 (F = 4.21, df = 2, 9, *P* = 0.05, Table 2.3) and 2004 (F = 5.38, df = 2, 9, *P* = 0.03, Table 2.3). Litter depth did not differ among years (F = 1.19, df = 2, 18, *P* = 0.35) and there was no time x treatment interaction (F = 2.14, df = 2, 18, *P* = 0.18). In 2002, litter depth was greater in control plots (1.79 ± 1.57 cm) than in planted plots (0 ± 0 cm; F = 8.84, df = 2, 9, *P* = 0.02), but did not differ in 2003 (F = 0.02, df = 2, 9, *P* = 0.90) or 2004 (F = 0.15, df = 2, 9, *P* = 0.71). Because there was no treatment x time interaction (*F*=0.29, df = 2, 18, *P* = 0.604) in the ANOVA of grass height, we examined its change between years and detected a decline from 2003 to 2004 (F = 66.95, df = 2, 18, *P* ≤ 0.0001). Within each year, grass height was greater in planted study plots than in control plots (2003, F = 5.13, df = 2, 9, *P* = 0.05; 2004, F = 4.80, df = 2, 9, *P* = 0.05). There was an interaction between time and treatment for vertical vegetation density and a change over time (F = 8.30, df = 2, 18, *P* = 0.001 and F = 4.25, df = 2, 18, *P* = 0.02 respectively). Vertical vegetation density was higher in control study plots in 2002 (F = 6.49, df = 2, 9, *P* = 0.007, Table 2.3). There was no treatment effect in 2003, but by 2004, planted study plots had higher vertical vegetation density (F = 7.20, df = 2, 9, *P* = 0.02). Percent shrub cover in control plots (28.7 ± 19.4%) was about 3 times greater than planted plots (9.4 ± 5.7%, F = 13.29, df = 2, 9, *P* = 0.005) in 2004, 2 years after planting NWSGs.

Analysis of winter vegetation data revealed structural differences between treatments during 2004, but not 2003. There was an interaction between year and treatment in the ANOVA of winter vertical vegetation density (F = 10.33, df = 1, 9, *P* = 0.011). In 2004, vertical vegetation density was greater in planted plots (73.4 ± 8.6%) than in control plots (29.5 ± 33.1%; F = 15.61, df = 2, 9, *P* = 0.003), but there was no difference in vegetation density between treatments in 2003 (planted, 12.3 ± 5.1%; control, 25.0 ± 28.4%; F = 1.38, df = 2, 9, *P* = 0.27).

Assessment of the Avian Community

In 3,144.3 net hours of summer mist netting, 33 species of birds were captured. Twenty-six species were captured in control study plots and 23 species were captured in planted plots. Species abundances (birds/100 net hours) varied among years and between treatments (Table 2.4). Ten species, including the black-and-white warbler (*Mniotilta varia*), blue-gray gnatcatcher (*Polioptila caerulea*), eastern meadowlark, red-eyed vireo (*Vireo olivaceus*), and summer tanager (*Pirange rubra*), were only captured in control plots. The common yellowthroat (*Geothlypis trichas*), eastern bluebird (*Sialia sialias*), eastern phoebe (*Sayornis phoebe*), hooded warbler (*Wilsonia citrina*), northern bobwhite, savannah sparrow, and veery (*Catharus fuscescens*) were only captured in planted plots.

Diversity (F = 6.38, df = 2, 18, *P* = 0.02) and richness (F = 7.51, df = 2, 18, *P* = 0.02) changed over time, but there was no interaction between time and treatment for diversity (F = 6.38, df = 2, 18, *P* = 0.84) or richness (F = 0.36, df = 2, 18, *P* = 0.71, Table

2.5). Total conservation value changed over time (F = 5.31, df = 2, 18, *P* = 0.03), but there was no difference between treatments (F = 1.72, df = 2, 18, *P* = 0.24). Overall abundance did not change over time (F = 2.23, df = 2, 18, *P* = 0.17) or between treatments (F = 0.85, df = 2, 18, *P* = 0.46).

Forty-three species were recorded during breeding season point counts in control and planted plots (Table 2.6). Thirty-four species were detected in control plots and 36 species were identified in planted plots. Nine species, including the blue jay (*Cyanocitta cristata*), downy woodpecker (*Picoides pubescens*), red-shouldered hawk (*Melanerpes erthrocephalus*), and eastern meadowlark, occurred only in control plots. Grasshopper sparrow (*Ammodramus savannarum*), eastern bluebird, northern bobwhite, and eastern phoebe are among the 8 species that were only found in planted plots. Nineteen species were detected during point count surveys, but not during mist netting. Because point count data were only available for Piedmont NWR in 2002, we analyzed 2002 separately from the other years of data. In 2002, there was no difference in diversity, richness, TCV, or abundance between control and planted plots (Table 2.7). There were no time or treatment x time interaction effects in repeated measures analysis of point count data from 2003 and 2004 for diversity, richness, TCV, or abundance (Table 2.7).

Twenty-seven bird species were captured in 1,931.2 net hours of winter mist netting (Table 2.8). Twenty-two species were captured in control plots and 20 species in planted plots. The Carolina wren (*Thryothorus ludovicianus*), fox sparrow (*Passerella iliaca*), golden-crowned kinglet (*Regulus satrapa*), northern mockingbird (*Mimus polyglottos*), red-bellied woodpecker (*Melanerpes carolinus*), yellow-rumped warbler

(*Dendroica coronata*), and palm warbler (*Dendroica palmarum*) were only captured in control plots. The American kestrel (*Falco sparverius*), brown thrasher (*Toxostoma rufum*), ruby-crowned kinglet (*Regulus calendula*), vesper sparrow (*Pooecetes gramineus*), and white-throated sparrow (*Zonotrichia albicollis*) were only captured in planted plots.

Of the species that were captured in both treatments in 2003 (Table 2.8), Carolina chickadees (*Poecile carolinensis*), chipping sparrows, eastern bluebirds, field sparrows, savannah sparrows, and song sparrows (*Melospiza melodia*) were more abundant in planted plots. Pine warblers (*Dendroica pinus*) were more abundant in control plots. Field sparrows, pine warblers, and song sparrows were captured more frequently in planted plots in 2004. Tufted titmice and savannah sparrows were more abundant in control plots in 2004.

Winter diversity and richness did not change over time (diversity: F = 1.14, df = 1, 9, P = 0.31; richness F = 2.08, df = 1, 9, P = 0.18) nor was there a treatment effect (diversity: F = 0.76, df = 1, 9, P = 0.41; richness: F = 0.35, df = 1, 9, P = 0.56, Table 2.9). Total conservation value changed over time (F = 16.47, df = 1, 9, P = 0.003), but there was no effect treatment x time interaction (F = 2.78, df = 1, 9, P = 0.123). Total conservation value was higher in planted plots in 2003 (F = 5.02, df = 2, 9, P = 0.05) and 2004 (F = 4.46, df = 2, 9, P = 0.05). Mean abundance (birds/100 net hours) decreased from 2003 to 2004 (F = 23.32, df = 1, 9, P = 0.0009), but the decline was not consistent between treatments (F = 6.02, df = 1, 9, P = 0.04). Mean capture rate was nearly 3 times higher in planted plots in 2003 (F = 4.28, df = 2, 9, P = 0.05), but there was no difference between planted and control plots in 2004 (F = 1.53, df = 2, 9, P =

0.26). Sparrow abundance (sparrows/100 net hours) decreased between years ($F = 17.49$, df = 1, 9, $P = 0.002$), but the relationship between treatments changed over time (time x treatment, $F = 10.07$, df = 1, 9, $P = 0.01$). In 2003, more sparrows were captured in planted plots than control plots ($F = 6.26$, df = 2, 9, $P = 0.03$), but capture rates did not differ between treatments in 2004 ($F = 2.87$, df = 2, 9, $P = 0.12$).

DISCUSSION

Native warm season grasses can be successfully restored in old field habitat in central Georgia. The low species richness of grasses in planted plots in 2002 indicated that initial site preparation was effective in suppressing exotic pasture grasses. After the first growing season, planted plots were characterized by sparse native grass cover and low vegetation density. The grass seed we used for restoration came from Texas and Missouri because there were not sufficient seed resources available in Georgia or the surrounding states to plant any sizable area in NWSGs. Poor response during the first growing season led to concern that genotypic differences between grasses adapted to Georgia and those from several states away might preclude successful establishment, but these concerns were unfounded. Even with drought conditions during 2002 and seed not adapted to the local conditions, planted fields had over 50% native grass cover by the end of the second growing season far surpassing native grass cover in control plots.

The structure of grasslands can have an impact on avian occupancy (Pons et al. 2003), reproductive success (Dale et al. 1997, Shochat et al. 2005), and food resource availability (Bock et al. 1996, McCraken et al. 2004). As we hypothesized, control and planted plots differed in vegetative structure. Planted plots had tall, distinct bunches of grass. Grass in control plots was low to the ground and dense forming a thick, ground-level mat of vegetation. Grass cover and vertical vegetation density decreased over time in control plots and increased over time in planted plots. Grass cover decreased slightly between 2002 and 2004 in control plots, but nearly doubled in the planted plots between 2002 and 2003. In planted plots, vertical vegetation density increased from 34.2% in 2002 to 81.1% in 2004, further supporting our assessment of successful establishment of NWSGs. The end of a 4-year drought in Georgia occurred during the spring to summer 2003, during which time the study sites received twice the amount of rainfall that had fallen during 2002. The increased precipitation likely contributed to the strong response of planted grasses during 2003.

The extensive forest cover within the landscape of our study areas poses management challenges for maintaining early successional habitat. Increasing woody plant cover in grasslands decreases the suitability of habitat for grassland birds (O'Leary and Nyberg 2000, Coppedge et al. 2001). Many plant species disperse from the forest into adjacent grasslands. The current management prescription (control treatment) uses annual mowing and periodic burning to control the invasion of woody plant species. Although planted and control plots had a similar number of woody species throughout the study, the preparation required to establish native grasses effectively reduced woody vegetation in planted plots. In the third year of the study, shrub cover was <10% in planted plots while it was nearly 30% in control plots. We expect that the lack of shrubs and other woody vegetation in planted plots make these areas especially attractive to grassland associated birds, as was found in southern pine parklands, where reducing woody ground cover was beneficial to grassland birds of concern (Conner et al. 2002).

Contrary to what we expected, our assessment of the avian community during the breeding season did not detect benefit of NWSG establishment on diversity, richness, TCV, or abundance. Although avian community indices in planted plots increased from 2002 to 2004, the differences were not significant. Similar diversity and richness between treatments was not completely unexpected. Grassland ecosystems support far fewer species than forested ecosystems (Wiens 1973, 1974; Kobal et al. 1999) making detection of differences in richness and diversity more difficult.

Even though field size was not a predictor of bird density in other studies of grassland birds (Ribic and Sample 2001), our study plots (1.2-4.0 ha) may have been too small to support large populations of grassland birds, masking any subtle differences between treatments. Some species of management concern are area sensitive in other parts of their range. Area sensitive species generally avoid small, isolated patches of habitat occurring more frequently and breeding more successfully in large patches. Grasshopper sparrows (Kobal et al. 1999, Bakker et al. 2002) and eastern meadowlarks (Granfors et al. 1996, Walk and Warner 2000) are both considered area sensitive in the Midwest.

Small field sizes did not limit the benefit of NWSG restoration in Pennsylvania where fields ranged from 0.8-2.9 ha. Abundance, richness, and nesting success of birds was greater in the NWSG fields (Giuliano and Daves 2002). In Texas, richness did not differ between native and exotic prairies, but breeding birds were more abundant in native grass (Flanders et al. 2006). Why were differences not detected in Georgia?

Many of the grassland obligate birds that winter in Georgia head north to wintering grounds in the Midwest, the northern U.S., and southern Canada, leaving far fewer grassland obligate bird species in the Southeast during the spring and summer (Rising and Beadle 1996, Sibley 2000). Except for the Bachman's sparrow (Hammerson 1994), Georgia is not the core breeding habitat for any grassland bird, but does support breeding populations of grasshopper sparrow, eastern kingbird, northern bobwhite, and eastern meadowlark.

According to both mist net and point count data, Bachman's and grasshopper sparrows were more often encountered in planted plots. Eastern kingbirds were only found in planted plots in 2003 and 2004. Northern bobwhite were only found in planted plots. The eastern meadowlark is the only species of concern that was found more often in control plots. Eastern meadowlarks prefer dense grass, deep litter, and little bare ground (Wiens 1969) that develop when grassland has not had a recent, significant disturbance. The sparse grass, shallow litter, and large patches of bare ground that characterized the planted plots, especially the first season after planting, did not provide suitable habitat for eastern meadowlarks.

The planted plots also provided habitat for several shrub associated species. Chipping sparrows (*Spizella passerina*) and indigo buntings (*Passerina cyanea*) were more abundant in planted plots in all 3 summers. Field sparrows (*Spizella pusilla*) and prairie warblers (*Dendroica discolor*) were more abundant in control plots in 2002 and 2004, but they were more abundant in planted plots in 2003. Yellow-breasted chats (*Icteria virens*) were only found in control plots in 2002, but they were at least 3 times more abundant in planted plots than control plots in 2003 and 2004.

Breeding season community indices calculated from mist net data were generally lower than the same indices calculated from point count data (Table 2.5 and 2.7). Mist nets in grasslands are more visible making mist netting more difficult (Martin 1969). High visibility may have decreased capture rates and reduced the number of species captured in mist nets. Many of the species detected in point counts, but not in mist netting, were forest birds such as brown-headed nuthatch, downy woodpecker, and northern parula. The landscape at both study sites was dominated by forest cover. Black-and-white warblers (*Mniotilta varia*), summer tanagers (*Piranga rubra*), and other forest dwellers wandered into open habitat. Generally, they were observed foraging (McMellen upublished data). These species were most commonly detected near the wooded edges of the fields. Their foraging and breeding behavior probably precluded their inclusion in our mist net sample and accounted for differences in community metrics between breeding season mist net and point count data.

Research on winter ecology and habitat use are research priorities for grassland bird conservation (Vickery and Herkert 2001). This research contributes to the small, but growing body of knowledge about wintering grassland birds. Unlike during the breeding season, NWSG restoration positively impacted the wintering birds in Georgia. In 2003, we found the highest density of wintering birds and sparrows in the study plots we planted with NWSGs. Wintering sparrows may be attracted to recent disturbances. The highest density of over-wintering sparrows in Southern Texas was found in shrub-grasslands (Igl and Ballard 1999). In Oregon, sparrow diversity was highest in areas with low vegetation complexity and recent disturbance (Patterson 2002). Vesper and savannah sparrows, which were commonly observed in our plots, responded positively to fire disturbance in Arizona (Gordon 2000a). In our study, mean sparrow abundance was ≥5 times greater in planted fields than control fields in 2003. Because the planted study plots had only been planted the previous spring, these plots had low vertical vegetation density that may have been attractive to wintering sparrows.

Birds that are grassland specialists during the breeding season tended to exhibit this habitat specificity on the winter grounds birds as well (Igl and Ballard 1999). In this study, sparrow abundance was 5 times higher in planted fields during 2003. Birds can use their mobility to track fluctuating resources and distribute themselves according to resource abundance (Hurlbert and Haskell 2003). Wintering sparrows may have been attracted to planted plots due to increased seed availability. Annual plants dominate restored prairie areas for the first 5 years (Kindscher and Tieszen 1998) producing large quantities of seed. Fields enrolled in conservation reserve program (CRP) with a similar species composition to our planted fields had significantly more seed availability than nearby pastures (Klute et al. 1997). Wintering grassland birds select sites that have high seed density (Robinson et al. 2004). In addition to being more abundant, seeds may have been easier to acquire in planted fields because the low vegetation density during 2003 left large areas of bare earth. Ten of 14 bird species wintering in agricultural grasslands in England were associated with bare earth suggesting that avian foraging opportunities were better in those fields (Perkins et al. 2000).

Wintering bird abundance was highly variable from year to year. Abundance in 2004 was a quarter of what it had been in 2003. The distribution and abundance of wintering sparrows in Southeastern Arizona also varied significantly between years across the region (Gordon 2000a, 2000b). Winter bird abundance in the Virginia

Piedmont was much lower than in the breeding season (Childers et al. 1986). Wintering sparrows are short distance migrants (Root 1988) moving only as far as necessary to get the resources needed. Wintering bird distribution is driven by weather in Scotland (French and Picozzi 2002). The second winter of this study was very mild across large parts of the United States. Birds may not have had to move as far south to find food resources and escape inclement weather.

MANAGEMENT IMPLICATIONS

To assess the management potential of restoring native grasses versus maintaining current management, we calculated TCV (Nuttle et al. 2003) for control and planted plots. Land managers and management agencies can use avian conservation scores to quickly assess habitat quality (Wood et al. 2004). During the breeding season, there was no difference in TCV between planted and control plots in any year. Several species of management concern were found only in planted plots, but their low abundance did not impact TCV. Winter TCV was significantly higher in planted plots in 2003 and 2004. Overall captures rates declined by more than 75% between 2003 and 2004. Even with this decline in number of birds counted, TCV in planted plots was higher than in control plots in 2004. Savannah sparrows, vesper sparrows, and other high priority species were much more common in the planted plots. Avian community indices are more closely tied to vegetation in the winter (Anderson et al. 1983). Even given the small plot size, the planted plots seemed to provide real benefit to wintering sparrow populations. Management generally focuses on breeding season habitat requirements. Our data suggest that habitat manipulation for wintering birds can have positive conservation value.

Habitat preferences of individual species influence the spatial structure of grassland communities (Mikami and Kawata 2002, Pons et al. 2003). Grassland birds select different types and ages of grasslands. Eastern meadowlarks require older grasslands without recent severe disturbance (Wiens 1969). Grasshopper sparrows prefer sparse vegetation with bare earth patches (Arguedas-Negrini 2001). No single management prescription will provide habitat for both of these species or for all declining grassland-associated species. Landscape-level management of grassland resources is recommended for providing habitat for the variety of declining species. Distribution of patches of grassland habitat within the landscape may be just as important as the quantity of grassland habitat within the landscape (Gordon 2000b).

Grassland birds respond to a fine scale (0.9-16.5 ha) mosaic of grassland types (Pons et al. 2003). Patch sizes in our study were small, but patch size may not be the only determinant of occupancy. Landscape context of early successional habitat can determine its utility (Coppedge et al. 2001, Shochat et al. 2005). Land managers need to provide vegetative communities that are currently scarce (Wood et al. 2004). In the forest dominated landscape of central Georgia, early successional habitat is in short supply. The remaining habitat should be managed to maximize utility for breeding and wintering grassland birds.

LITERATURE CITED

Anderson, B. W., R. D. Ohmart, and J. Rice. 1983. Avian and vegetation community structure and their season relationships in the lower Colorado River valley. Condor 85:392-405.

Arguedas-Negrini, N. 2001. Distribution, habitat and behavior of grasshopper sparrows, *Ammodramus savannarum* (Passeriformes: Emberizidae) in northwestern Nicaragua. Revista de Biologia Tropical 49:703-708.

Askins, R. A. 2000. Restoring North America's birds: lessons from landscape ecology. Second edition. New Haven: Yale University Press.

Bakker, J. P. and F. Berendse. 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. Tree 14:63-67.

Bakker, K. K., D. E. Naugle, and K. F. Higgins. 2002. Incorporating landscape attributes into models for migratory grassland bird conservation. Conservation Biology 6:1638-1646.

Beissinger, S. B., J. M. Reed, J. M. Wunderle, Jr., S. K. Robinson, and D. M. Finch. 2000. Report of the AOU conservation committee on the Partners in Flight species prioritization plan. The Auk 117:549-561.

Braun-Blanquet, J. 1932. Plant sociology: the study of plant communities. New York: McGraw-Hill Book Company, Inc.

Brawn, J. D., S. K. Robinson, and F. R. Thompson, III. 2001. The Role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32:251-276.

Brennan, L. A., and W. P. Kuvlesky. 2005. North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1-13.

Childers, E. L., T. L. Sharik, and C. S. Adkisson. 1986. Effects of loblolly pine plantations on songbird dynamics in the Virginia Piedmont. Journal of Wildlife Management 1986:406-413

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 114:324-352.

Cooper, R. J. 1999. Partners in Flight bird conservation plan for the southern Piedmont (physiographic area 11). The Plains, VA: American Bird Conservancy.

Coppedge, B. R., D.M. Engle, S. D. Fuhlendorf, R. E. Masters, and M. S. Gregory. 2001. Landscape cover type and pattern dynamics in fragmented southern Great Plains grasslands, USA. Landscape Ecology 16:677-690.

Cowell, M. C. 1998. Historical change in vegetation and disturbance on the Georgia Piedmont. American Midland Naturalist 140:78-89.

Dale, B. C., P. A. Martin, and P. S. Taylor. 1997. Effects of hay management on grassland songbirds in Saskatchewan. Wildlife Society Bulletin 25:616-626.

Denevan, W. M. 1992. The pristine myth: the landscape of the Americas in 1492. Annals of the Association of American Geographers 82:369-385.

DeVault, T. L., P. E. Scott, R. A. Bajema, and S. L. Lima. 2002. Breeding bird communities of reclaimed coal-mine grasslands in the American midwest. Journal of Field Ornithology 73:268-275.

Flanders, A. A., W. P. Kuvlesky, D. C. Ruthven, R. E. Zaiglin, R. L. Bingham, T. E. Fulbright, F. Hernandez, and L. A. Brennan. 2006. Effects of invasive exotic grasses on South Texas rangeland breeding birds. The Auk 123:171-182.

French, D. D., and N. Picozzi. 2002. 'Functional groups' of bird species, biodiversity and landscapes in Scotland. Journal of Biogeography 29:231-259.

Giuliano, W. M., and S. E. Daves. 2002. Avian response to warm-season grass use in pasture and hayfield management. Biological Conservation 106:1-9.

Gordon, C. E. 2000a. Fire and cattle grazing on wintering sparrows in Arizona grasslands. Journal of Range Management 53:384-389.

Gordon, C.E. 2000b. Movement patterns of wintering grassland sparrows in Arizona. The Auk 117:748-759.

Granfors, D. A., K. E. Church, and L. M. Smith. 1996. Eastern Meadowlarks nesting in rangelands and Conservation Reserve Program fields in Kansas. Journal of Field Ornithology 67:222-235.

Gray, L. C. 1933. History of agriculture in the southern United States to 1860. Volume II. Baltimore: Waverly Press, Inc.

Grzybowski, J. A. 1982. Population structure in grassland bird communities during winter. The Condor 84:137-152.

Grzybowski, J. A. 1983. Patterns of space use in grassland bird communities during winter. The Wilson Bulletin 95:591-602.

Hamel, P. B., W. P. Smith, D. J. Twedt, J. R. Woehr, E. Morris, R. D. Hamilton, and R. J. Cooper. 1996. A land manager's guide to point counts of birds of the Southeast. USDA Forest Service General Technical Report SO-120.

Hammerson, G. 1994. Bachman's sparrow (*Aimophila aestivalis*) species management abstract. The Nature Conservancy.

Harper, F., editor. 1998. The travels of William Bartram. Athens, GA: The University of Georgia Press.

Hunter, W. C., L. H. Peoples, and J. A. Collazo. 2001. Partners in Flight conservation plan for the South Atlantic Coastal Plain. U.S. Fish and Wildlife Service.

Hurlbert, A. H., and J. P. Haskell. 2003. The effect of energy and seasonality on avian species richness and community composition. The American Naturalist 161:83-97.

Hutto, R. L., S. M. Pletschet, and P. Hendricks. 1986. A fixed-radius point count method for nonbreeding and breeding-season use. The Auk 103:593-602.

Igl, L. D., and B. M. Ballard. 1999. Habitat associations of migrating and overwintering grassland birds in Southern Texas. The Condor 101:771-782.

Kempton, R. A., and L. R. Taylor. 1976. Models and statistics for species diversity. Nature 262:818-820.

Kindscher, K., and L. L. Tieszen. 1998. Floristic and soil organic matter changes after five and thirty-five years of native tallgrass prairie restoration. Restoration Ecology 6:181-196.

Klute, D. S., R. J. Robel, and K. E. Kemp. 1997. Seed availability in grazed pastures and Conservation Reserve Program fields during winter in Kansas. Journal of Field Ornithology 68:253-258.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247-257.

Kobal, S. N., N. F. Payne, and D. R. Ludwig. 1999. Habitat/area relationships, abundance, and composition of bird communities in 3 grassland types. Transactions of the Illinois State Academy of Science 92:109-131.

Marcus, J. F., W. E. Palmer, and P. T. Bromley. 2000. The effects of farm field borders on overwintering sparrow densities. Wilson Bulletin 112:517-523.

Martin, S. 1969. A technique for capturing nesting grassland birds with mist nets. Bird Banding 40:233-237.

McCraken, D. I., E.M. Bignal, S. Blake, and G. N. Foster. 2004. Productivity and profitability: the effects of farming practices on the prey of insectivorous birds. Pages 75-87 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

Mikami, O. K., and M. Kawata. 2002. The effects of individual interactions and habitat preferences on spatial structure in a grassland bird community. Ecography 25:200-214.

National Resources Conservation Service. 2005. Soil Data Mart. http://soildatamart.nrcs.usda.gov/ Accessed 05/05/2006.

Nur, N., S. L. Jones, and G. R. Geupel. 1999. A statistical guide to data analysis of avian monitoring programs. U.S. Department of the Interior, Fish and Wildlife Service. Report BTP-R6001-1999.

Nuttle, T., A. Leidolf, and L. W. Burger, Jr. 2003. Assessing conservation value of bird communities with Partners in Flight-based ranks. The Auk 120:541-549.

O'Leary, C. H., and D. W. Nyberg. 2000. Treelines between fields reduce the density of grassland birds. Natural Areas Journal 20:243-249.

Patterson, M. 2002. Analysis of bird point count data for the Mill Pond Site, Neawanna Ecological Observatory, Seaside, Clatsop Co., Oregon. Oregon Birds 28:168-171.

Perkins, A. J., M. J. Whittingham, R. B. Bradbury, J. D. Wilson, A. J. Morris, and P. R. Barnett. 2000. Habitat characteristics affecting use of lowland agricultural grassland by birds in winter. Biological Conservation 95:279-294.

Peterjohn, B. G., and J. R. Sauer. 1999. Population status of North American grassland birds from the North American Breeding Bird Survey, 1966-1996. Studies in Avian Biology 19:27-44.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1998. Site fidelity of wintering Henslow's Sparrows. Journal of Field Ornithology 69:486-490.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1999. Habitat requirements of Henslow's Sparrows wintering in silvicultural lands of the Gulf Coastal Plain. The Auk 116:109-115.

Pons, P., B. Lambert, E. Rigolot, and R. Prodon. 2003. The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape. Biodiversity and Conservation 12:1843-1860.

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

Rising, J. D., and D. D. Beadle. 1996. A guide to the indentification and natural history of the sparrows of the United States and Canada. San Diego: Academic Press.

Robinson, R. A., J. D. Hart, J. M. Holland, and D. Parrott. 2004. Habitat use by seed-eating birds: a scale dependent approach. Ibis 146:87-98.

Root, T. 1988. Atlas of wintering North American birds: an analysis of Christmas Bird Count data. Chicago: The University of Chicago Press.

SAS Institute. 2005. SAS OnlineDoc version 9.1.3. Cary, NC: SAS Institute, Inc.

Shochat, E., M. A. Patten, D. W. Morris, D. L. Reinking, D. H. Wolfe, and S. K. Sherrod. 2005. Ecological traps in isodars: effects of tallgrass prairie management on bird nest success. Oikos 111:159-169.

Sibley, D. A. 2000. The Sibley guide to birds. New York: Chanticleer Press, Inc.

Tucker, J. W., Jr., and W. D. Robinson. 2003. Influence of season and frequency of fire on Henslow's sparrows (Ammodramus Henslowii) wintering on Gulf Coast pitcher plant bogs. The Auk 120:96-106.

Vickery, P. D., J. Herkert, R., F. L. Knopf, J. Ruth, and C. E. Keller. 2000. Grassland birds: an overview of threats and recommended management strategies. Pages 74–77 in R. Bonney, D. N. Pashley, R. J. Cooper, and L. Niles, editors. Strategies for bird conservation: the Partners in Flight planning process. Proceedings of the 3rd Partners in Flight workshop (1995). Proceedings RMRS-P-16. U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah.

Vickery, P. D., and J. R. Herkert. 2001. Recent advances in grassland bird research: where do we go from here? The Auk 118:11-15.

Vogl, R. J. 1973. Fire in the Southeastern grasslands. Proceedings of the Tall Timbers Fire Ecology Conference 12:175-198.

Walk, J. W., and R. E. Warner. 2000. Grassland management for the conservation of songbirds in the Midwestern USA. Biological Conservation 94:165-172.

Wiens, J. A. 1969. An approach to the study of ecological relationships among grassland birds. Ornithological Monographs 8:1-93.

Wiens, J.A. 1973. Pattern and process in grassland bird communities. Ecological Monographs 43:237-270.

Wiens, J.A. 1974. Habitat heterogeneity and avian community structure in North American grasslands. American Midland Naturalist 91:195-213.

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

Table 2.1. Modified Braun-Blanquet scale for grass cover determination in grassland habitat.

Cover score	Grass cover
0	0%
1	0-5%
2	5-25%
3	25-50%
4	50-75%
5	75+

Table 2.2. Species richness (no. of species) for forbs, grasses, and shrubs and trees during summer, 2002-2004, in central Georgia for fields managed with annual mowing and periodic burning (control) or planted with native warm season grasses (planted).

		Forbs	SE	Grasses	SE	Shrubs and trees	SE
2002							
	Control	13.5	5.1	7.2*	2.4	4.2	2.1
	Planted	13.5	6.1	3.7	1.0	2.2	1.5
2003							
	Control	17.3	9.6	6.7	2.5	4.2	1.7
	Planted	25.5	7.7	9.8*	2.8	4.2	2.3
2004							
	Control	18.7	6.7	4.3	1.9	6.0	2.1
	Planted	23.0	4.9	5.3	1.5	4.5	2.1

*Indicates significant treatment effect within year (ANOVA, randomized complete block design $p \leq 0.05$)

Table 2.3. Vertical vegetation density (%), grass cover score (1-5), litter depth (cm), grass height (cm), and shrub cover (%) during summer, 2002-2004, in central Georgia for fields managed with annual mowing and periodic burning (control) or planted with native warm season grasses (planted). Missing values indicate data was not collected during that year.

	Grass cover	SE	Native Grass Cover	SE	Litter depth	SE	Grass height	SE	Vertical veg density	SE	Shrub cover	SE
2002												
Control	3.9*	0.8	2.7	2.0	2.0*	1.6			72.5*	21.4		
Planted	1.8	1.0	2.8	1.2	0.0	0.0			34.2	16.0		
2003												
Control	3.5	1.2	2.3	1.3	1.5	1.2	44.7	13.4	69.7	14.2		
Planted	3.2	1.4	4.1*	0.6	1.4	2.2	69.7*	22.4	71.1	18.5		
2004												
Control	2.9	1.0	2.4	1.6	1.8	1.6	20.1	3.7	60.3	13.3	28.7*	19.4
Planted	3.1	0.7	4.6*	0.3	1.5	1.2	39.5*	26.7	81.1*	13.0	9.4	5.7

*Indicates significant treatment effect within a year (ANOVA, randomized block design, $p \leq 0.05$)

Table 2.4. Capture rates (birds/100 net hours) during the breeding season, 2002-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR Georgia, USA. Two most abundant species for each treatment-year combination are in **bold**. Missing values indicate no captures.

Common name	Scientific name	2002		2003		2004	
		Plant	Control	Plant	Control	Plant	Control
American goldfinch[ac]	*Carduelis tristis*			0.63	0.62		
Bachman's sparrow[c]	*Aimophila aestivalis*			0.29	0.15		
Black-and-white warbler	*Mniotilta varia*				0.15		0.17
Blue grosbeak	*Passerina caerulea*	0.53	0.25	0.70	**0.70**	0.47	1.27
Blue-gray gnatcatcher	*Polioptila caerulea*				0.15		
Carolina wren[a]	*Thryothorus ludovicianus*				0.70		0.31
Chipping sparrow[a]	*Spizella passerina*	**2.66**	0.25	0.42		**1.58**	0.33
Common yellowthroat	*Geothlypis trichas*			0.46		1.39	
Eastern bluebird[a]	*Sialia sialias*	1.55					
Eastern kingbird[c]	*Tyrannus tyrannus*		0.29	0.24		0.16	
Eastern meadowlark	*Sturnella magna*		0.57				0.16
Eastern phoebe[a]	*Sayornis phoebe*	0.29		0.14		0.15	
Eastern towhee	*Pipilo erythrophthalmus*						0.17
Field sparrow[a]	*Spizella pusilla*	0.29	**4.11**	**2.18**	0.39	1.36	**3.78**
Hooded warbler[c]	*Wilsonia citrina*			0.24			
Indigo bunting	*Passerina cyanea*	**2.70**	**1.67**	**2.15**	**1.87**	**4.40**	**3.33**
Northern bobwhite	*Colinus virginianus*			0.14			

(continued)

Table 2.4. (continued) Capture rates (birds/100 net hours) during the breeding season, 2002-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR Georgia, USA. Two most abundant species for each treatment-year combination are in **bold**. Missing values indicate no captures.

Common name	Scientific name	2002		2003		2004	
		Plant	Control	Plant	Control	Plant	Control
Northern cardinal[a]	*Cardinalis cardinalis*		0.59	0.50	0.69	0.60	0.64
Orchard oriole	*Icterus spurius*	0.47	0.25			0.16	
Palm warbler[bc]	*Dendroica palmarum*			0.14	0.15	0.46	0.33
Pine warbler	*Dendroica pinus*	0.47	0.24				0.32
Prairie warbler	*Dendroica discolor*		0.49	0.30	0.20	0.31	0.94
Red-eyed vireo	*Vireo olivaceus*				0.39		
Ruby-throated hummingbird[c]	*Archilochus colubris*			0.14	0.35	0.16	0.15
Savannah sparrow[bc]	*sandwichensis*					0.16	
Summer tanager	*Piranga rubra*		1.23				0.32
Swamp sparrow[bc]	*Melospiza georgiana*					0.31	0.16
Tufted titmouse	*Baeolophus bicolor*				0.15	0.64	0.31
Veery[bc]	*Catharus fuscescens*					0.16	
White-eyed vireo	*Vireo griseus*						0.15
Yellow-breasted chat	*Icteria virens*		0.24	0.99	0.20	0.92	0.33
Yellow-rumped warbler[bc]	*Dendroica coronata*						0.33
Yellow-throated warbler	*Dendroica dominica*		0.24				

[a]Species that were captured throughout summer and winter.

[b]Winter resident or migrant captured at the beginning of 2004 breeding season only.

[c]Species that were captured during summer, but not detected on point counts.

Table 2.5. Diversity (expressed as exp H'), richness (no. of species), total conservation value (TCV), and abundance (birds/100 net hours) calculated from breeding season mist net data, 2002-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR in Georgia, USA.

	Diversity	SE	Richness	SE	TCV[a]	SE	Abundance	SE
2002								
Control	2.6	0.9	2.8	1.0	16.7	9.2	10.9	6.2
Planted	2.3	2.0	2.5	2.4	12.2	13.3	9.0	8.4
2003								
Control	3.8	2.2	4.2	2.7	15.0	11.9	7.0	6.0
Planted	4.3	1.2	5.0	1.8	22.7	12.1	9.8	5.7
2004								
Control	4.3	2.6	5.3	3.5	36.7	37.0	13.5	12.9
Planted	4.5	1.3	5.8	2.0	33.2	13.5	13.4	5.7

[a]Calculated according to Nuttle et al 2003.

Table 2.6. Density (birds/ha) calculated from breeding season point count data, 2002-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR in Georgia, USA.

Common name	Scientific name	2002[a]		2003		2004	
		Control	Planted	Control	Planted	Control	Planted
Acadian flycatcher[b]	*Empidonax virescens*	0.42			0.21		0.21
American crow[b]	*Corvus brachyrhynchos*			0.21	0.21		
Black-and-white warbler	*Mniotilta varia*		0.42	0.42	0.21	0.85	
Blue grosbeak	*Passerina caerulea*	0.42	0.42	0.21	0.42		0.21
Blue jay[b]	*Cyanocitta cristata*			0.21		0.42	
Blue-gray gnatcatcher	*Polioptila caerulea*	0.42		1.70	1.70	1.49	1.91
Brown-headed cowbird[b]	*Molothrus ater*	0.85			0.42		
Brown-headed nuthatch[b]	*Sitta pusilla*	0.42	0.42				
Carolina chickadee[b]	*Poecile carolinensis*	0.42	0.42	0.85		0.21	0.42
Carolina wren	*Thryothorus ludovicianus*	0.42	0.42	0.64	0.64		0.85
Chipping sparrow	*Spizella passerina*	1.70	1.70	2.55	1.91	1.06	2.55
Common yellowthroat	*Geothlypis trichas*			0.21	0.64	0.21	0.64
Downy woodpecker[b]	*Picoides pubescens*	0.42					
Eastern bluebird	*Sialia sialias*		0.42				
Eastern meadowlark	*Sturnella magna*			0.64		0.64	
Eastern phoebe	*Sayornis phoebe*		0.85		0.21		
Eastern towhee	*Pipilo erthrophthalmus*		0.42	0.21		0.42	0.85

(continued)

Table 2.6. (continued) Density (birds/ha) calculated from breeding season point count data, 2002-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR in Georgia, USA.

Common name	Scientific name	2002[a]		2003		2004	
		Control	Planted	Control	Planted	Control	Planted
Eastern wood-pewee[b]	*Contopus virens*						0.21
Field sparrow	*Spizella pusilla*	1.27	0.42	1.06	0.64	0.64	0.21
Grasshopper sparrow[b]	*Ammodramus savannarum*						0.21
Great-crested flycatcher[b]	*Myiarchus crinitus*	0.85		0.42	0.64	0.21	
Indigo bunting	*Passerina cyanea*	0.42	2.55	2.55	4.24	1.91	1.27
Mourning dove[b]	*Zenaida macroura*		1.27	0.21	0.21		
Northern bobwhite	*Colinus virginianus*				0.21		
Northern cardinal	*Cardinalis cardinalis*			1.06	1.91	0.64	1.91
Northern flicker[b]	*Colaptes auratus*		0.42				
Northern parula[b]	*Parula americana*	1.27	2.12		0.42		
Orchard oriole	*Icterus spurius*					0.21	
Pileated woodpecker	*Dryocopus pileatus*			0.21			
Pine warbler	*Dendroica pinus*			0.64			0.64
Prairie warbler	*Dendroica discolor*	0.42		0.42	0.85		
Red-bellied woodpecker[b]	*Melanerpes carolinus*	0.42		0.64	0.21	0.64	0.42
Red-eyed vireo	*Vireo olivaceus*		0.85	0.42	0.85	0.64	0.21

(continued)

Table 2.6. (continued) Density (birds/ha) calculated from breeding season point count data, 2002-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR in Georgia, USA.

Common name	Scientific name	2002[a]		2003		2004	
		Control	Planted	Control	Planted	Control	Planted
Red-headed woodpecker[b]	*Melanerpes erthrocephalus*	1.27					
Red-shouldered hawk[b]	*Buteo lineatus*		0.42				
Summer tanager	*Piranga rubra*	0.85			0.42	0.42	0.85
Tufted titmouse	*Baeolophus bicolor*		0.85	0.21		1.27	1.49
White-eyed vireo	*Vireo griseus*				0.64		
Yellow warbler[b]	*Dendroica petechia*	0.85					
Yellow-billed cuckoo[b]	*Coccyzus americanus*				0.64		
Yellow-breasted chat	*Icteria virens*				0.64	0.21	0.42
Yellow-throated vireo[b]	*Vireo flavifrons*	0.85		1.49	1.06		0.42
Yellow-throated warbler	*Dendroica dominica*					0.64	0.21

[a]Only includes data from Piedmont NWR.

[b]Species detected on point counts, but not captured during summer mist netting.

Table 2.7. Diversity (expressed as exp H'), richness (no. of species), total conservation value (TCV), and abundance (birds/ha) calculated from breeding season point count data, 2002-2004 for fields managed with annual mowing and periodic burning (control) or planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR in Georgia, USA.

		Diversity	SE	Richness	SE	TCV[a]	SE	Abundance	SE
2002									
	Control	8.1	4.4	8.7	4.9	26.7	19.6	14.4	10.2
	Planted	6.9	1.7	7.7	1.5	26.0	7.0	14.4	4.1
2003									
	Control	6.2	2.6	7.7	3.5	32.3	16.1	17.2	7.5
	Planted	7.5	3.1	9.0	3.0	36.8	13.8	20.2	6.7
2004									
	Control	4.9	3.3	5.5	3.5	23.3	11.4	12.9	6.2
	Planted	6.8	2.2	7.7	2.7	29.0	8.6	16.6	4.9

[a]Calculated according to Nuttle et al 2003.

Table 2.8-Capture rates (birds/100 net hours) during January-February, 2003-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR in Georgia, USA. Two most abundant species for each treatment-year combination are in bold. Missing values indicate no captures.

Common name	Scientific name	2003		2004	
		Control	Plant	Control	Plant
American goldfinch	*Carduelis tristis*	0.29	2.00		
American kestrel[a]	*Falco sparverius*		0.27		
American robin[a]	*Turdus migratorius*	1.20		0.54	0.85
Blue-headed vireo[a]	*Vireo solitarius*	0.26			0.17
Brown thrasher[a]	*Toxostoma rufum*				0.14
Carolina chickadee[a]	*Poecile carolinensis*	0.24	0.49		0.14
Carolina wren	*Thryothorus ludovicianus*			0.18	
Chipping sparrow	*Spizella passerina*	0.96	9.72	0.18	
Eastern bluebird	*Sialia sialias*	0.49	0.76	0.53	
Eastern phoebe	*Sayornis phoebe*	0.79		0.18	0.14
Field sparrow	*Spizella pusilla*	1.48	6.84	0.18	0.88
Fox sparrow[a]	*Passerella iliaca*			0.18	
Golden-crowned kinglet[a]	*Regulus satrapa*	0.25			
Hermit thrush[a]	*Catharus guttatus*	0.29			0.52
Northern cardinal	*Cardinalis cardinalis*	0.24	0.25		0.17
Northern mockingbird	*Mimus polyglottos*	0.48			
Palm warbler	*Dendroica palmarum*	0.25			
Pine warbler	*Dendroica pinus*	2.18	1.73	0.35	0.56
Red-bellied woodpecker[a]	*Melanerpes carolinus*	0.25			

(continued)

Table 2.8. (continued) Capture rates (birds/100 net hours) during January-February, 2003-2004, for fields managed with annual mowing and periodic burning (control) and planted with native warm season grasses (planted) at Joe Kurz WMA and Piedmont NWR in Georgia, USA. Two most abundant species for each treatment-year combination are in bold. Missing values indicate no captures.

Common name	Scientific name	2003		2004	
		Control	Plant	Control	Plant
Ruby-crowned kinglet[a]	*Regulus calendula*		0.27		0.84
Savannah sparrow	*Passerculus sandwichensis*	0.24	2.55	0.90	0.57
Song sparrow[a]	*Melospiza melodia*	1.30	8.74	0.18	1.71
Swamp sparrow	*Melospiza georgiana*	1.00			0.70
Tufted titmouse	*Baeolophus bicolor*			0.35	0.28
Vesper sparrow[a]	*Pooecetes gramineus*		0.50		0.14
White-throated sparrow[a]	*Zonotrichia albicollis*		0.25		
Yellow-rumped warbler	*Dendroica coronata*	0.48			

[a]Species that were only captured during the winter.

Table 2.9. Diversity (expressed as exp H'), richness (no. of species), total conservation value (TCV), abundance (birds/100 net hours), and sparrow abundance (sparrows/100 net hours) calculated from mist net data in central Georgia, January-March 2003-2004 for fields managed with annual mowing and periodic burning (control) or planted with native warm season grasses (planted).

		Diversity	SE	Richness	SE	TCV[a]	SE	Abundance	SE	Sparrow abundance	SE
2003											
	Control	3.7	2.6	4.2	3.5	20.3	25.6	12.7	15.4	5.0	6.8
	Planted	3.3	1.8	4.7	2.3	50.5*	28.3	34.6*	16.8	28.6*	14.8
2004											
	Control	2.2	0.9	2.2	1.2	6.8	4.0	3.9	2.3	1.6	2.4
	Planted	3.2	2.0	3.8	2.5	18.2*	12.2	7.8	4.8	4.0	4.4

[a]Calculated according to Nuttle et al 2003.
*Indicates significant treatment effect at $p \leq 0.05$.

CHAPTER 3

IMPACT OF MANAGEMENT FOR EARLY SUCCESSIONAL HABITAT ON ARTHROPOD COMMUNITIES[2]

[2]McMellen, A.B., D.E.W. Drumtra, and S. H. Schweitzer. To be submitted to *Conservation Biology*.

ABSTRACT

Grassland birds are experiencing rapid population declines in the eastern United States. Enhancing invertebrate abundance and diversity should be a priority within management plans because arthropods are a key component of avian diets during the breeding season. Within 21 plots, from 1.2–4.0 ha each, at 4 locations in central Georgia, we investigated arthropod abundance and richness in 7 current management strategies for creating and maintaining early successional habitat in the southeastern United States. Management strategies ranged from prescribed burning only to re-establishment of native grasses in fallow fields. Arthropods were sampled using 0.6-m sweep nets during summer 2003 and 2004. Fire ants (*Solenopsis invicta*) were sampled in 2004 using bait traps baited with hot dog. Local and landscape habitat variables were measured each summer. One-way analysis of variance and canonical correspondence analysis (CCA) were used to analyze data. In 2003, arthropod richness was highest in native grass plots and lowest in plots with the greatest percentage of canopy cover. In 2004, richness was higher in open native grass plots than in open exotic grass plots and in plots with a high percentage of canopy cover. Abundance was greatest in open native grass plots in 2003 and 2004. Araneae were most common in native grass plots in 2003, but most common in open woodland in 2004. In both years, orthopterans were most abundant in native grass plots. Fire ants did not impact arthropod abundance or richness. The CCA revealed that local (forb and grass cover) and landscape (canopy cover, forest cover) variables affected the structure of the arthropod community. Plots with a high percentage of canopy cover had low

abundance and diversity of arthropods. Managing plots to increase forb and grass cover and decrease canopy cover, likely will increase prey availability for breeding birds.

INTRODUCTION

Grassland birds are undergoing widespread population declines that are steeper and more consistent than any other group of birds (Knopf 1994). Forty percent of North American birds dependent on a disturbance ecosystem experienced a significant decline between 1969 and 1998 (Brawn et al. 2001). Fire suppression, declining beaver populations, urbanization, aforestation, short rotation pine plantations, intensive agriculture, and introduction of exotic pasture grasses have all been implicated in the decline of eastern grassland birds (Askins 2000). Breeding Bird Survey data from the eastern U.S. revealed significant population declines for 13 of 14 grassland birds while only 2 of 40 forest birds had declining trends (Askins 2000).

Southeastern grasslands have undergone centuries of modification including exclusion of fire, intensive grazing, and the introduction of cultivated, exotic, sod-forming grasses. These modifications have resulted in an increase in hardwood trees and shrubs, changes in herbaceous species composition, loss of native grasses, and the near extirpation of native warm-season grass species (Rasnake 1992). Most southeastern pastures are now planted in introduced cool and warm-season grass species such as fescue (*Festuca arundinacea*), bermuda (*Cynodon dactylon*), and bahia grass (*Paspalum notatum*). Exotic grass pastures may be good for widespread avian species, but grassland specalists are rarely found within these pastures (DeVault et al. 2002).

Invertebrate abundance and diversity should be considered a top priority in avian management (Robel et al. 1995). Arthropods are a key component of avian diets during the spring and summer. Both chicks and older birds rely on this protein rich food source to fuel energetic demands of rapid growth and reproduction, respectively. If food resources are limited, adult birds may face a trade-off between survival and reproductive success (Martin 1995). Availability of arthropods to provision nestlings may directly limit nestling survival (Moreby 2004).

The link between birds and arthropod abundance in early successional habitat has been the focus of much research in Europe. A review of 27 years of insect and bird data in Scotland found a link between the decline of insect species and bird species (Benton et al. 2002). The declines in birds and arthropods were the result of changing land management practices. Over the last 40 years, arthropod abundance and the birds that rely on arthropods during the breeding season have declined in Britain with modification and intensification of land management practices (reviewed in Barker 2004). Barker (2004) suggests that the declines of arthropods are an important cause of the decline of farmland birds in the British Isles.

Fire is the most common management technique for creating and maintaining early successional habitat in the southeastern United States. The abundance (Anderson et al. 1989) and biomass (Shochat et al. 2005b) of arthropods are greater immediately after fire. Arthropod abundance in Oklahoma was higher on plots that were burned or grazed than on plots that received no management (Shochat et al. 2005a). Intensively managed grasslands had more widespread, less specialist arthropod species (DiGiulio et al. 2001). Ant communities quickly recover after prescribed fire

(Izhaki et al. 2003), but fire can reduce ant density and change community composition (Castano-Meneses and Palacios-Vargas 2003).

Early successional habitat in the Southeast exists in a spectrum from open, agricultural land to grassy understories of open-canopy woodland. Many eastern birds may be defined as disturbance-dependent rather than early successional or grassland birds because they are not restricted to a single habitat type, but exist across a spectrum ranging from grasslands to gaps in mature forests (Hunter et al. 2001). Management for early successional bird species in the eastern United States requires an understanding of the impacts of various land management alternatives on avian food resources. Arthropod communities of grassland ecosystems in the United States are poorly understood (Whiles and Charlton 2006). Different management strategies could have varying impacts on arthropod abundance, influencing the availability, abundance, and size of arthropods (McCraken et al. 2004).

The purpose of this study was to examine arthropod community responses to management practices that create and maintain early successional habitat in the southeastern United States. We hypothesized that arthropod richness and abundance would be greatest in treatments with the highest native grass and forb cover. We also hypothesized that fire ant (*Solenopsis invicta*) abundance would negatively impact the richness and abundance of other arthropods.

STUDY AREAS

We conducted field work at Piedmont National Wildlife Refuge (NWR), Joe Kurz Wildlife Management Area (WMA), Oconee National Forest (NF), and Clark's Hill WMA in central Georgia. All locations were in the transition zone between the Piedmont and coastal plain physiographic areas and within the historic range of pineland savannah habitat (Vogl 1973). Piedmont NWR and Oconee NF were located in Jones and Jasper Counties (Figure 3.1). Joe Kurz WMA was in Meriwether County and Clark's Hill WMA was in McDuffie County.

Piedmont NWR was approximately 14,000 ha of upland pine (*Pinus* spp.) habitat managed by the U.S. Fish and Wildlife Service. The landscape was >95% forested, consisting of loblolly pine (*Pinus taeda*) canopy with an understory of flowering dogwood (*Cornus florida*), redbud (*Cercis canadensis*), and various native and non-native grasses. The bottomlands were mixed hardwoods dominated by oaks (*Quercus* spp.). The endangered red-cockaded woodpecker (*Picoides borealis*) was the primary management focus. Large areas of the refuge were burned on a 3-5 year rotation to create and maintain open pine parklands, preferred foraging habitat of this woodpecker. Six forest openings at Piedmont NWR were used as study plots. The forest openings originally were established as food plots for white-tailed deer (*Odocoileus virginianus*), eastern wild turkey (*Meleagris gallopavo*), and other wildlife, but had not been used in that capacity since the 1970s when refuge personnel began to manage them as early successional habitat using annual mowing and periodic burning. Hence, the forest openings were old field habitat consisting of grasses, forbs, shrubs, and scattered saplings. Dominant grasses were bahia, broomsedge (*Andropogon virginicus*), Johnson grass (*Sorghum halpense*), and fescue. Blackberry (*Rubus* spp.) was the most common woody plant. Most saplings were loblolly pine and sweetgum (*Liquidambar styraciflua*).

Joe Kurz WMA was a privately held farm until it was acquired by the Georgia Department of Natural Resources (GA DNR) in 1995. The surrounding landscape was 70-75% forested with row crop agricultural, pasture, and hay fields accounting for most of the remaining land cover. Joe Kurz WMA was approximately 1,495 ha of old field habitat, upland loblolly pine, and bottomland oaks. The bottomland hardwood forest was more extensive at Joe Kurz WMA than Piedmont NWR. The vegetative structure of our 6 study plots was old field habitat similar to plots at Piedmont NWR.

Clark's Hill WMA was approximately 5,139 ha of land leased by GA DNR from the Army Corps of Engineers. Most of the habitat was upland loblolly pine. Several sections of pine were thinned in the early 1980s and had been maintained at a low basal area (9-11 m^2/ha) with a floristically diverse understory of native grasses (*Panicum anceps*, *Sorghastrum nutans*, and *Andropogon gyrans*) and forbs by using frequent (3-year rotation), low-intensity winter fire.

Oconee NF was 46,393 ha of forested land in central Georgia under multi-use management. Over 70% of the forest was pine, dominated by loblolly and shortleaf (*Pinus echinata*) pines. Established in 1959, most of Oconee NF consisted of pines between 60-80 years old. Because it was considered a potential key site for red-cockaded woodpecker conservation and restoration, the forest management plan aimed to create or restore 400 ha of woodlands, savannahs, and grasslands, and thin an additional 500 ha of loblolly and shortleaf pine (USDA Forest Service 2004).

Seven current management strategies for creating and maintaining early successional or disturbance-dependent habitat were investigated across the 4 locations. Each management strategy was replicated in 3 plots for a total of 21 study plots (Table

3.1). Each plot was between 1.2-4 ha in size and separated by a minimum of 1 km. All plots were burned during winter 2001-2002. Open planted (OP) and forest opening planted (FP) plots were planted with a mixture of switch grass (*Panicum virgatum*), big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), and Indian grass (*Sorghastrum nutans*) during spring 2002 (see Chapter 2 for planting details).

METHODS

Arthropod sampling

Arthropods were sampled using sweep nets (Schotzko and O'Keeffe 1989; 1986). Plots were sampled once in June 2003, and once in April, May, and June 2004. Sample periods were chosen to correspond with peak avian nesting and demand for arthropod food resources in central Georgia.

Three 10-m transects were randomly located in each plot during each sampling period. Each transect was sampled with 12 sweeps of a 0.6-m net. Contents of the sweep net were transferred to a plastic bag, placed on ice in the field, and frozen upon return to the laboratory. Arthropods were removed from plant material in the lab and sorted into orders, then families, using standard guides and keys (Chu and Cutkomp 1992, Borror et al. 1989, Bland and Jaques 1978, Borror and White 1970). Acari, Araneae, Diplopoda, Opilionidae, and Psocoptera were only identified to order and therefore were not included in calculations of family richness.

Total arthropod family richness and abundance were calculated for all plots using data collected from sweep netting. Richness and abundance for orthopterans, lepidopterans, and Araneae were also calculated as these are primary food items selected by birds during the breeding season (McIntyre and Thompson 2003).

Richness was defined as the number of families detected in the plot during all sampling within a season. Arthropod abundance was defined as the average number of individuals captured during a sample period (*n* = 3 transects).

Ant sampling

To examine the relationship between fire ants and arthropod communities, we sampled the ant community using bait traps (Wojcik 1994) in 2004. Once during May and June 2004, bait sample transects were laid out in each plot. Ten bait stations, each separated by 10 m, were established in each plot along the long axis. A piece of hot dog (~10 g and 2-cm^3) was placed on a 5-cm^2 piece of aluminum foil at each bait station for 90 min then collected. In the lab, ants were preserved in alcohol, identified to species using identification keys (Bolton 1994, Mackay and Mackay 2003), and counted. Species identification was verified by C. R. Carroll (Institute of Ecology, University of Georgia, Athens) using specimens from the Museum of Natural History, University of Georgia, Athens. Voucher specimens of all identified ant species were catalogued at the D. B. Warnell School of Forestry and Natural Resources, University of Georgia, Athens.

Habitat assessment

Landscape and local habitat variables were measured in all plots. Percent forest cover in the landscape was determined using Georgia GAP data based on a 1998 landcover at a resolution of 30 m (Natural Resource Spatial Analysis Laboratory 1998). Buffers of 500 m and 1,000 m were created around the GPS centroid of each plot. This study was part of a larger project examining bird community response to land management. Buffer sizes were chosen based on common buffer sizes used in the avian landscape ecology literature (Saab 1999, Mayer and Cameron 2003). We determined the area within each buffer classified as forest. All types of forest (deciduous, evergreen, mixed, and wetland) were included in our definition of forest cover.

Local habitat variables were measured in July 2003-2004. Twenty variables were measured in 1-m^2 square sample subplots. Sample subplots were distributed through plots in a grid-like fashion. All subplots were ≥10 m from the plot edge and ≥40 m from other sample subplots. Cover (%) of exotic grass, native grass, shrubs, and bare ground was visually estimated using a modified Braun-Blanquet cover scale (Braun-Blanquet 1932, Table 3.2). We identified all species of forbs, grasses, and woody plants within each subplot. Litter depth (cm) and grass height (cm) were measured in the center of each subplot. Hardwood and pine basal areas were determined from the center of each sample point using an optical prism. A 2.0-m cover board (Nudds 1977) was used to estimate vertical vegetation density (%) in the first 2 0.5-m strata (0-0.5 m and 0.5-1.0 m) by an observer 10 m from the center of the sample subplot. Vertical vegetation density was estimated in the 4 cardinal directions and averaged to determine a value for each subplot. Canopy cover was categorized as present (1) or absent (0) for each subplot using a densitometer at the center of the sample plot. These values were converted to percent canopy cover for each plot.

Statistical analysis

One-way analysis of variance (ANOVA) was used to compare richness and abundance of total arthropods, orthopterans, and lepidopterans, and abundance of araneae among treatments by year. Due to temporal differences in sampling period between years, we did not pool data over years. Duncan's multiple range test was used for *post hoc* comparisons of significant ANOVA results ($\alpha = 0.05$). Pearson's correlation coefficients were calculated to determine the relationship between fire ant abundance and arthropod community metrics. Correlation coefficients were calculated for May and June 2004 using the ant and arthropod data collected during those 2 sampling periods. The ANOVA and Pearson's correlations were calculated using PROC GLM and PROC CORR of SAS, respectively (SAS Institute 2005).

Arthropod community structure was analyzed using canonical correspondence analysis (CCA) on abundance data (ter Braak 1986). A series of CCAs was run with landscape and local habitat variables to assess colinearity between variables. A variable with a variance inflation factor (VIF) >20 indicated that it was perfectly correlated with another environmental variable in the dataset and did not contribute any unique information to the ordination (ter Braak 1986); therefore, variables with VIFs >20 were eliminated one at a time until all VIFs were <20. Variables with the largest VIFs were eliminated first. Elimination of all correlated variables left 12 variables that were considered in the CCAs. CCAs were run for each sampling period. The significance of the relationship between the landscape and local habitat variables and the species data was tested using Monte-Carlo global permutation tests (1,000 random permutations) provided in CANOCO (ter Braak and Šmilauer 2002). Forward selection tests were performed to determine the environmental variables that best explained the species data. All CCAs were performed using CANOCO for Windows 4.5 (ter Braak and Šmilauer 2002).

RESULTS

Arthropod abundance and richness

In 2003, 4,839 arthropods from 63 taxa were captured using sweep nets during the single sampling period in June. During 3 sampling periods in 2004, 9,180 individuals from 84 taxa were captured. The lowest number of captures was in April (1,346 individuals, 61 taxa) and the highest number was in June (4,255 individuals, 63 taxa) 2004. Arthropod richness differed among land management treatments in 2003 ($F = 4.52$, df = 6,14, $P = 0.0094$) and 2004 ($F = 3.03$, df = 6,56, $P = 0.01$, Table 3.3). In 2003, arthropod richness was highest in the 2 treatments that included native grass plantings (OP and FP) and lowest in the 2 treatments with the greatest amount of canopy cover (TF and UTF). In 2004, arthropod richness was greater in the open planted treatment than the open control (OUP) and the 2 treatments with the greatest amount of canopy cover (TF and UTF). The unplanted forest opening (FUP) had more species of arthropods than the unthinned forest (UTF), but arthropod richness in FUP was similar to all other treatments. The richness of orthopteran or lepidopteran families was not different among management treatments.

Total arthropod abundance differed among management treatments in 2003 ($F = 11.33$, df = 6,14, $P = 0.0001$) and 2004 ($F = 5.95$, df = 6,56, $P < 0.001$, Figure 3.2a). The open planted treatment (OP) had greater abundance of arthropods than all but the planted forest openings (FP) in 2004 and was greater than all treatments in 2003. Lepidopteran abundance in 2003 and 2004 did not differ among treatments (Figure 3.2b). Araneae abundance differed among treatments in 2003 ($F = 6.99$ df = 6,14, $P = 0.001$) and 2004 ($F = 7.12$ df = 6,56, $P < 0.001$, Figure 3.2c). In 2003, both of the planted treatments (OP and FP) had more spiders than the 2 densest forest treatments (TF and UTF). In 2004, the LBAF treatment had the most spiders and the TF and UTF treatments had the least amount of spiders. Orthopteran abundance differed among treatments in 2003 (F = 4.51, df = 6,14, *P* = 0.01) and 2004 (F = 4.36, df = 6,56, *P* = 0.001, Figure 3.2d). In both years, treatments planted with native grasses had the highest abundance of grasshoppers and the 3 forested treatments (LBAF, TF, and UTF) had the lowest abundance of grasshoppers.

Fire ant impact

During May and June 2004, we captured 197,583 ants of 12 species from 420 bait stations. Most captured ants were fire ants (90% of total ant captures). In only 6 of the 420 samples were fire ants found on bait with another ant species. Five of the 6 samples were in plots with high canopy cover.

Ant abundance (excluding fire ants) and richness were negatively correlated with fire ant abundance in May and June (Table 3.4). Fire ant abundance was positively correlated with arthropod abundance during both sampling periods. Arthropod richness was positively correlated with fire ant abundance in May, but not in June. Araneae abundance was positively correlated with fire ant abundance in May, but not in June. Orthopteran abundance was positively correlated with fire ant abundance in May and June.

Canonical correspondence analysis

Forward selection of environmental variables for the 2003 arthropod data indicated that percentage of grass cover, forb cover, and bare ground; and vegetation density (0.5-1.0 m) were the most important explanatory variables, but only grass cover significantly affected the ordination (*P* = 0.03, Table 3.5). Eigenvalues of axes 1 and 2 were 0.399 and 0.253, respectively. The first 2 axes accounted for 38.5% of the variation in the arthropod data and 54.8% of the variance in the arthropod-environment relationship. According to Monte-Carlo permutation tests, the contribution of all canonical axes was significant (*P* = 0.02), indicating a significant relationship between environmental variables and species data. Graphically, we detected overlap between the arthropod communities in the different treatments, but the 2 treatments with most forest component (TF and UTF) were clustered and associated with low grass cover and vegetation density (Figure 3.3).

During each sampling period in 2004, the first 2 axes accounted for about 25-30% of the variance in the arthropod data and about 43-46% of the variance in the arthropod-environment relationship (Figure 3.4 a-c). Forward selection of environmental variables during each sampling period selected different environmental variables as being most important. Grass cover, forb cover, and canopy cover contributed significantly to the ordination ($P < 0.05$) during April. Native grass cover, forb cover, canopy cover, forest cover, and litter depth were significant ($P < 0.05$) during the May sampling period. Canopy cover and forest cover were significant in June. Average number of fire ants was selected during forward selection of the June data, but it did not impact the ordination. The contribution of all canonical axes was significant in April (F = 2.37, *P* = 0.04), May (F = 2.37, *P* = 0.001), and June (F = 1.84, *P* = 0.001). Graphically, the April 2004 ordination showed overlap in communities among all treatments except TF and UTF, which clustered together (Figure 3.4a). These 2 treatments were characterized by a high percentage of canopy cover. In May, the 3 treatments with a forest component (LBAF, TF, and UTF) clustered together and were associated with a high percentage of canopy cover and greater litter depth (Figure 3.4b). In June, the separation between the 3 treatments with a tree component (LBAF, TF, UTF) and the 4 treatments without trees (OP, OUP, FP, and FUP) was most distinct. A high percentage of canopy cover was associated with the 3 tree treatments while fire ants and native grass cover were associated with open treatments (Figure 3.4c).

DISCUSSION

Our sampling of arthropods with sweep nets undoubtedly missed families of invertebrates both due to sampling method and timing of sampling. Sweep nets are biased against ground dwelling insects and miss taxa captured in pitfall traps (Standen 2000). Arthropod species have variable emergence (Borror et al. 1989) making it difficult to sample the entire arthropod community within a short time period. However, because the same sampling technique was used in all of our plots, any bias inherent in our technique was consistent across treatments. Further, our interest in sampling arthropods was to look at food resource availability for breeding birds. Therefore, the resources that are available during the peak breeding time (April-June) were of most interest to us. In 2003, we only sampled arthropods in June, but in 2004 and another study (Jonas et al. 2002), June had the highest abundance, richness, and diversity of the summer months, so our June 2003 sample likely represented the arthropod community well.

Bait traps only captured a small portion of the 144 taxa of ground-dwelling ants known in Georgia (Ipser et al. 2004). A hot dog is a common bait used for fire ants

(Porter and Tschinkel 1987), but it might miss other ant species that have specialized foraging requirements. In a comparison of ant baits in Georgia, each of 4 bait types attracted different species of ants, but all baits attracted fire ants (Brinkman et al. 2001). Because the purpose of our bait traps was to examine the impact of fire ant abundance on the arthropod community, any bias of our bait against other ant species was inconsequential.

Our results supported our hypothesis that arthropod richness and abundance would be greatest in treatments with high grass and forb cover. Plots with less canopy cover and high forb and grass cover had high arthropod abundance (Figure 3.2a). Forb cover was an important determinant of arthropod community structure in 3 of our sampling periods. A relationship between forb cover and arthropods was not unexpected. Diversity and richness of arthropods in Kansas was positively correlated to plant diversity and richness (Jonas et al. 2002). Abundance and diversity of invertebrates on British farmland declined on farms with management practices that decreased the number and diversity of forbs (Atkinson et al. 2004).

Canopy cover was also an influential habitat variable in all 3 sampling periods in 2004. Treatments with high canopy cover (LBAF, TF, UTF) had low abundance and diversity of arthropods. Canonical correspondence analysis characterized these same treatments as having low forb and grass cover. High canopy cover probably reduced light penetration and development of an understory rich with forbs and grasses. The current forest management plan for Oconee NF calls for thinning upland pine and reducing the hardwood understory (USDA Forest Service 2004). Our data suggest that implementation of the management plan could increase the abundance of understory arthropods.

Fire ants are a serious problem in the southeastern United States but their impacts on invertebrate and vertebrate communities are still uncertain (Vogt et al. 2002, Wojcik et al. 2001, Jusino-Atresino and Phillips 1994, Wojcik 1994, Porter and Savignano 1990). Fire ants negatively impact invertebrates through direct mortality and competition for food and habitat (Wojcik et al. 2001). In our study, fire ant abundance was positively correlated with arthropod abundance (Table 3.4). Conversely, in central Texas, areas infested with fire ants had 40% fewer arthropod species and 75% fewer individuals than areas without fire ants (Porter and Savignano 1990).

The positive correlation between arthropod abundance and fire ants detected in our study may have more to do with landscape level factors than an actual positive impact of fire ants on arthropods. Several studies have found a negative impact of shaded conditions on fire ant abundance. In Florida, fire ants were less abundant in shaded conditions (Porter and Savignano 1990). Fire ant density in South Carolina was positively related to open canopy and disturbance suggesting that direct sunlight may be a requirement for fire ant colonization (Stiles and Jones 1998). In our study, plots with low fire ant and low arthropod abundance were shaded. Low arthropod abundance was likely a result of shading and not the presence of fire ants. Shading reduced the cover of forbs and grass in the understory, thereby reducing the numbers of arthropods sampled by sweep netting.

Orthopteran abundance was reduced in both of our treatments that were maintained with annual mowing (OUP and FUP). Orthopterans are negatively impacted by mechanical disturbance (Jonas et al. 2002). Lepidopteran abundance and richness did not differ among treatments likely because sweep netting is not the best method for sampling larval or adult lepidopterans. Further work using other arthropod sampling techniques would need to be completed to determine the response of lepidopterons to management treatments such as those in our study. Lepidopterans are important food resource for breeding birds (McIntyre and Thompson 2003) and are thus an important component of the environment. Spiders were abundant in all but the most dense forest treatment (UTF) in 2003 and were most abundant in the open woodland treatment (LBAF) in 2004. The presence of trees in early successional habitat did not seem to negatively impact spider abundance.

Arthropod communities can reflect differences in landscape level factors as well as plant communities (Jonas et al. 2002). Our CCA data suggested that both local (forb cover, grass cover, litter depth) and landscape level features (canopy cover, forest cover) were influencing the structure of arthropod communities in our study. Arthropod abundance was highest in our treatments planted with native grasses and lowest in treatments with the highest landscape level forest cover. A recent study of Texas rangelands found higher arthropod abundance in areas with native grasses (Flanders et al. 2006). Native prairie had the highest arthropod abundance when compared to native and non-native conservation reserve program (CRP) fields (McIntyre and Thompson 2003). Plant species composition may be more important than structure in shaping arthropod communities in early successional habitat. Arthropod abundance in Ohio was not influenced by the proximity of a mature forest edge (Rodewald and Vitz 2005).

Breeding bird density has been linked to arthropod abundance in early succesional ecosystems (Zimmerman 1992, Howe et al. 2000, Shochat et al. 2005a). Although arthropods were more abundant in the more open treatments with substantial grass canopy, simple abundance measurements of arthropods may not reflect their availability as food resources to birds (Cooper and Whitmore 1990). Vegetation structure modifies avian foraging behavior through changes in prey detectability, accessibility, and bird mobility (Butler and Gillings 2004).

Large observed differences in nutrient content between different arthropod taxa and grasslands (Robel et al. 1995) indicate that further detailed investigation into arthropod communities in central Georgia may be warranted. Arthropod abundance and richness is a starting point for further studies on the impacts of early successional management on avian food resources.

LITERATURE CITED

Anderson, R.C., T. Leahy, and S.S. Dhillion. 1989. Numbers and biomass of selected insect groups on burned and unburned sand prairie. American Midland Naturalist 122:151-162.

Askins, R.A. 2000. Restoring North America’s birds: lessons from landscape ecology. Second edition. New Haven: Yale University Press.

Atkinson, P.W., D. Buckingham, and A.J. Morris. 2004. What factors determine where invertebrate-feeding birds forage in dry agricultural grasslands? Ibis 146:99-107.

Barker, A.M. 2004. Insects as food for farmland birds: is there a problem? Pages 37-50 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

Benton, T.G., D.M. Bryant, L. Cole, and H.Q.P. Crick. 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. Journal of Applied Ecology 39:673-687.

Bland, R.G. and H.E. Jaques. 1978. How to know the insects. Third edition. Boston: McGraw-Hill.

Bolton, B. 1994. Identification guide to the ant genera of the world. Cambridge, MA: Harvard University Press.

Borror, D.J., C.A. Triplehorn, and N.F. Johnson. 1989. An introduction to the study of insects. Sixth edition. Fort Worth: Harcourt Brace College Publishers.

Borror, D.J. and R.E. White. 1970. A field guide to the insects: America north of Mexico. Boston: Houghton Mifflin Company.

Braun-Blanquet, J. 1932. Plant sociology: the study of plant communities. New York: McGraw-Hill Book Company, Inc.

Brawn, J.D., S.K. Robinson, and F.R. Thompson, III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32:251-276.

Brinkman, M.A., W.A. Gardern, R.M. Ipser, and S.K. Diffie. 2001. Ground-dwelling ant species attracted to four food baits in Georgia. Journal of Entomological Science 36:461-463.

Butler, S.J. and S. Gillings. 2004. Quantifying the effects of habitat structure on prey detectability and accessibility to farmland birds. Ibis 146:123-130.

Castano-Meneses, G. and J.G. Palacios-Vargas. 2003. Effects of fire and agricultural practices on neotropical ant communities. Biodiversity and Conservation 12:1913-1919.

Chu, H.F. and L.K. Cutkomp. 1992. How to know the immature insects. Second edition. Boston: McGraw-Hill Publishers.

Cooper, R.J. and R.C. Whitmore. 1990. Arthropod sampling methods in ornithology. Studies in Avian Biology 13:29-37.

DeVault, T.L., P.E. Scott, R.A. Bajema, and S.L. Lima. 2002. Breeding bird communities in reclaimed coal-mine grasslands in the American Midwest. Journal of Field Ornithology 73:268-275.

DiGiulio, M., P.J. Edwards, and E. Meister. 2001. Enhancing insect diversity in agricultural grasslands: the roles of management and landscape structure. Journal of Applied Ecology 38:310-319.

Flanders, A.A., W.P. Kuvlesky, D.C. Ruthven, R.E. Zaiglin, R.L. Bingham, T.E. Fulbright, F. Hernandez, and L.A. Brennan. 2006. Effects of invasive exotic grasses on South Texas rangeland birds. Auk 123:171-182.

Howe, F.P., R.L. Knight, L.C. McEwen, and T.L. George. 2000. Diet switching and food delivery by shrubsteppe passerines in response to an experimental reduction in food. Western North American Naturalist 60:139-154.

Hunter, W.C., D.A. Buehler, R.A. Canterbury, J.L. Confer, and P.B. Hamel, 2001. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin 29:440-455.

Ipser, R.M., M.A. Brinkman, W.A. Gardner, and H.B. Peeler. 2004. A survey of ground-dwelling ants (Hymenopter: Formicidae) in Georgia. Florida Entomologist 87:253-260.

Izhaki, I., D.J. Levey, and W.R. Silva. 2003. Effects of prescribed fire on an ant community in Florida pine savanna. Ecological Entomology 28:439-448.

Jonas, J.L., M.R. Whiles, and R.E. Charlton. 2002. Aboveground invertebrate responses to land management differences in a central Kansas grassland. Environmental Entomology. 31:1142-1152.

Jusino-Atresino, R. and S.A. Philips, Jr. 1994 Impact of red imported fire ants on the ant fauna of central Texas. Pages 258-269 In D.F. Williams, editor. Exotic ants: biology, impact, and control of introduced species. Boulder, CO: Westview Press.

Mackay, W. and E. Mackay. 2003. The ants of North America. Centennial Museum, Laboratory for Environmental Biology. The University of Texas, El Paso, TX.

Martin, T.E. 1995. Avian life history evolution in relation to nest sites, nest predation, and food. Ecological Monographs 65:101-127.

Mayer, A.L. and G.N. Cameron. 2003. Landscape characteristics, spatial extent, and breeding bird diversity in Ohio, USA. Diversity and Distributions 9:297-311.

McCraken, D.I., E.M. Bigmal, S. Blake, and G.N. Foster. 2004. Productivity and profitability: the effects of farming practices on the prey of insectivorous birds. Pages 75-87 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

McIntyre, N.E. and T.R. Thompson. 2003. A comparison of conservation reserve program habitat plantings with respect to arthropod prey for grassland birds. American Midland Naturalist 150:291-301.

Moreby, S.J. 2004. Birds of lowland arable farmland: the importance and indentification of invertebrate diversity in the diet of chicks. Pages 21-35 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Pubishers.

Natural Resource Spatial Analysis Laboratory. 1998. 1998 land cover map of Georgia. Institute of Ecology, University of Georgia, Athens, GA.

Nudds, T.D. 1977. Quantifying the vegetation structure of wildlife cover. Wildlife Society Bulletin 5:113-117.

Porter, S.D. and D.A. Savignano. 1990. Invasion of polygyne fire ants decimates native ants and disrupts arthropod community. Ecology 71:2095-2106.

Porter, S.D. and W.R. Tschinkel. 1987. Foraging in *Solenopsis invicta* (Hymenoptera: Formicidae): effects of weather and season. Environmental Entomology 16:802-808.

Rasnake, M. 1992. Management of warm season bunch grasses. Proceedings of the Southern Pasture and Forage Crop Improvement Conference 48:38-39.

Robel, R.J., B.M. Press, B.L. Henning, K.W. Johnson, H.D. Blocker, and K.E. Kemp. 1995. Nutrient and energetic characteristics of sweepnet collected invertebrates. Journal of Field Ornithology 66:44-53.

Rodewald, A.D. and A.C. Vitz. 2005. Edge- and area-sensitivity of shrubland birds. Journal of Wildlife Management 69:135-151.

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

Samson, F. and E.L. Knopf. 1994. Prairie conservation in North America. Bioscience 44:418-422.

SAS Institute 2005. SAS OnlineDoc version 9.1.3. Cary, NC: SAS Institute, Inc.

Schotzko, D.J. and L.E. O'Keeffe. 1986. Comparison of sweepnet, d-vac, and absolute sampling for *Lygus Hesperus* (Heteroptera: Miridae) inlentils. Journal of Economic Entomology 79:224-228.

Schotzko, D.J. and L.E. O'Keeffe. 1989. Comparison of sweepnet, d-vac, and absolute sampling, and diel variation of sweepnet sampling estimates in lentils for pea aphid (Homoptera: Aphididae), nabids (Hemiptera: Nabidae), lady beetles (Coleoptera: Coccinellidae), and lacewings (Neuroptera: Chrysopidae). Journal of Economic Entomology 82:491-506.

Shochat, E., M.A. Patten, D.W. Morris, D.L. Reinking, D.H. Wolfe, and S.K. Sherrod. 2005a. Ecological traps in isodars: effects of tallgrass prairie management on bird nest success. Oikos 111:159-169.

Shochat, E., D.H. Wolfe, M.A. Patten, D.L. Reinking, and S.K. Sherrod. 2005b. Tallgrass prairie management and bird nest success along roadsides. Biological Conservation 121:399-407.

Standen, V. 2000. The adequacy of collecting techniques for estimating species richness of grassland invertebrates. Journal of Applied Ecology 37:884-893.

Stiles, J.H. and R.H. Jones. 1998. Distribution of the red imported fire ant, *Solenopsis invicta*, in road and powerline habitats. Landscape Ecology 13335-346.

ter Braak, C.J.F. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. Ecology 67:1167-1179.

ter Braak, C. J. F., and P. Šmilauer. 2002 CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 4.5). Ithaca, NY: Microcomputer Power.

USDA Forest Service. 2004. Chattahoochee-Oconee National Forests land and resource management plan. U.S. Department of Agriculture, Forest Service, Southern Region. Report Management Bulletin R8-MB 113 A.

Vogl, R.J. 1973. Fire in the Southeastern grasslands. Proceedings of the Tall Timbers Fire Ecology Conference 12:178-193.

Vogt, J.T., R.A. Grantham, E. Corbett, S.A. Rice, and R.E. Wright. 2002. Dietary habitats of *Solenopsis invicta* (Hymenoptera: Formicidae) in four Oklahoma habitats. Environmental Entomology 31:47-53.

Whiles, M.R. and R.E. Charlton. 2006. The ecological significance of tallgrass prairie arthropods. Annual Review of Entomology 51:387-412.

Wojcik, D.P. 1994. Impact of the red imported fire ants on native ant species in Florida. Pages 269-281 *in* D.F. Williams, editor. Exotic ants: biology, impact, and control of introduced species. Boulder, CO: Westview Press.

Wojcik, D.P., C.R. Allen, R.J. Brenner, E.A. Forys, D.P. Jouvenaz, and R.S. Lutz. 2001. Red imported fire ants: impact on biodiversity. American Entomologist 47:16-23.

Zimmerman, J.L. 1992. Density-independent factors affecting the avian diversity of the tallgrass prairie community. Wilson Bulletin 104:85-94.

Figure 3.1. Map of the state of Georgia showing the 4 counties where field work occurred in 2002-2004. Research investigated 7 land management prescriptions for managing early successional habitat at the transition between the Georgia Piedmont and Upper Coastal Plain ecological regions.

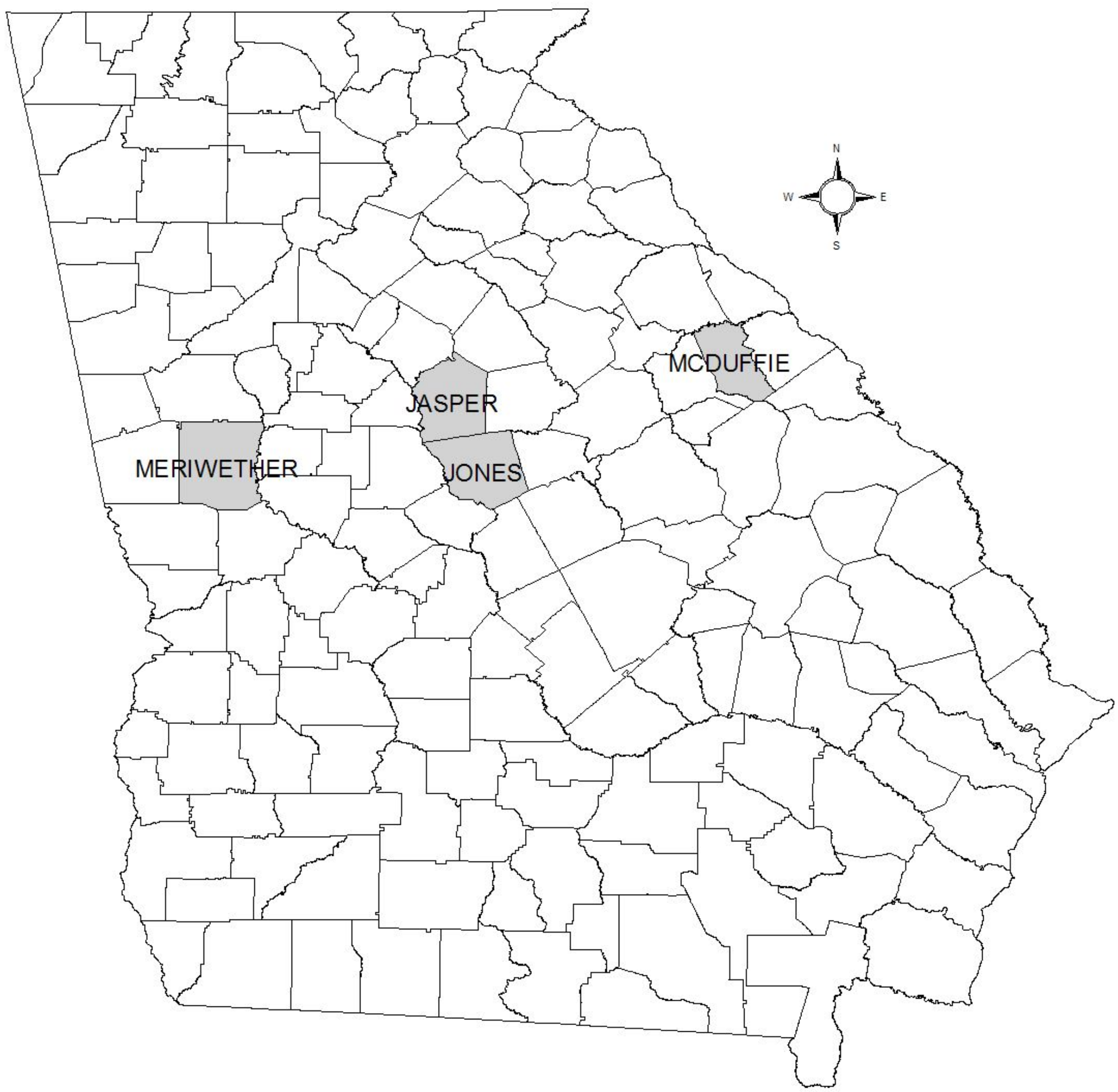
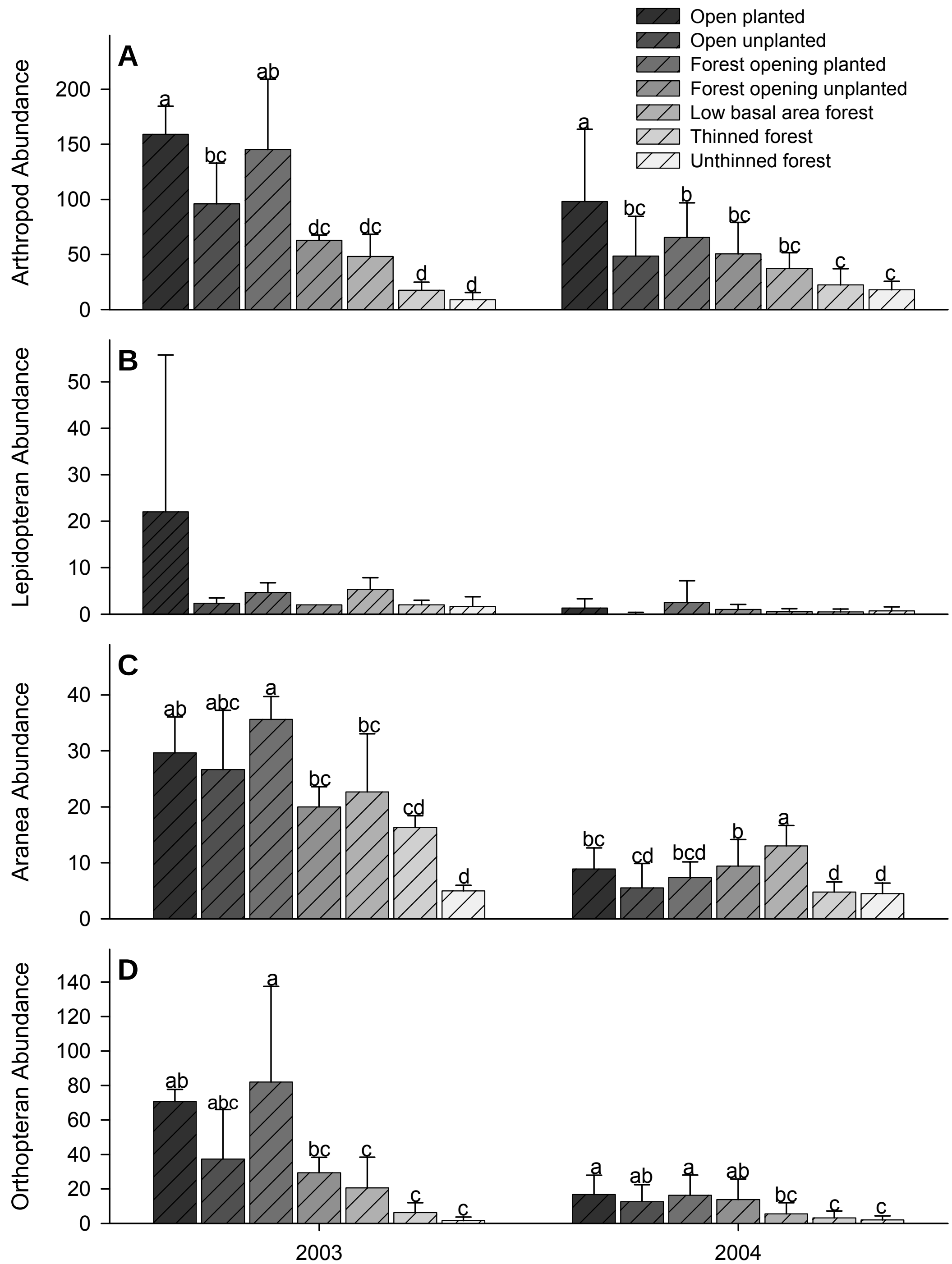

N
W
E
S
MCDUFFIE
JASPER
MERIWETHER
JONES

Figure 3.2. Arthropod abundance in seven land management prescriptions for maintaining early successional habitat in central Georgia, 2003-2004 (2003 n=3; 2004 n=9). Bars represent standard error. Different letters within a year indicate a significant difference ($P \leq 0.05$) for Duncan's multiple range test.


Open planted
Open unplanted
Forest opening planted
Forest opening unplanted
Low basal area forest
Thinned forest
Unthinned forest
A
Arthropod Abundance
a
bc
ab
dc
dc
d
d
a
bc
b
bc
bc
c
c
B
Lepidopteran Abundance
C
Aranea Abundance
ab
abc
a
bc
bc
cd
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cd
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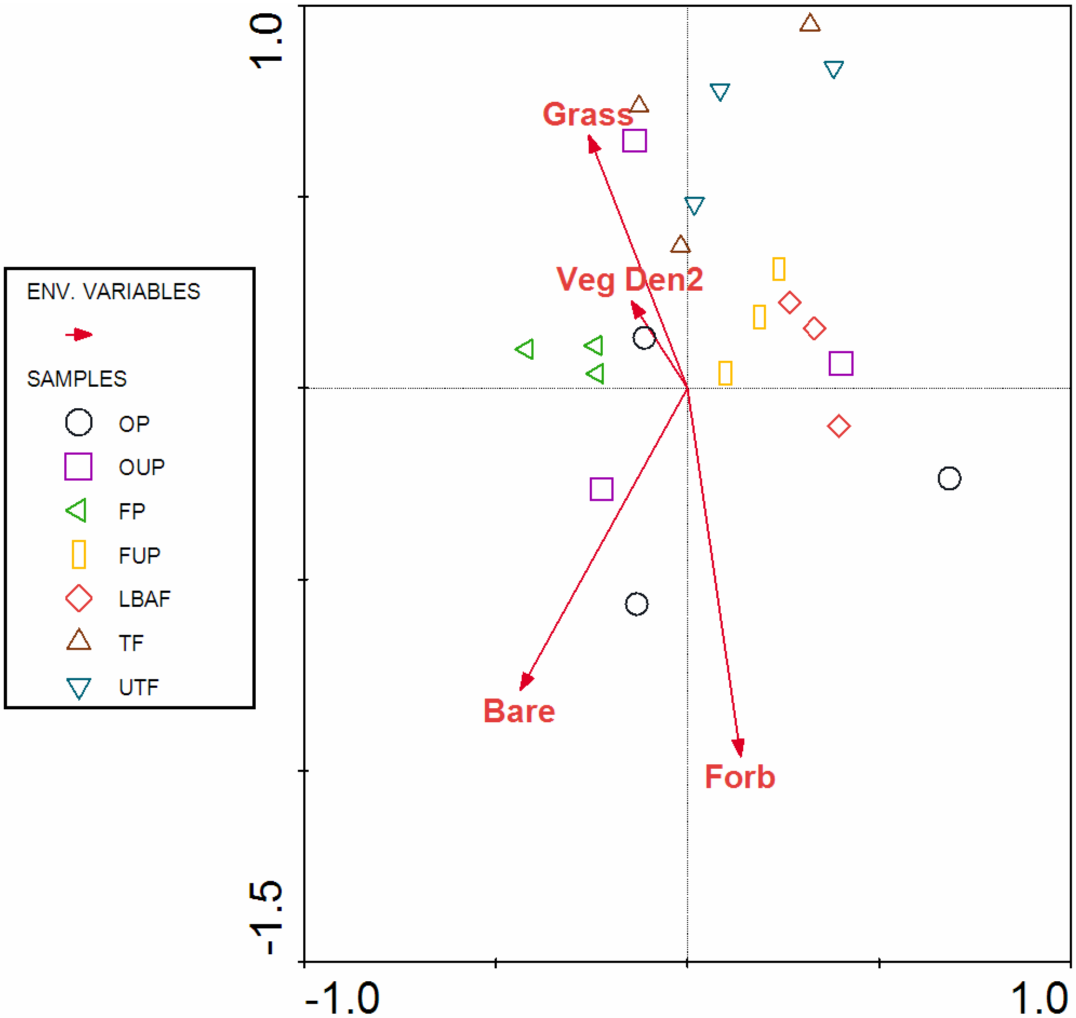
a
d
d
D
Orthopteran Abundance
ab
abc
a
bc
c
c
c
a
ab
a
ab
bc
c
c
2003
2004

Figure 3.3. Canonical correspondence analysis (CCA) biplot of plots and environmental variables for 2003 arthropod abundance data collected using sweep nets for 7 early successional land management prescriptions in central Georgia, USA. Arrows represent the 4 best environmental variables selected using Monte-Carlo forward selection; length of the arrow indicates the importance of the variable to the ordination. (OP=open planted, OUP=open unplanted, FP=forest opening planted, FUP=forest opening unplanted, LBAF=low basal area forest, TF=thinned forest, UTF=unthinned forest)



1.0
-1.5
-1.0
1.0
Grass
Veg Den2
Bare
Forb
ENV. VARIABLES
SAMPLES
OP
OUP
FP
FUP
LBAF
TF
UTF

Figure 3.4. Canonical correspondence analysis (CCA) biplot of plots and environmental variables for 2004 (a. April; b. May; c. June) arthropod abundance data collected using sweep nets for 7 early successional land management prescriptions in central Georgia, USA. Arrows represent the 4 best environmental variables selected using Monte-Carlo forward selection; length of the arrow indicates the importance of the variable to the ordination. (OP=open planted, OUP=open unplanted, FP=forest opening planted, FUP=forest opening unplanted, LBAF=low basal area forest, TF=thinned forest, UTF=unthinned forest)

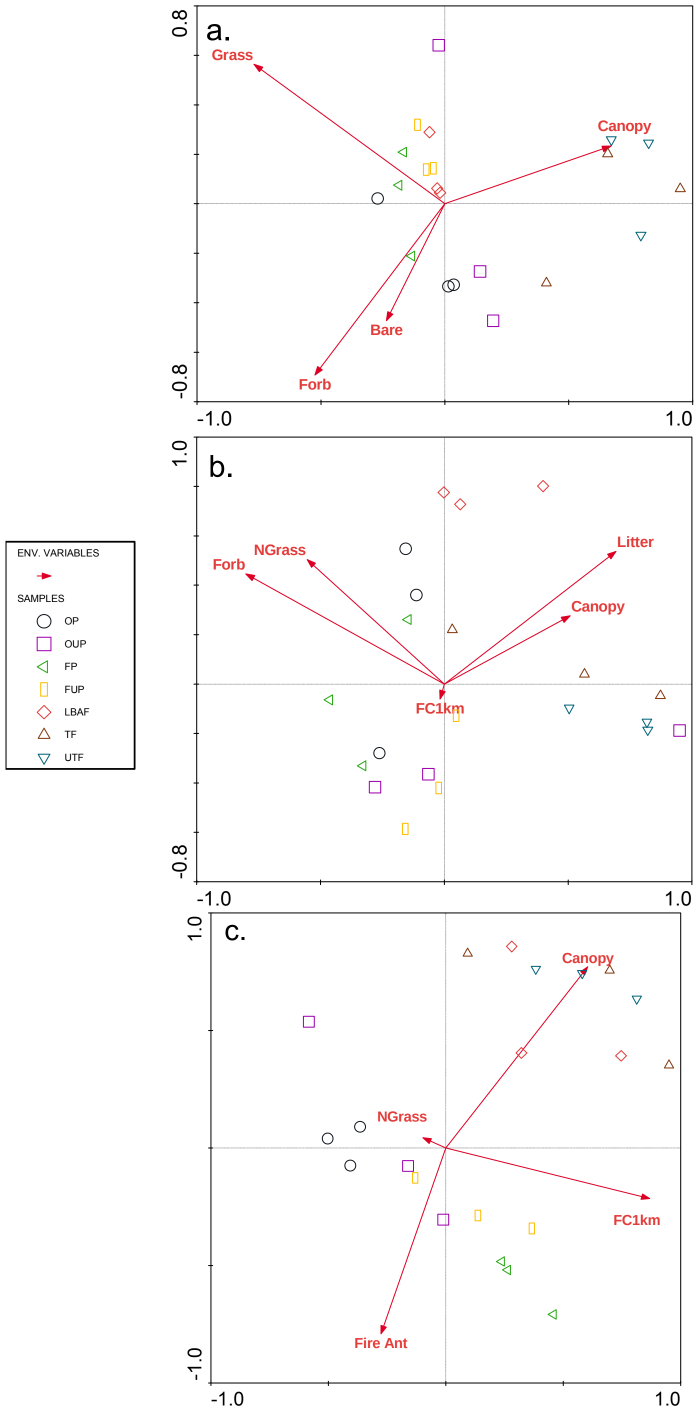

a.
Grass
Canopy
Bare
Forb
0.8
-0.8
-1.0
1.0
b.
NGrass
Forb
Litter
Canopy
FC1km
1.0
-0.8
-1.0
1.0
c.
Canopy
NGrass
FC1km
Fire Ant
1.0
-1.0
-1.0
1.0
ENV. VARIABLES
SAMPLES
OP
OUP
FP
FUP
LBAF
TF
UTF

Table 3.1. Experimental management prescriptions for maintaining early successional habitat in central Georgia, 2002-2004.

Treatment	Code	Description	Location
Open planted	OP	Old fields planted with mixture of native grasses in spring 2002	Joe Kurz WMA
Open unplanted	OUP	Old field habitat maintained with burning and mowing	Joe Kurz WMA
Forest opening planted	FP	Forest openings planted with native grasses in spring 2002	Piedmont NWR
Forest opening unplanted	FUP	Forest openings maintained with mowing and burning	Piedmont NWR
Low basal area forest	LBAF	Upland pine forest thinned 1980s, 9 m^2/ha basal area, 3-year fire rotation	Clark's Hill WMA
Thinned forest	TF	Upland pine thinned 1990s; maintained with fire	Oconee NF
Unthinned forest	UTF	Upland pine forest without recent thin; maintained with fire	Oconee NF

Table 3.2. Modified Braun-Blanquet scale for vegetation cover determination in grassland habitat.

Cover Score	Grass Cover
0	0%
1	0-5%
2	5-25%
3	25-50%
4	50-75%
5	75+

Table 3.3. Richness of arthropods in seven land management treatments for maintaining early successional habitat in central Georgia (n=3 for each treatment). Different letters across a row indicate a significant difference ($P \leq 0.05$) for Duncan's multiple range test.

Richness	Treatment						
	OP	OUP	FP	FUP	LBAF	TF	UTF
2003							
All Arthropods	25.7 ± 2.9[a]	22.3 ± 4.7[ab]	26.0 ± 4.6[a]	24.0 ± 2.6[a]	21.3 ± 5.9[ab]	14.7 ± 7.4[bc]	10.0 ± 4.6[c]
Orthopterans	2.3 ± 0.6	2.0 ± 1.0	3.0 ± 1.0	2.0 ± 0.0	2.7 ± 0.6	1.0 ± 1.0	1.0 ± 1.0
Lepidopterans	1.3 ± 0.6	1.3 ± 0.6	2.7 ± 0.6	1.7 ± 0.6	1.7 ± 0.6	1.0 ± 0.0	1.3 ± 1.5
2004							
All Arthropods	22.0 ± 4.9[a]	15.6 ± 5.8[bc]	17.8 ± 4.1[abc]	18.6 ± 3.4[ab]	17.8 ± 4.1[abc]	16.7 ± 5.3[bc]	13.8 ± 2.5[c]
Orthopterans	2.1 ± 0.8	1.8 ± 0.7	2.1 ± 0.6	2.0 ± 1.3	2.2 ± 1.1	2.1 ± 1.2	1.8 ± 0.4
Lepidopterans	1.3 ± 0.9	0.3 ± 0.5	1.4 ± 1.3	1.2 ± 0.7	0.9 ± 1.0	0.9 ± 0.8	0.9 ± 0.6

Table 3.4. Pearson correlation coefficeints (r) of mean number of fire ants relative to ant and arthropod community measures in central Gerogia, USA for May and June 2004 (n=21 for each comparison).

Community Metric	May	June
Abundance		
Ant	-0.43[a]	-0.80[c]
Arthropod	0.72[c]	0.49[a]
Aranea	0.52[b]	0.23
Orthoptera	0.71[c]	0.63[b]
Lepidoptera	0.04	0.32
Richness		
Ant	-0.63[b]	-0.78[c]
Arthropod	0.68[c]	0.02
Orthoptera	0.23	-0.10
Lepidoptera	0.01	0.31

[a]$P < 0.05$,[b]$P < 0.01$,[c]$P < 0.001$

Table 3.5. P-values from Monte-Carlo forward selection permutation test of envrionmental variables used in canonical correspondence analysis of arthropod community data from 7 land management strategies in central Georgia, 2003-2004. P-values in **bold** indicate that variable contributed significantly to the model.

Code	Definition	June 2003	April 2004	May 2004	June 2004
Grass(cm)	Grass height in cm				
Grass	Average grass cover (%) in plot	**0.03**	**0.001**		
NGrass	Average native grass cover (%) in plot			**0.009**	0.08
Forb	Average forb cover (%) in plot	0.10	**0.01**	**0.001**	
Woody/Shrub	Average shrub cover (%) in plot				
Bare	Average amount of bare ground (%)	0.07	0.34		
Canopy	Canopy cover (%)		**0.05**	**0.05**	**0.002**
Litter	Average litter depth (cm)			**0.001**	
Veg Den2	Vegetation density (%) 0.5-1.0m above ground	0.13			
Veg Den1	Vegetation density (%) 0.0-0.5m above ground				
FC1km	Forest cover in 1-km buffer around plot			**0.005**	**0.001**
Fire Ant*	Average number of fire ants in plot				0.13

*Only added to environmental variables for May and June 2004 CCAs

CHAPTER 4

RESPONSE OF SUMMER AND WINTER BIRD COMMUNITIES TO LAND MANAGEMENT TECHNIQUES FOR EARLY SUCCESSIONAL HABITAT IN THE SOUTHEASTERN UNITED STATES

[3]McMellen, A.B. and S.H. Schweitzer. To be submitted to *Journal of Wildlife Management.*

ABSTRACT

Southeastern early-successional habitat has experienced large-scale conversion to high-intensity agriculture, pine plantations, and exotic grass pastures. Many native grasses have been eliminated. Birds that depend on grasslands for breeding and/or wintering habitat have experienced precipitous declines. We established 21 plots, from 1.2–4.0 ha each, at 4 locations in central Georgia to investigate bird community response to current management practices that promote early successional habitat and to experimental native grass restoration. Study plots ranged from open agricultural land to open pine stands. During spring 2002, 6 plots were planted with a combination of big bluestem, little bluestem, switch grass, and Indian grass. All remaining sites were burned during winter 2001-2002. Mist netting and vegetation measurements were conducted in all plots each spring, 2002-2004, and winter, 2003-2004, to evaluate avian and vegetative response to the various land management alternatives. Data were analyzed using one-way analysis of variance and canonical correspondence analysis (CCA). Avian richness, diversity, total conservation value (TCV), and total abundance did not differ among treatments during the breeding season. Abundance of specific groups did vary between treatments. Grass-herb birds were most abundant in open pine stands. Forest-disturbance birds were most abundant in old field habitat. Grass-herb birds were positively correlated with grass cover. Shrub-scrub and forest-disturbance birds were negatively correlated with forest cover within a 1 km buffer. Local (vegetation density, litter depth, grass cover) and landscape (forest cover, canopy cover) habitat variables contributed to breeding bird community structure according to CCA. Winter bird richness was greatest in open native grass plots, thinned forest, and

old field habitat. Open native grass plots had high abundance of grass-herb birds. Forest-disturbance birds were more abundant in forest patches than more open habitat and were negatively correlated with grass cover. Winter grass-herb and shrub-scrub birds were negatively correlated with forest cover within a 1-km buffer. Local habitat variables were most important in explaining avian community structure during the winter. To maintain a diverse breeding and wintering bird community, a diversity of early successional habitat types should be maintained in the landscape. Small patches of early successional habitat (1.2–4 ha) can provide conservation benefit to breeding and wintering birds.

INTRODUCTION

Forty percent of North American birds dependent on a frequently disturbed ecosystem experienced a significant decline between 1969 and 1998 (Brawn et al. 2001). Grassland birds are undergoing widespread population declines that are steeper and more consistent than any other group of birds (Knopf 1994). Breeding Bird Survey data from 1966-1996 in the eastern U.S. revealed significant population declines for 13 of 14 grasslands birds while only 2 of 40 forest birds had declining trends (Peterjohn and Sauer 1999, Askins 2000). Fire suppression, declining beaver populations, urbanization, aforestation, short rotation pine plantations, intensive agriculture, and introduction of exotic pasture grasses have all been implicated in the decline of eastern disturbance dependent birds (Askins 2000).

Several songbird species are closely associated with the structure of the native bunchgrass-forb communities that formed the understory of the longleaf pine (*Pinus palustris*) and slash pine (*P. elliotti*) forests of the past. Partners in Flight identified the

Henslow's sparrow (*Ammodramus henslowii*), Bachman's sparrow (*Aimophila aestivalis*), red-cockaded woodpecker (*Picoides borealis*), and prairie warbler (*Dendroica discolor*) as priority species in the pine savanna and grassland ecosystems of the southeast (Cooper 1999). Other species warranting attention include the loggerhead shrike (*Lanius ludovicianus*), northern bobwhite (*Colinus virginianus*), eastern meadowlark (*Sturnella magna*), and savannah sparrow (*Passerculus sandwichensis*).

The landscape of the Southeast has undergone extensive human modification. Before European settlement, vegetation in the Georgia Piedmont was dominated by fire-tolerant species (Cowell 1998). Native Americans modified the landscape through centuries of fire use, cultivation, and other activities (Denevan 1992). Southeastern Piedmont and Coastal Plain regions were covered by pine savannas that included plants from the Great Plains tall-grass prairies as well as Appalachian balds (Vogl 1973). In his travels through the Southeast in the 1700s, William Bartram described pine parklands, grassy openings, and large cane breaks (Harper 1998) that provided abundant habitat for many grassland birds in the southeastern United States.

Intense grazing pressure on native grasses from large herds of livestock was introduced by the Europeans. By the mid-1700s, large herds of free-ranging cattle began to damage the wire grass (*Aristida stricta*) community within pine savannas, and by the 1840s, range conditions were degraded by overgrazing (Gray 1933). Exclusion of fire, intensive grazing, and the introduction of cultivated, exotic, sod-forming grasses resulted in an increase in hardwood tree and shrub cover, changes in herbaceous species composition, and loss of native grasses and the near extirpation of native warm-season species such as switch grass (*Panicum virgatum*), big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), Indian grass (*Sorghastrum nutans*), and eastern gamma grass (*Tripsacum dactyloides*) (Rasnake 1992). Currently, most southeastern pastures are planted in introduced cool and warm-season grass species such as fescue (*Festuca arundinacea*), bermuda (*Cynodon dactylon*), and bahia grass (*Paspalum notatum*). Exotic grass pastures may be good for widespread bird species, but grassland specialists are rarely found within these pastures (DeVault et al. 2002).

An increasing percentage forest cover within a landscape negatively impacts the abundance and occupancy of grassland birds in some areas (King et al. 2001, Ribic and Sample 2001, Berg 2002), but large patches of treeless grassland have probably never been common in the southeastern United States. Early successional habitat in the Southeast exists in a spectrum from open, agricultural land to the grassy understory of open woodlands. Many Eastern birds are better defined as disturbance-dependent rather than grassland birds. These bird species are not restricted to a single habitat type, but exist across a spectrum ranging from open grasslands to gaps in mature forests (Hunter et al. 2001a). The impact of a tree component within early successional habitat on the avian community is not known for the Southeast.

Management for early successional bird species in the eastern United States requires an understanding of the impacts of various land management alternatives on the abundance and distribution of disturbance-dependent birds. Birds associated with early successional habitats in the southeastern United States have received little attention in the literature (see Plentovich et al. 1998, 999; Marcus et al. 2000). Most research on, and restoration of these habitats in the Southeast have focused on the needs of the red-cockaded woodpecker. Management for the red-cockaded woodpecker has had a positive impact on shrub and grassland associated birds (Conner et al. 2002). Wood et al. (2004) found that several species of management concern were more common in the pine-grassland restoration areas of Mississippi than in traditional forest management areas of the southeastern United States. Many grassland associated birds in the Southeast continue to decline, thus additional investigations are warranted.

Eastern grasslands are fire dependent systems and most management has focused on re-introducing fire. However, prescribed burning alone may not be enough to restore native vegetation if the local seed bank is impoverished and seed dispersal is limited by a fragmented landscape (Kindscher and Tieszen 1998, Bakker and Berendse 1999). The Partners in Flight migratory bird program set habitat objectives for the South Atlantic Coastal Plain region that include restoring or converting cool-season grass (e.g., fescue) pastures to native warm-season grasslands on more than 10 million acres (4 million hectares; Hunter et al. 2001b).

The Southeast provides both breeding and wintering habitat to early successional birds. Many early successional birds are short distance migrants that winter in the southern United States and northern Mexico (Root 1988). Winter is an often neglected part of the annual cycle of birds. Little research has focused on wintering birds within the United States (but see Grzybowski 1982, 1983; Gordon 2000a, 2000b; Tucker and Robinson 2003). In the Southeast, the winter habitat needs of the declining Henslow's sparrow have received attention (Plentovich et al. 1996, 1998; Tucker and Robinson 2003), but there are still large gaps in our knowledge. Winter ecology and habitat use are research priorities for conservation of early successional birds (Vickery et al. 2000, Vickery and Herkert 2001).

Priorities for grassland bird research include multiple-species management, replacement of exotic grasses with native grasses, and wintering ecology (Brennan and Kuvlesky 2005). The purpose of this study was to examine avian community responses to land management techniques for early successional, disturbance-dependent birds in the Southeast during the summer and winter. We wanted to compare the ability of current land management techniques, that range from restoring native grasses to periodic burning of pine stands, to support communities of disturbance dependent birds during breeding and wintering seasons. We also sought to identify environmental variables that were driving the community structure across the spectrum of early successional habitat in the southeastern United States. Contrary to research in the Midwest, we hypothesized that abundance of breeding populations of southeastern early successional birds would not be negatively impacted by the percentage of forest cover in the landscape. We hypothesized that short-distance migrants that winter in the Southeast would be negatively associated with percentage of forest cover in the landscape.

STUDY AREAS AND TREATMENTS

To assess the impact of common management strategies used to maintain disturbance habitat, we conducted field work at Piedmont National Wildlife Refuge (NWR), Joe Kurz Wildlife Management Area (WMA), Oconee National Forest (NF), and Clark’s Hill WMA in central Georgia. All locations are in the transition zone from the piedmont to coastal plain physiographic areas, and within the historic range of pineland savannah habitat (Vogl 1973). Piedmont NWR and Oconee NF are located in Jones and Jasper Counties (Figure 4.1). Joe Kurz WMA is located in Meriwether County and Clark's Hill WMA is in McDuffie County.

Seven management strategies for creating and maintaining early successional or disturbance-dependent habitat were investigated across the 4 locations. Each management strategy (treatment) was replicated in 3 plots for a total of 21 study plots (Table 4.1). Each plot was 1.2 to 4 ha and separated by ≥1 km. All plots were burned during winter 2001-2002.

Joe Kurz WMA was a privately held farm until it was acquired by the Georgia Department of Natural Resources (GA DNR) in 1995. This site had the most open landscape of our 4 sites. The landscape surrounding our study sites was 70-75% forested with row crop fields, pasture, and hay fields accounting for most of the remaining land cover in the area. Joe Kurz was approximately 1,495 ha of old field habitat, upland loblolly pine (*Pinus taeda*), and bottomland oaks (*Quercus* spp.). Most of the open area at Joe Kurz WMA was old field habitat consisting of grasses, forbs, shrubs, and scattered saplings. Dominant grasses were bahia (*Paspalum notatum*), broomsedge (*Andropogon virginicus*), and fescue (*Festuca arundinacea*). Blackberry (*Rubus* spp.) was the most common woody plant. Lolblolly pine and sweetgum (*Liquidambar styritifolia*) accounted for most of the saplings. Three old field sites (OP treatment) were planted in 2002 with native warm season grasses (see Chapter 2 for details). Three additional old fields (OUP treatment) with similar size, shape, and surrounding landscape were left under current management of annual mowing and periodic burning.

Piedmont NWR is approximately 14,000 ha of upland pine habitat managed by the U.S. Fish and Wildlife Service. The landscape was over 95% forested consisting of loblolly pine canopy with an understory of flowering dogwood (*Cornus florida*), redbud (*Cercis canadensis*) and various native and non-native grasses. The bottomlands are mixed hardwood dominated by oaks (*Quercus* spp.). The endangered red-cockaded woodpecker is the primary management focus. Large areas of the refuge are burned on a 3-5 year rotation in an effort to create and maintain open pine parklands that are the preferred foraging habitat of this woodpecker. Six forest openings at Piedmont NWR were used as study plots. The forest openings were originally established as food plots for white-tailed deer (*Odocoileus virginianus*), eastern wild turkey (*Meleagris gallopavo*), and other wildlife, but have not been used in that capacity since the 1970s. Since that time, the openings have been managed as early successional habitat using annual mowing and periodic burning. The forest openings were old field habitat similar in structure and species composition to the fields at Joe Kurz. Three forest openings (FP treatment) were planted with native warm season grasses in spring 2002. Three additional forest openings (FUP treatment) similar in size and surrounding landscape composition, were left under current management of annual mowing and periodic burning.

Clark's Hill WMA is approximately 5,139 ha of land leased from the Army Corps of Engineers and managed by GA DNR. Most of the habitat was upland loblolly. Several sections of pine were thinned in the early 1980s and have been maintained with frequent (3 year rotation) low-intensity winter fire, resulting in pine stands with 9-11 m^2/ha basal area and a floristically diverse understory of native grasses (*Panicum anceps*, *Sorghastrum nutans*, and *Andropogon gyrans*) and forbs. Three of these pine stands were selected as study plots for this study (LBAF treatment).

Oconee NF is 46,393 ha of forested land in central Georgia. More than 70% of the forest is dominated by loblolly and shortleaf (*Pinus echinata*) pines. Established in 1959, most of Oconee NF consists of pine stands between 60-80 years old. The forest was under multi-use management. Considered a potential key site for red cockaded woodpecker conservation and restoration, the forest management plan aimed to create or restore 400 ha of woodlands, savannahs, and grasslands, and thin an additional 500 ha of lobllolly and shortleaf pine (USDA Forest Service 2004). We chose 6 stands that had been maintained with 2 management strategies. Three stands (TF treatment) were thinned between 1992 and 1995, and were maintained with fire on a 3-5 year rotation. Three stands (UTF treatment) had not been thinned but were maintained with frequent fire.

METHODS

Habitat Assessment

Landscape and local habitat variables were measured in all study plots to examine environmental differences underlying avian community structure. Percent forest cover in the landscape was determined using Georgia GAP data based on 1998 landcover and 30-m resolution (Natural Resource Spatial Analysis Laboratory 1998). Buffers of 500 m and 1000 m were created around the GPS centroid of each plot. We determined the percentage of forest cover of the landscape within each buffer. All types of forest (deciduous, evergreen, mixed, and wetland) were included in our definition of forest cover.

All local vegetation measurements except shrub cover were determined in 1-m^2 square sample plots. Permanent sample plots were distributed through study plots in a grid-like fashion. Each sample plot was ≥10 m from the study plot edge and ≥40 m from other sample plots. Data were collected from permanent sample plots during July 2002, 2003, and 2004 to minimize our impacts on nesting birds.

The percentage of grass cover was visually estimated in each 1-m^2 square sample plot using a modified Braun-Blanquet cover scale (Table 4.2). We identified all species of forbs, grasses, and woody plants within each sample plot. Litter depth and grass height were measured in the center of each sample plot. A 2.0-m cover board (Nudds 1977) was used to estimate vertical vegetation density (%) in the first 2 0.5-m strata (0-0.5 m and 0.5-1.0 m) by an observer 10 m from the center of the sample subplot. Cover board measurements were taken in the 4 cardinal directions and averaged to determine a value for each sample plot. Canopy cover was categorized as present (1) or absent (0) for each subplot using a densitometer. These values were converted to percent canopy cover for each plot. In 2004, we used 20 20-m transects to determine shrub cover using the line intercept method. During the winter, we measured vertical vegetation density, litter depth, and grass height within the permanent sample plots.

Avian Community Assessment

Breeding season-We sampled the bird community using mist nets from late April to early July 2002-2004 (UGA AUCP Permit A2002-10095). Approximately every 10 days, 12-m long mist nets with 36-mm mesh were erected in each plot. In 2002, 2 nets were used in each plot, but in 2003 and 2004, 4 nets were used. Net locations from 2002 were maintained and 2 additional locations added for the new nets. Nets were arranged in a V formation (Martin 1969). We conducted 6 rounds of mist netting in each study plot. Nets were opened at sunrise and closed 6 hours later or when the temperature reached 29 C. Species, weight, wing chord, bill length, tail length, sex, breeding status, and age were recorded for each bird captured. In addition, each bird was banded with a uniquely numbered USGS aluminum leg-band for future identification (BBL Permit #22746).

Winter-Mist netting was conducted to assess avian community indices in study plots during January-March 2003-2004. Approximately every 10 days, 4 12-m long mist nets with 36-mm mesh were erected in each plot for a total of 12 nets per treatment. Nets were erected at the same location during each sampling period. We conducted 3 rounds of mist netting in each study plot each year. Nets were opened at sunrise and closed after 6 hours, but were not opened on days with adverse weather such as extreme cold, rain, or high winds. Species, weight, wing chord, bill length, tail length, sex, and age were recorded for each bird captured. In addition, each bird was banded with a uniquely numbered USGS aluminum leg-band for future identification.

Statistical Analysis

Diversity, richness, total conservation value (TCV), and an index of abundance were calculated for all plots using breeding season and winter mist net data. The transformed Shannon diversity index (exp H’) was used because it is better able to detect diversity differences between areas or treatments (Kempton and Taylor 1976).

Richness was defined as the number of species detected in the plot during all sampling within a season. We used capture rate (birds/100 net hours) as an index of abundance.

To assess the overall avian conservation value of our treatments, we calculated a TCV index for each study plot during each breeding and winter (Nuttle et al. 2003). Partners in Flight (PIF) priority scores were used to give each bird species a rank from 0-4. Ranks were used instead of the summed PIF priority scores to avoid previously identified problems with multicolinearity (Bessinger et al. 2000, Nuttle et al. 2003). Each species was given a breeding season and winter rank (Table 4.3). Species PIF rank was multiplied by abundance for each species in the plot during each season. Capture rate was used as an index of relative abundance for the calculations. Scores were summed to give a total conservation value for each plot.

To assess the utility of the management treatments for different categories of disturbance dependent birds, species were assigned to 5 habitat association categories based on Hunter et al. (2001a; Table 4.3). Grass-herb birds are associated with ground cover dominated by forbs and grass. Shrub-scrub birds are found in areas of shrub cover or along forest edges. Disturbed-woodland birds are those species associated with open pine or pine-oak savannah communities. Forest-disturbance birds are associated with disturbances in otherwise mature forests. The remaining birds were placed in a fifth 'other' category that generally consisted of mature forest birds. Abundance of birds in each disturbance habitat category was calculated for each study plot during each season.

One-way analysis of variance (ANOVA) was used to compare richness, diversity, and TCV of breeding and winter season bird communities. An ANOVA was also used to detect differences in total abundance, as well as abundance of birds in each habitat category among the 7 treatments. Duncan's multiple range test was used for post hoc comparisons of significant ANOVA results ($\alpha = 0.05$). Pearson's correlation coefficients were calculated to examine relationships between avian community metrics and vegetation variables on local (grass cover) and landscape (forest cover) scales that best characterized management treatments. Correlation coefficients were calculated separately for breeding season and winter data. Data were pooled among years for each season. The ANOVAs and Pearson's correlations were conducted SAS using PROC GLM and PROC CORR (SAS Institute 2005).

Avian community structure during the summer and winter was analyzed using canonical correspondence analysis (CCA) on abundance data (ter Braak 1986). The CCAs were run for each season and year combination for a total of 5 CCAs. For both the summer and winter data, a series of CCAs were run with all environmental variables to assess colinearity between variables. A variable with a variance inflation factor (VIF) >20 indicated that it was perfectly correlated with another environmental variable in the dataset and did not contribute any unique information to the ordination (ter Braak 1986). Variables with VIFs >20 were eliminated one at a time until all VIFs were under 20. The significance of the relationship between environmental variables and bird species data was tested using Monte-Carlo global permutation tests (1,000 random permutations) provided in CANOCO (ter Braak and Šmilauer 2002). Forward selection tests were performed to determine the environmental variables that best explained the species data. All CCAs were performed using CANOCO for Windows 4.5 (ter Braak and Šmilauer 2002).

RESULTS

Avian Community Assessment:

Breeding season—In 5,349.8 net hours of summer mist netting, 48 species of birds were captured (Table 4.3). Avian species richness was consistently low across all treatments with no difference among treatments (F = 0.32, df = 6, 50, *P* = 0.92; Table 4.4). Richness values ranged from 3.9 ± 2.3 species in unplanted forest opening (FUP) study plots to 5.0 ± 2.9 species in thinned forest (TF) study plots. Diversity was also relatively low and there were no treatment effects (F = 1.79, df = 6, 50, *P* = 0.12). Total conservation values did not differ among treatments (F = 0.66, df = 6, 50, *P* = 0.68). The TCVs ranged from 13.9 ± 10.4 in unplanted forest openings (FUP) to 31.7 ± 30.5 in open unplanted plots (OUP, Table 4.4).

Total bird abundance ranged from 6.51 ± 3.28 birds/100 net hours in the unthinned forest plots (UTF) to 14.36 ± 10.66 birds/100 net hours in open planted plots (OP), but no differences were detected among treatments (F = 1.05, df = 6, 50, *P* = 0.41; Table 4.4). Abundance of grass-herb associated birds was greater in open pine stands (LBAF) than all other treatments (F = 8.45, df = 6, P < 0.0001). Shrub-scrub birds were more abundant in OP and OUP than in all other treatments (F = 2.38, df = 6, 50, *P* = 0.04). The abundance of disturbed-woodland birds did not differ among treatments (F = 0.32, df = 6, 50, *P* = 0.92), but ranged from 0 birds/100 net hours in UTF plots to 2.58 ± 2.45 birds/100 net hours in thinned forest plots (TF) . The abundance of forest-disturbance birds differed among treatments (F = 2.55, df = 6, 50, *P* = 0.03) and ranged from 0.31 ± 0.48 birds/100 net hours in UTF to 3.93 ± 3.75 birds/100 net hours in OUP.

Only the abundance of grass-herb birds was correlated with percentage of grass cover in study plots (r = 0.25, P = 0.05; Table 4.5). No bird community index (richness, diversity, TCV) was correlated with percentage of grass cover. Percentage of forest cover within the 500-m buffer was not correlated with any abundance measure or community index. Abundance of shrub-scrub birds (r = -0.42, P = 0.001) and forest-disturbance birds (r = -0.31, P = 0.01) were negatively correlated with percentage of forest cover within 1 km of each study plot.

Canonical correspondence analysis

Canonical correspondence analysis was run on each year of breeding season mist net and vegetation data. Elimination of all correlated variables left 12 variables in the breeding season CCAs (Table 4.6). Forward selection of environmental variables for the 2002 breeding bird data indicated that litter depth, vegetation density (0 - 0.05 m stratum), forest cover in 1-km buffer, and forb cover influenced bird community data, but none of them contributed significantly to the ordination model (Table 4.6). Eigenvalues of axes 1 and 2 were 0.74 and 0.63, respectively. Axis 1 had a negative loading for forest cover within 1 km, and a positive loading for litter depth. Axis 2 was characterized by negative loading for litter depth and positive loading for vegetation density (0-0.05 m stratum). The first two axes accounted for 23.7% of the variation in the bird data and 62.4% of the variance in the bird-environment relationship. According to Monte-Carlo permutation tests, the contribution of all canonical axes was significant (F = 1.38, P = 0.04), indicating a significant relationship between environmental variables and species data. There was overlap in the bird communities among all treatments (Figure 4.2a).

Grass cover (*P* = 0.02), forb cover (*P* = 0.001), and canopy cover (*P* = 0.02) contributed significantly to the ordination in 2003. Eigenvalues of axes 1 and 2 were 0.55 and 0.42, respectively. The first 2 axes accounted for 19.3% of the variation in the bird data and 61% of the bird-environment relationship. The relationship between environmental variables and bird data was significant (F = 1.85, *P* = 0.001). Clusters of treatment plots were more distinct in the 2003 ordination diagram (Figure 4.2b). The OP plots formed a distinct cluster with high positive loadings on axis 1, indicating high forb cover and low canopy cover. The LBAF clustered around negative loadings on both axes indicating the plots had high canopy cover, high grass cover, and low vegetation density near ground level (0-0.5 m stratum). Similar to 2002, the OP plots did not cluster closely together.

Forest cover within 1 km (*P* = 0.05), vegetation density within the 0-0.5-m stratum (*P* = 0.01), and vegetation density within the 0.5-1.0-m stratum (*P* = 0.02) contributed significantly to the 2004 ordination. Eigenvalues of axis 1 were 0.39 and 0.32 for axis 2. The first 2 axes accounted for 18.7% of the bird species data 62% of the bird-environment relationship. There was a significant relationship between the bird species data and environmental variables (F = 1.72, *P* = 0.001). Treatment plots formed clusters in 2004 (Figure 4.2c). Thinned forest (TF) and UTF plots were clustered together with positive loadings on axis 1 and negative loadings on axis 2, indicating these plots had low vegetation density (0-0.05 m stratum) and low grass cover.

Winter—Thirty-three bird species were captured in 3,391.3 net hours of winter mist netting (Table 4.7). Short distance migrants that breed in the northern United States and southern Canada and winter in the southern United States and northern

Mexico accounted for 14 species. Nineteen species were year-round residents. Management treatment influenced richness (F = 2.93, df = 6, 35, *P* = 0.02). Thinned forest plots (TF, 5.7 ± 4.2 species), OP plots (5.0 ± 2.9 species), OUP (4.3 ± 3.4 species), and planted forest openings (FP, 3.5 ± 1.5 species) had higher richness values than the LBAF (1.2 ± 1.2 species). Diversity was not influenced by treatment (F = 1.82, df = 6, 35, *P* = 0.13), but followed a similar pattern to richness (Table 4.8). Total conservation value ranged from 11.8 ± 14.0 in unthinned forest plots to 41.0 ± 32.43 in open planted plots, but differences were not significant (F = 1.99, df = 6, 35, *P* = 0.09).

Bird abundance ranged from 12.1 ± 8.6 birds/100 net hours in UTF plots to 23.2 ± 19.9 birds/100 net hours in OP plots. Open planted plots (2.7 ± 3.1 birds/100 net hours) had higher abundance of grass-herb associated birds than all 3 treatments with a tree component (LBAF, TF, UTF) and the unplanted forest plots (FUP; F = 2.25, df = 6, 35, *P* = 0.01). Open planted plots (8.3 ± 7.9 birds/100 net hours) had greater abundance of shrub-scrub birds than any other treatment (F = 8.33, df = 6, 35, P < 0.001). Abundance of disturbed woodland birds did not change among treatments (F = 1.28, df = 6, 35, P = 0.29). Forest-disturbance birds were more abundant in thinned (2.1 ± 1.9 birds/100 net hours) and unthinned (1.8 ± 2.5 birds/100 net hours) forest patches than in more open treatments (F = 3.15, df = 6, 35, P = 0.01).

Grass cover was negatively correlated with abundance of forest-disturbance birds (r = -0.39, *P* = 0.01), richness (r = -0.36, *P* = 0.01), and diversity (r = -0.32, *P* = 0.04; Table 4.9). Forest cover within 500 m of the plot was not correlated with any abundance measure or community index. Forest cover within 1 km of the plot was negatively correlated with abundance of grass-herb birds (r = -0.42, *P* = 0.005) and shrub-scrub birds (r = -0.055, *P* < 0.001).

Canonical correspondence analysis was run on each year of winter mist net and vegetation data. Elimination of all correlated variables left 11 environmental variables that were considered in the breeding season CCAs (Table 4.6). Forward selection of environmental variables for the 2003 winter season indicated that exotic grass cover (*P* = 0.001), forb cover (*P* = 0.001), shrub cover (*P* = 0.04), and winter vegetation density (0 - 0.5 m stratum) were the most important explanatory variables for the ordination (Table 4.10). Eigenvalues of axes 1 and 2 were 0.69 and 0.56, respectively. Axis 1 had a negative loading for exotic grass cover and forb cover. Axis 2 was characterized by a negative loading for exotic grass cover and a positive loading for winter vegetation density (0 - 0.5 m stratum).

The first 2 axes accounted for 23.6% of the variation in the bird data and 64.9% of the variance in the bird-environment relationship. According to Monte-Carlo permutation tests, the contribution of all canonical axes was significant (F = 2.14, *P* = 0.001) indicating a significant relationship between environmental variables and species data. The ordination separated treatments into 2 fairly distinct groups (Figure 4.3a). Open treatments (OP, OUP, FUP, FP) loaded negatively on axis 1 indicating these plots had little exotic grass and high forb cover. The treatments with more forest cover (TF, UTF) had positive loadings on axis 1 indicating they had higher exotic grass cover and reduced cover of forbs. Open pinelands (LBAF) were overlapped these 2 categories.

Forward selection of 2004 winter data identified grass cover, forb cover, shrub cover, and canopy cover as explaining the most variation in the data, but only canopy cover (P = 0.01) contributed significantly to the ordination. Eigenvalues of axes 1 and 2 were 0.76 and 0.56, respectively. The first 2 axes accounted for 18.1% of the variation in the bird data and 63.3% of the bird-environment relationship. Axis 1 had high positive loading on canopy. Axis 2 had a high positive loading on forb cover. The relationship between environmental variables and bird data was nearly significant (F = 1.2, P = 0.06). As in 2003, the ordination separately the treatments into two distinct groups of open habitat treatments (OP, OUP, FP, FUP) and forest habitat treatments (TF, UTF, Figure 4.3b). Open habitat treatments had negative loadings on axis 1. The forested treatments loaded positively on axis 1. Open woodlands (LBAF) overlapped the two groups.

DISCUSSION

Abundance and TCV results suggest that accommodating the habitat needs of all disturbance-dependent species in the eastern United States will require a mosaic of successional stages in the landscape. No management strategy clearly provided superior benefit to all groups of disturbance dependent birds within or between seasons. During both summer and winter, open woodlands (LBAF) had one of the highest TCVs, but the open treatments also had high conservation value in the summer (OUP) and winter (OP).

The differences in TCV were not significant in either season suggesting that all treatments are contributing conservation value to the system. In the late nineteenth century, 55-60% of the eastern United States was young, open forest, but that percentage has dropped to less than 20% (Lorimer 2001). Managing for early successional habitat does not have to be a trade-off between mature forest birds and disturbance-dependent birds. Patches of early successional habitat within a forested landscape may actually benefit forest birds, especially immediately following breeding, and early successional species. Post breeding juvenile and adult mature forest species used regenerating clear cuts in Ohio (Vitz and Rodewald 2006) and patches of early successional habitat in Virginia (Childers et al. 1986). Shrub-scrub birds occurred in both group cuts and clear cuts with equal nesting success (King et al. 2001). Forest management that includes 2-3 ha openings might be able to benefit both mature forest birds and early successional species.

Breeding season avian richness values were not different among treatments. There are 2 likely reasons for this. Early successional systems have less niche space and support fewer species (Kobal et al. 1999). In our study, richness ranged from a 3.9 species (FUP) to 5.0 species (TF). Such a narrow response range makes detecting treatment differences difficult. Further, we sampled our plots using mist nets. Concealing mist nets in early successional habitat is very difficult (Martin 1969). Mist nets might have been avoided by species due to their visibility causing our richness and diversity indices to be underestimated.

Breeding bird community change in a successional gradient from grass to woodlands was driven by changes in vegetation density and complexity in South Africa (Skowno and Bond 2003). In order to determine which environmental variables were driving community structure, we used CCA. Forward selection of environmental variables in the breeding season CCA indicated that a mix of local and landscape level variables were shaping the bird community in our successional gradient. Differences in selected variables among years may have been due to slight changes in the bird community between years. The breeding bird community was made up of species associated with grass-herb, shrub-scrub, disturbed-woodland, forest-disturbance, and mature forest habitat (Table 4.3). (Titeux et al. 2004) suggests that local environmental factors drive species assemblages across a wide-range of habitats, but other studies have found species specific response to scale (Morimoto and Wasserman 1991, Robinson et al. 2004). It is likely that the diversity of environmental variables selected reflect the diverse nature of disturbance dependent birds in the Southeast. The bird community of a pitch pine-scrub oak forest, similar in physical structure to the pine-oak savannahs of the southeast, was determined by environmental variables at the plot, patch, and landscape scale (Grand and Cushman 2003). Similar to our findings, shrub-scrub bird occupancy was influenced by landscape level features and grassland bird occupancy by grass cover in Idaho (Knick and Rotenberry 1995).

Forward selection of the environmental variables included in our winter CCA indicated that local plot-level characteristics were more important in shaping the winter bird communities than landscape features. There were slight differences in the variables selected in 2003 and 2004 which may reflect changes in abundance and community composition between years. There is no literature on the relative importance of local and landscape variables on wintering bird communities, but changes in the area sensitivity of species on their breeding and wintering grounds point to differences in seasonal requirements. Savannah sparrows are area sensitive during the breeding season (Bakker et al. 2002), but were found in both of our open (OP and OUP) as well as the planted forest opening (FP) study plots that were not more than 4 ha in size during the winter. A similar disparity between winter and summer habitat requirements has been found for the Henslow's sparrow which also winters in the Southeast. Henslow's sparrows are considered area sensitive on the breeding ground (Herkert 1994, Kobal et al. 1999), but were found on small (<0.25 ha) patches of pitcher plant-bogs on their wintering grounds along the Gulf Coast (Tucker and Robinson 2003). Protecting small patches of early successional habitat may benefit wintering sparrows in the Southeast.

In Britain, a small (10-20%) change in forest cover change avian community composition (Berg 2002). A similar pattern was reflected in our study. Forest cover at our study sites ranged from 70-95%, but even this small range seemed to have an impact on the bird community. In our study, forest cover within a 1-km buffer was negatively correlated with the abundance of shrub-scrub birds during the summer and winter. Shrub-scrub birds in Idaho also responded negatively to forest cover in the landscape during the summer (Knick and Rotenberry 2000). The scale of the landscape attribute being measured influences its relationship to the community metric (Mayer and Cameron 2003) and this was evident from our study. Unlike forest cover within a 1-km buffer, forest cover within 500-m of our plots was not correlated with any breeding season or winter community metric. Abundance and distribution of bird species in a riparian forest was also more influenced by landscape attributes at 1-km than at a smaller scale (Saab 1999).

Grass-herb bird abundance was positively correlated to grass cover within the study plot and showed no relationship with forest cover at either scale (500-m or 1-km) during the summer, but was negatively associated with forest cover during the winter. The Bachman's sparrow was the most common breeding grass-herb bird in our study.

Bachman's sparrows breed in the grassy understory of southern pine savannahs (Cooper 1999). Our data suggest that the local characteristics of the understory may be more important than the surrounding landscape during the breeding season. Savannah and vesper sparrows comprised the grass-herb bird community during the winter. Species specific response to environmental variables (Robinson et al. 2004) may be responsible for response differences between summer and winter.

Except for restoration of native grasses, most of the land management strategies investigated in this study have been used for extended periods of time in the Southeast. Only recently has there been an interest in restoring native grass species to benefit early successional bird species. Restoration is increasingly important in regions with scarce quantities of quality habitat (Vickery et al. 1999) or where habitat change can not be remedied with less intense management alternatives (Briggs et al. 2005). The largest grasslands in the eastern United States are reclaimed coal-mines that are providing habitat for large populations of Henslow's sparrows (Bajema et al. 2001). Native warm season grass restoration has had positive impacts on breeding bird communities in Texas (Flanders et al. 2006) and Pennsylvania (Giuliano and Daves 2002). Native grass restoration did not stand out as being superior to all other treatments during the breeding season, but NWSG study plots (OP and FP) did support substantial numbers of shrub-scrub and forest-disturbance birds. In the winter, open planted study plots had large numbers of grass-herb and shrub-scrub. Native warm season grass restoration has the potential to benefit disturbance dependent species and should be added to the land manger's toolbox.

A mixture of grassland types is desirable in the landscape. The eastern meadowlark and grasshopper sparrow are two species of concern in Georgia, but they have very different habitat requirements. Grasshopper sparrows prefer light to intermediate litter layer showing up 1-2 years after a burn (Powell 2000) while eastern meadowlarks are found in areas with a deep litter and no recent disturbance (Granfors et al. 1996, Kobal et al. 1999). Modification to farmland may allow us to incorporate beneficial early successional habitat into the eastern United States. Increasing heterogeneity in the farmland landscape through edges, fallow areas, and different types of fields can make the landscape more hospitable to breeding (Benton et al. 2002) and wintering (Smith et al. 2005) early successional birds.

LITERATURE CITED

Askins, R. A. 2000. Restoring North America's birds: lessons from landscape ecology. Second edition. New Haven: Yale University Press.

Bajema, R. A., T. L. DeVault, P. E. Scott, and S. L. Lima. 2001. Reclaimed coal mine grasslands and their significance for Henslow's Sparrows in the American Midwest. The Auk 118:422-431.

Bakker, J. P., and F. Berendse. 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. TREE 14:63-67.

Bakker, K. K., D. E. Naugle, and K. F. Higgins. 2002. Incorporating landscape attributes into models for migratory grassland bird conservation. Conservation Biology 6:1638-1646.

Benton, T. G., D. M. Bryant, L. Cole, and H. Q. P. Crick. 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. Journal of Applied Ecology 39:673-687.

Berg, A. 2002. Composition and diverstiy of bird communities in Swedish farmland-forest mosaic landscapes. British Trust for Ornithology 49:153-165.

Beissinger, S. B., J. M. Reed, J. M. Wunderle, Jr., S. K. Robinson, and D. M. Finch. 2000. Report of the AOU conservation committee on the Partners in Flight species prioritization plan. The Auk 117:549-561.

Brawn, J. D., S. K. Robinson, and F. R. Thompson, III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32:251-276.

Brennan, L. A., and W. P. Kuvlesky. 2005. North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1-13.

Briggs, J. N., A. K. Knapp, J. M. Blair, J. L. Heisler, G. A. Hoch, M. S. Lett, and J. K. McCarron. 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55:243-254.

Childers, E. L., T. L. Sharik, and C. S. Adkisson. 1986. Effects of loblolly pine plantations on songbird dynamics in the Virginia Piedmont. Journal of Wildlife Management 1986:3.

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 114:324-352.

Cooper, R. J. 1999. Partners in Flight bird conservation plan for the southern Piedmont (physiographic area 11). The Plains, VA: American Bird Conservancy.

Cowell, M. C. 1998. Historical change in vegetation and disturbance on the Georgia Piedmont. American Midland Naturalist 140:78-89.

Denevan, W. M. 1992. The pristine myth: the landscape of the Americas in 1492. Annals of the Association of American Geographers 82:369-385.

DeVault, T. L., P. E. Scott, R. A. Bajema, and S. L. Lima. 2002. Breeding bird communities of reclaimed coal-mine grasslands in the American midwest. Journal of Field Ornithology 73:268-275.

Flanders, A. A., W. P. Kuvlesky, D. C. Ruthven, R. E. Zaiglin, R. L. Bingham, T. E. Fulbright, F. Hernandez, and L. A. Brennan. 2006. Effects of invasive exotic grasses on South Texas rangeland breeding birds. The Auk 123:171-182.

Giuliano, W. M., and S. E. Daves. 2002. Avian response to warm-season grass use in pasture and hayfield management. Biological Conservation 106:1-9.

Gordon, C. E. 2000a. Fire and cattle grazing on wintering sparrows in Arizona grasslands. Journal of Range Management 53:384-389.

Gordon, C.E. 2000b. Movement patterns of wintering grassland sparrows in Arizona. The Auk 117:748-759.

Grand, J., and S. A. Cushman. 2003. A multi-scale analysis of species-environment relationships: breeding birds in a pitch pine-scrub oak (*Pinus rigida-Quercus ilicifolia*) community. Biological Conservation 112:307-317.

Granfors, D. A., K. E. Church, and L. M. Smith. 1996. Eastern Meadowlarks nesting in rangelands and Conservation Reserve Program fields in Kansas. Journal of Field Ornithology 67:222-235.

Gray, L. C. 1933. HIstory of agriculture in the southern United States to 1860. Volume II. Baltimore: Waverly Press, Inc.

Grzybowski, J. A. 1982. Population structure in grassland bird communities during winter. The Condor 84:137-152.

Grzybowski, J.A. 1983. Patterns of space use in grassland bird communities during winter. The Wilson Bulletin 95:591-602.

Harper, F., editor. 1998. The travels of William Bartram. Athens, GA: The University of Georgia Press.

Herkert, J. R. 1994. The effects of habitat fragmentation on midwestern grassland bird communities. Ecological Applications 4:461-471.

Hunter, W. C., D. A. Buehler, R. A. Canterbury, J. L. Confer, and P. B. Hamel. 2001a. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin 29:440-455.

Hunter, W. C., L. H. Peoples, and J. A. Collazo. 2001b. Partners in Flight conservation plan for the South Atlantic Coastal Plain. U.S. Fish and Wildlife Service.

Kempton, R. A., and L. R. Taylor. 1976. Models and statistics for species diversity. Nature 262:818-820.

Kindscher, K., and L. L. Tieszen. 1998. Floristic and soil organic matter changes after five and thirty-five years of native tallgrass prairie restoration. Restoration Ecology 6:181-196.

King, D. I., R. M. Degraaf, and C. R. Griffin. 2001. Productivity of early successional shrubland birds in clearcuts and groupcuts in an eastern deciduous forest. Journal of Wildlife Management 65:345-350.

Knick, S. T., and J. T. Rotenberry. 1995. Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. Conservation Biology 9:1059-1071.

Knick, S. T., and J. T. Rotenberry. 2000. Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. Ecology 8:220-227.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247-257.

Kobal, S. N., N. F. Payne, and D. R. Ludwig. 1999. Habitat/area relationships, abundance, and composition of bird communities in 3 grassland types. Transactions of the Illinois State Academy of Science 92:109-131.

Lorimer, C. G. 2001. Historical and ecological roles of disturbance in Eastern North American forests: 9,000 years of change. Wildlife Society Bulletin 29:425-439.

Marcus, J. F., W. E. Palmer, and P. T. Bromley. 2000. The effects of farm field borders on overwintering sparrow densities. Wilson Bulletin 112:517-523.

Martin, S. 1969. A technique for capturing nesting grassland birds with mist nets. Bird Banding 40:233-237.

Mayer, A. L., and G. N. Cameron. 2003. Landscape characteristics, spatial extent, and breeding bird diversity in Ohio, USA. Diversity and Distributions 9:297-311.

Morimoto, D. C., and F. E. Wasserman. 1991. Dispersion pattersn and habitat associations of rufous-sided towhees, common yellowthroats, and prairie warblers in the southeastern Massachusetts pine barres. The Auk 108:264-276.

Natural Resource Spatial Analysis Laboratory. 1998. 1998 land cover map of Georgia. Institute of Ecology, University of Georgia, Athens, GA.

Nudds, T. D. 1977. Quantifying the vegetation structure of wildlife cover. Wildlife Society Bulletin 5:113-117.

Nuttle, T., A. Leidolf, and L. W. Burger, Jr. 2003. Assessing conservation value of bird communities with Partners in Flight-based ranks. The Auk 120:541-549.

Peterjohn, B. G., and J. R. Sauer. 1999. Population status of North American grassland birds from the North American Breeding Bird Survey, 1966-1996. Studies in Avian Biology 19:27-44.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1998. Site fidelity of wintering Henslow's Sparrows. Journal of Field Ornithology 69:486-490.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1999. Habitat requirements of Henslow's Sparrows wintering in silvicultural lands of the Gulf Coastal Plain. The Auk 116:109-115.

Powell, R. A. 2000. Animal home ranges and territories and home range estimators. Pages 65-110 *in* L. F. Boitani and T.K. Fuller, editors. Research Techniques in Animal Ecology. New York: Columbia University Press.

Rasnake, M. 1992. Management of warm season bunch grasses. Proceedings of the Southern Pasture and Forage Crop Improvement Conference 48:38-39.

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

Robinson, R. A., J. D. Hart, J. M. Holland, and D. Parrott. 2004. Habitat use by seed-eating birds: a scale dependent approach. Ibis 146:87-98.

Root, T. 1988. Atlas of wintering North American birds: an analysis of Christmas Bird Count data. Chicago: The University of Chicago Press.

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

SAS Institute. 2005. SAS OnlineDoc version 9.1.3. Cary, NC: SAS Institute, Inc.

Skowno, A. L. and W. J. Bond. 2003. Bird community composition in an actively managed savanna reserve, importance of vegetation structure and vegetation composition. Biodiversity and Conservation 12:2279-2294.

Smith, M. D., P. J. Barbour, W. Burger, Jr., and S. J. Dinsmore. 2005. Density and diversity of overwintering birds in managed field borders in Mississippi. Wilson Bulletin 117:258-269.

ter Braak, C. J. F. 1986. Canonical Correspondence Analysis: A New Eigenvector Technique for Multivariate Direct Gradient Analysis. Ecology 67:1167-1179.

ter Braak, C. J. F., and P. Šmilauer. 2002 CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 4.5). Ithaca, NY: Microcomputer Power.

Titeux, N., M. Dufrene, J.-P. Jacob, M. Paquay, and P. Defourny. 2004. Multivariate analysis of a fine-scale breeding bird atlas using a geographical information system and partial canonical correspondence analysis: environmental and spatial effects. Journal of Biogeography 31:1841-1856.

Tucker, J. W., Jr., and W. D. Robinson. 2003. Influence of season and frequency of fire on Henslow's sparrows (Ammodramus Henslowii) wintering on Gulf Coast pitcher plant bogs. The Auk 120:96-106.

USDA Forest Service. 2004. Chattahoochee-Oconee National Forests land and resource management plan. U.S. Department of Agriculture, Forest Service, Southern Region. Report Management Bulletin R8-MB 113 A.

Vickery, P. D., and J. R. Herkert. 2001. Recent advances in grassland bird research: where do we go from here? The Auk 118:11-15.

Vickery, P. D., J. Herkert, R., F. L. Knopf, J. Ruth, and C. E. Keller. 2000. Grassland birds: an overview of threats and recommended management strategies. Pages 74–77 *in* R. Bonney, D. N. Pashley, R. J. Cooper, and L. Niles, editors. Strategies for bird conservation: the Partners in Flight planning process. Proceedings of the 3rd Partners in Flight workshop (1995). Proceedings RMRS-P-16. U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah.

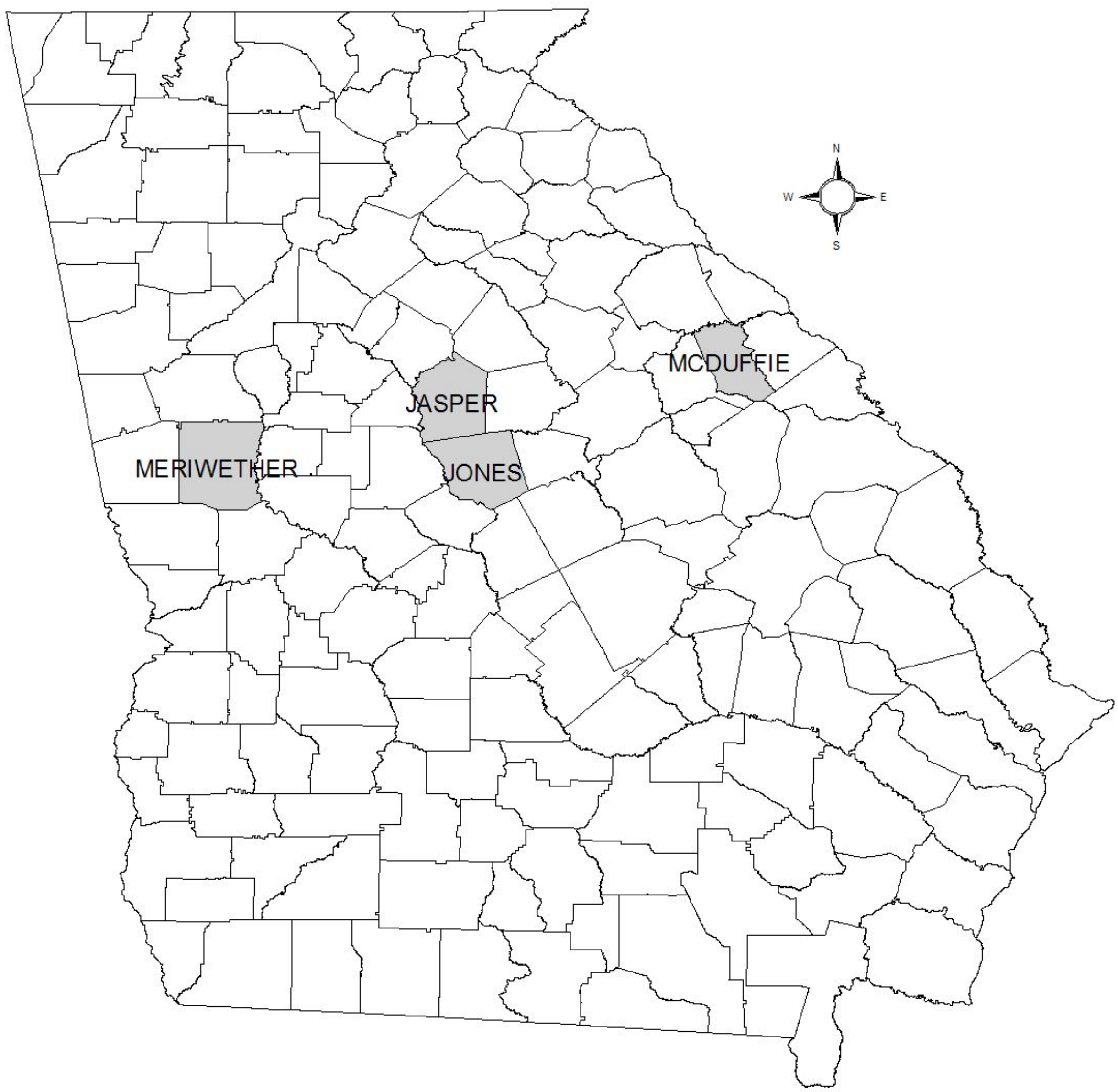
Vickery, P. D., P. L. Tubaro, J. M. Cardoso da Silva, B. G. Peterjohn, J. Herkert, R., and R. B. Cavalcanti. 1999. Conservation of grassland birds in the western hemisphere. Studies in Avian Biology 19:2-26.

Vitz, A. C., and A. D. Rodewald. 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127:477-486.

Vogl, R. J. 1973. Fire in the Southeastern grasslands. Proceedings of the Tall Timbers Fire Ecology Conference. 12:175-198.

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

Figure 4.1. Map of the state of Georgia showing the 4 counties where field work occurred in 2002-2004. Research investigated 7 land management prescriptions for managing early successional habitat at the transition between the Georgia Piedmont and Upper Coastal Plain ecological regions.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JONES

Figure 4.2. Canonical correspondence analysis (CCA) biplot of study plots and environmental variables for breeding bird community data collected using mist nets in central Georgia, USA. Arrows represent the 4 best environmental variables selected using Monte-Carlo forward selection; length of the arrow indicates the importance of the variable to the ordination. (OP=open planted, OUP=open unplanted, FP=forest opening planted, FUP=forest opening unplanted, LBAF=low basal area forest, TF=thinned forest, UTF=unthinned forest)

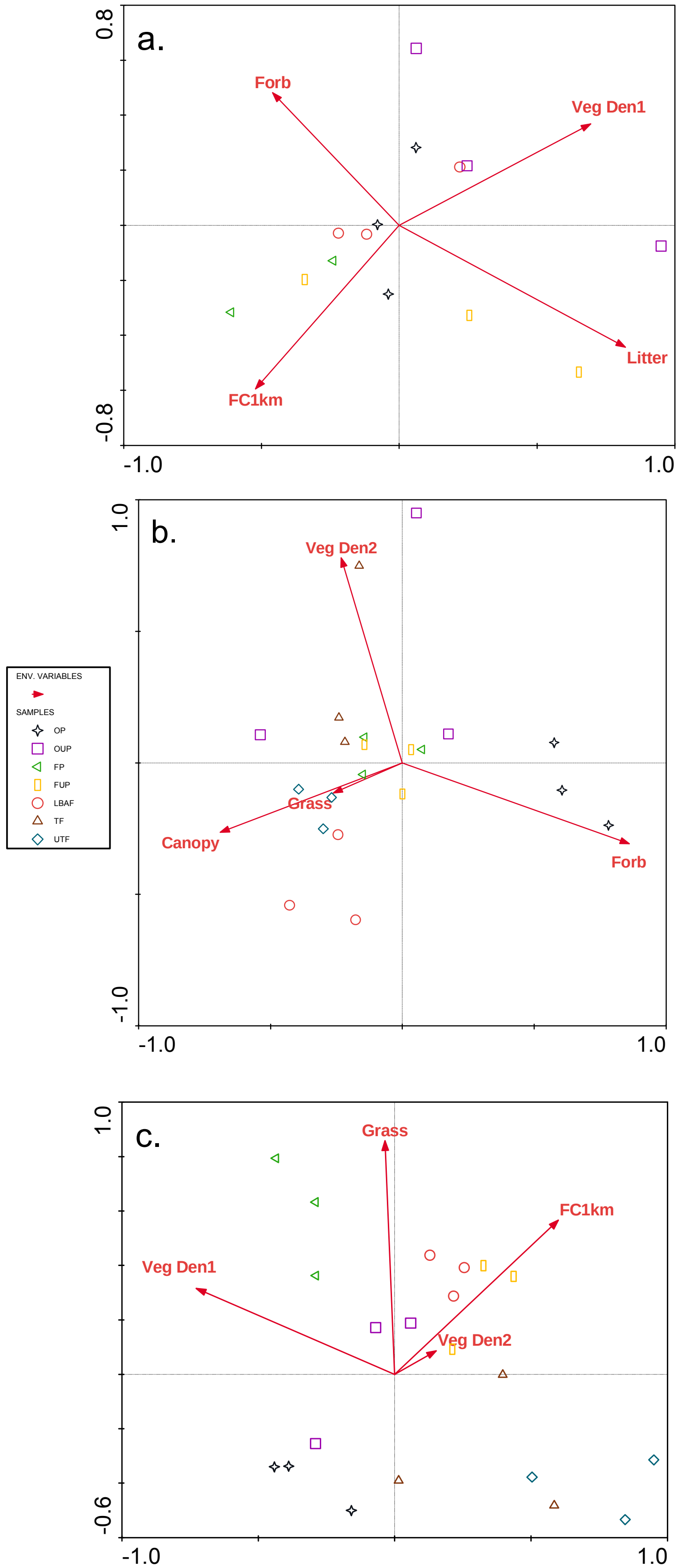
a.
0.8
-0.8
-1.0
1.0
Forb
Veg Den1
Litter
FC1km
b.
1.0
-1.0
-1.0
1.0
Veg Den2
Grass
Canopy
Forb
ENV. VARIABLES
SAMPLES
OP
OUP
FP
FUP
LBAF
TF
UTF
c.
1.0
-0.6
-1.0
1.0
Grass
FC1km
Veg Den1
Veg Den2

Figure 4.3. Canonical correspondence analysis (CCA) biplot of study plots and environmental variables for winter bird community data collected using mist nets in central Georgia, USA. Arrows represent the 4 best environmental variables selected using Monte-Carlo forward selection; length of the arrow indicates the importance of the variable to the ordination. (OP=open planted, OUP=open unplanted, FP=forest opening planted, FUP=forest opening unplanted, LBAF=low basal area forest, TF=thinned forest, UTF=unthinned forest)

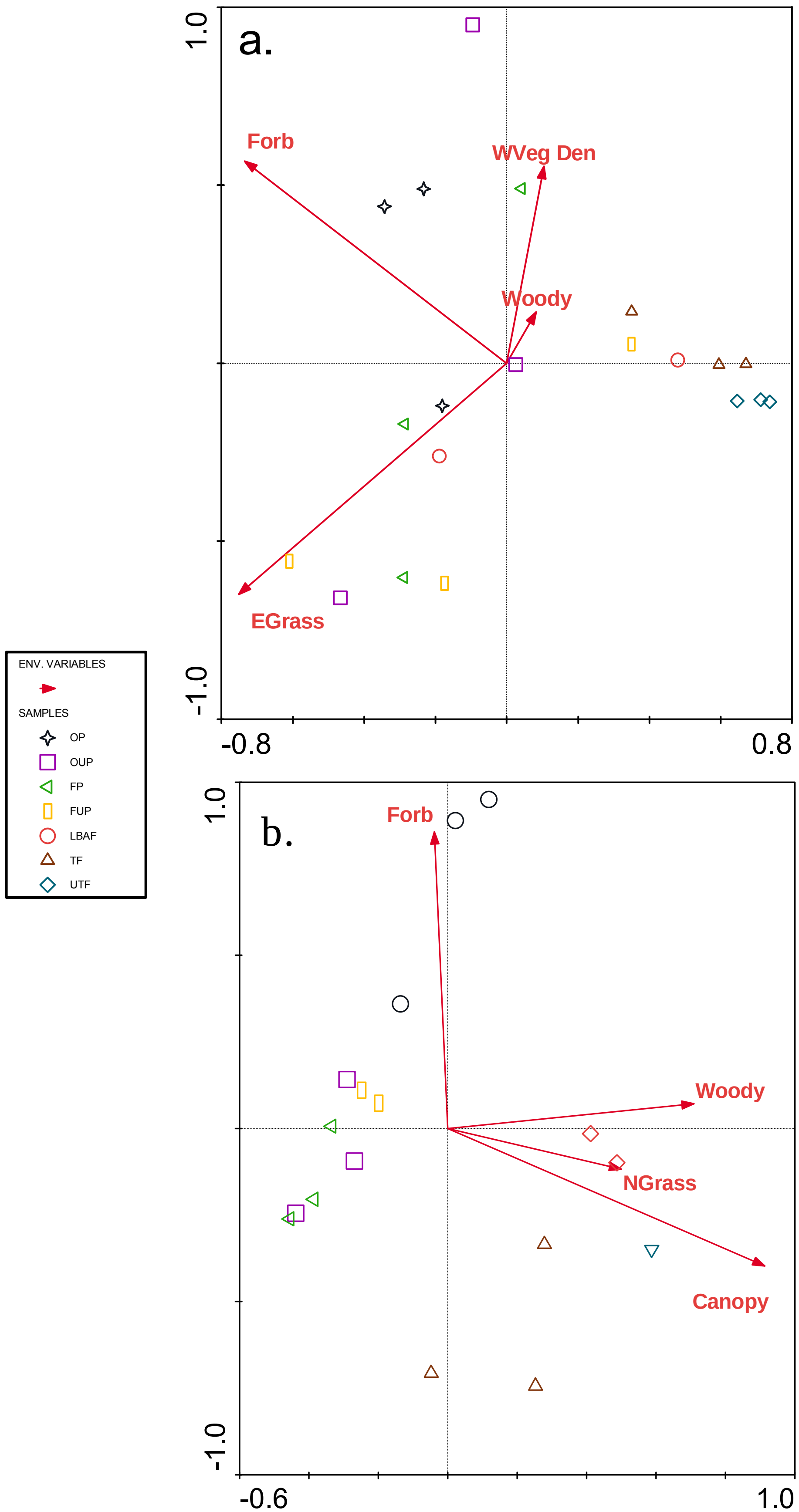
a.
Forb
WVeg Den
Woody
EGrass
1.0
-1.0
-0.8
0.8
ENV. VARIABLES
SAMPLES
OP
OUP
FP
FUP
LBAF
TF
UTF
b.
Forb
Woody
NGrass
Canopy
1.0
-1.0
-0.6
1.0

Table 4.1. Land management treatments investigated for their ability to provide habitat for disturbance-dependent birds during summer and winter in central Georgia, 2002-2004. Treatments were replicated 3 times at each location.

Treatment	Code	Description	Location
Open planted	OP	Old fields planted with mixture of native grasses in spring 2002	Joe Kurz WMA
Open unplanted	OUP	Old field habitat maintained with burning and mowing	Joe Kurz WMA
Forest opening planted	FP	Forest openings planted with native grasses in spring 2002	Piedmont NWR
Forest opening unplanted	FUP	Forest openings maintained with mowing and burning	Piedmont NWR
Low basal area forest	LBAF	Upland pine forest thinned 1980s, 9 m^2/ha basal area, 3-year fire rotation	Clark's Hill WMA
Thinned forest	TF	Upland pine thinned 1990s; maintained with fire	Oconee NF
Unthinned forest	UTF	Upland pine forest without recent thin; maintained with fire	Oconee NF

Table 4.2. Modified Braun-Blanquet scale for vegetation cover determination in grassland habitat.

Cover score	Grass cover
0	0%
1	0-5%
2	5-25%
3	25-50%
4	50-75%
5	75+

Table 4.3. Bird species captured during summer mist netting in central Georgia, 2002-2004 divided into habitat associations categories based on Hunter et al (2001). Partner in Flight (PIF) ranks are based on conservation priority (4=high priority).

Common name	Scientific name	Species code	Breeding PIF rank
Grass/herb:			
Bachman's sparrow	*Aimophila aestivalis*	BACS	4
Eastern meadowlark	*Sturnella magna*	EAME	3
Northern bobwhite	*Colinus virginianus*	NOBO	4
Savannah sparrow*	*Passerculus sandwichensis*	SAVS	
Shrub/scrub:			
Blue grosbeak	*Passerina caerulea*	BLGR	3
Common yellowthroat	*Geothlypis trichas*	COYE	2
Field sparrow	*Spizella pusilla*	FISP	3
Mourning dove	*Zenaida macroura*	MODO	2
Orchard oriole	*Icterus spurius*	OROR	3
Palm warbler*	*Dendroica palmarum*	WPWA	
Prairie warbler	*Dendroica discolor*	PRAW	3
Swamp sparrow*	*Melospiza georgiana*	SWSP	
Veery*	*Catharus fuscescens*	VEER	
Whip-poor-will	*Caprimulgus vociferus*	WPWI	4
Yellow warbler	*Dendroica petechia*	YWAR	2

(continued)

Table 4.3 (continued). Bird species captured during summer mist netting in central Georgia, 2002-2004 divided into habitat associations categories based on Hunter et al (2001). Partner in Flight (PIF) ranks are based on conservation priority (4=high priority).

Common name	Scientific name	Species code	Breeding PIF rank
Yellow-breasted chat	*Icteria virens*	YBCH	3
Disturbed woodland:			
American goldfinch	*Carduelis tristis*	AMGO	2
Eastern bluebird	*Sialia sialias*	EABL	2
Eastern kingbird	*Tyrannus tyrannus*	EAKI	3
Eastern wood-pewee	*Contopus virens*	EAWP	3
Summer tanager	*Piranga rubra*	SUTA	2
Forest disturbance:			
Black-and-white warbler	*Mniotilta varia*	BAWW	2
Blue-gray gnatcatcher	*Polioptila caerulea*	BGGN	4
Eastern towhee	*Pipilo erthrophthalmus*	EATO	3
Gray catbird*	*Dumetella carolinensis*	GRCA	
Hermit thrush*	*Catharus guttatus*	HETH	
Hooded warbler	*Wilsonia citrina*	HOWA	3
Indigo bunting	*Passerina cyanea*	INBU	2
Kentucky warbler	*Oporornis formosus*	KEWA	3
White-eyed vireo	*Vireo griseus*	WEVI	3
Wood thrush	*Hylocichla mustelina*	WOTH	3

(continued)

Table 4.3 (continued). Bird species captured during summer mist netting in central Georgia, 2002-2004 divided into habitat associations categories based on Hunter et al (2001). Partner in Flight (PIF) ranks are based on conservation priority (4=high priority).

Common Name	Scientific name	Species code	Breeding PIF rank
Other:			
Acadian flycatcher	*Empidonax virescens*	ACFL	3
Blue jay	*Cyanocitta cristata*	BLJA	2
Carolina wren	*Thryothorus ludovicianus*	CARW	2
Chipping sparrow	*Spizella passerina*	CHSP	2
Downy woodpecker	*Picoides pubescens*	DOWO	2
Eastern phoebe	*Sayornis phoebe*	EAPH	2
Great-crested flycatcher	*Myiarchus crinitus*	GCFL	3
Hairy woodpecker	*Picoides villosus*	HAWO	3
Northern cardinal	*Cardinalis cardinalis*	NOCA	2
Ovenbird	*Seiurus aurocapillus*	OVEN	1
Pine warbler	*Dendroica pinus*	PIWA	3
Red-eyed vireo	*Vireo olivaceus*	REVI	2
Ruby-throated hummingbird	*Archilochus colubris*	RTHU	3
Tufted titmouse	*Baeolophus bicolor*	ETTI	2
Yellow-rumped warbler*	*Dendroica coronata*	MYWA	
Yellow-throated vireo	*Vireo flavifrons*	YTVI	3
Yellow-throated warbler	*Dendroica dominica*	YTWA	3

*Indicates a winter resident with a single capture early in the breeding season. These species were excluded from calculations of TCV

Table 4.4. Avian community metrics across seven land management treatments for maintaining early successional/disturbance dependent habitat in central Georgia April-July 2002-2004 (n=9 for all treatments except TF and UTF; n=6 for TF and UTF). Different letters across a row indicate a significant difference (*P* ≤ 0.05) for Duncan's multiple range test.

Community metric	Treatment						
	OP	OUP	FP	FUP	LBAF	TF	UTF
Indices							
Richness	4.7 ± 2.0	4.3 ± 3.2	4.2 ± 2.9	3.9 ± 2.3	4.2 ± 3.2	5.0 ± 2.9	4.7 ± 1.2
Diversity	4.0 ± 1.5	3.4 ± 2.1	3.4 ± 2.0	3.7 ± 2.1	4.2 ± 2.8	6.7 ± 3.3	4.6 ± 1.5
TCV	24.1 ± 14.6	31.7 ± 30.5	21.2 ± 16.3	13.9 ± 10.4	28.9 ± 16.2	26.2 ± 15.4	17.8 ± 9.8
Abundance (birds/100 net hours)							
All birds	11.72 ± 5.03	14.36 ± 10.66	9.72 ± 8.06	6.62 ± 4.35	11.03 ± 6.68	9.32 ± 4.59	6.51 ± 3.28
Grass-herb birds	0.00 ± 0.00[b]	0.49 ± 1.15[b]	0.39 ± 0.47[b]	0.10 ± 0.31[b]	3.25 ± 3.24[a]	0.00 ± 0.00[b]	0.00 ± 0.00[b]
Shrub-scrub birds	5.83 ± 4.99[a]	7.29 ± 6.86[a]	1.89 ± 1.57[b]	1.91 ± 2.02[b]	1.43 ± 1.99[b]	1.70 ± 1.44[b]	1.02 ± 0.63[b]
Disturbed-woodland birds	1.36 ± 2.32	0.60 ± 1.02	0.36 ± 0.56	1.03 ± 1.59	0.53 ± 0.97	0.46 ± 0.50	0.44 ± 0.72
Forest-disturbance birds	3.58 ± 1.88[a]	3.93 ± 3.75[a]	2.74 ± 3.21[ab]	1.17 ± 1.24[ab]	2.03 ± 2.05[ab]	3.13 ± 2.36[a]	0.31 ± 0.48[b]

Table 4.5. Pearson correlation coefficeints (r) of grass cover (within study plot) and forest cover within 500m and 1km buffers around study plot with avian community metrics for study plots in central Gerogia, USA for April-July 2002-2004 (n=57 for each comparison). TCV (total conservation value) is defined within the text.

Community metric	Grass cover	Forest cover (500m)	Forest cover (1km)
Abundance			
All birds	-0.07	0.20	-0.21
Grass/Herb birds	0.25[a]	0.12	0.06
Shrub/Scrub birds	-0.05	0.15	-0.42[c]
Disturbed woodland birds	0.07	-0.14	-0.15
Forest disturbance birds	-0.22	0.15	-0.31[a]
Indices			
Richness	-0.01	-0.005	-0.005
Diversity	-0.06	-0.01	0.08
TCV	-0.07	0.09	0.31

[a]$P \leq 0.05$,[b]$P \leq 0.01$,[c]$P \leq 0.001$

Table 4.6. P-values from Monte-Carlo forward selection permutation test of envrionmental variables used in canonical correspondence analysis (CCA) of breeding avian community data from seven land management strategies for disturbance dependent birds in central Georgia (2002-2004). Significant p-values are in **bold** indicating that variable contributes significantly to the model.

Code	Definition	2002	2003	2004
Grass	Average grass cover (%) in plot		**0.02**	0.08
EGrass	Average exotic grass cover (%) in plot			
NGrass	Average native grass cover (%) in plot			
Forb	Average forb cover (%) in plot	0.35	**0.001**	
Woody-Shrub	Average shrub cover (%) in plot			
Bare	Average amount of bare ground (%)			
Canopy	Canopy cover (%)		**0.02**	
Litter	Average litter depth (cm)	0.06		
Veg Den2	Vegetation density (%) 0.5-1.0m above ground		0.06	**0.02**
Veg Den1	Vegetation density (%) 0.0-0.5m above ground	0.18		**0.01**
FC500m	Forest cover in 500-m buffer around plot			
FC1km	Forest cover in 1-km buffer around plot	0.24		**0.05**

Table 4.7. Bird species captured during winter mist netting in central Georgia, 2003-2004 divided into habitat associations categories based on Hunter et al (2001). Partner in Flight (PIF) ranks are based on conservation priority (4=high priority).

Common name	Scientific name	Species code	Wintering PIF rank
Grass/herb:			
Savannah sparrow	*Passerculus sandwichensis*	SAVS	2
Vesper sparrow	*Pooecetes gramineus*	VESP	3
Shrub/scrub:			
Field sparrow	*Spizella pusilla*	FISP	3
Palm warbler	*Dendroica palmarum*	WPWA	4
Swamp sparrow	*Melospiza georgiana*	SWSP	2
Disturbed woodland:			
American goldfinch	*Carduelis tristis*	AMGO	2
Eastern bluebird	*Sialia sialias*	EABL	2
Northern flicker	*Colaptes auratus*	YSFL	2
Yellow-bellied sapsucker	*Sphyrapicus varius*	YBSA	2
Forest disturbance:			
Brown thrasher*	*Toxostoma rufum*	BRTH	3
Dark-eyed junco	*Junco hyemalis*	UDEJ	2
Eastern towhee*	*Pipilo erthrophthalmus*	EATO	3
Fox sparrow	*Passerella iliaca*	FOSP	2
Hermit thrush	*Catharus guttatus*	HETH	2
White-throated sparrow	*Zonotrichia albicollis*	WTSP	2

(continued)

Table 4.7. (continued) Bird species captured during winter mist netting in central Georgia, 2003-2004 divided into habitat associations categories based on Hunter et al (2001). Partner in Flight (PIF) ranks are based on conservation priority (4=high priority).

Common name	Scientific name	Species code	Wintering PIF rank
Other:			
American kestrel*	*Falco sparverius*	AMKE	3
American robin*	*Turdus migratorius*	AMRO	2
Blue-headed vireo	*Vireo solitarius*	BHVI	3
Carolina chickadee*	*Poecile carolinensis*	CACH	3
Carolina wren*	*Thryothorus ludovicianus*	CARW	2
Chipping sparrow*	*Spizella passerina*	CHSP	2
Downy woodpecker*	*Picoides pubescens*	DOWO	3
Eastern phoebe*	*Sayornis phoebe*	EAPH	2
Golden-crowned kinglet	*Regulus satrapa*	GCKI	2
Northern cardinal*	*Cardinalis cardinalis*	NOCA	2
Northern mockingbird*	*Mimus polyglottos*	NOMO	2
Pine warbler*	*Dendroica pinus*	PIWA	3
Red-bellied woodpecker*	*Melanerpes carolinus*	RBWO	3
Ruby-crowned kinglet	*Regulus calendula*	RCKI	2
Song sparrow	*Melospiza melodia*	SOSP	2
Tufted titmouse*	*Baeolophus bicolor*	ETTI	2
Yellow-rumped warbler	*Dendroica coronata*	MYWA	2

*Indicates a species that also breeds in the study sites.

Table 4.8. Avian community metrics across seven land management treatments for maintaining early successional/disturbance dependent habitat in central Georgia January-March 2003-2004 (n=6 for each treatment). Different letters across a row indicate a significant difference (*P* ≤ 0.05) for Duncan's multiple range test.

Community metric	Treatment						
	OP	OUP	FP	FUP	LBAF	TF	UTF
Abundance (birds/100 net hours)							
All birds	23.16 ± 19.91	13.52 ± 14.82	19.26 ± 18.23	3.13 ± 2.04	14.71 ± 20.81	13.54 ± 10.03	12.09 ± 8.57
Grass-herb birds	2.69 ± 3.14^{a}	1.14 ± 2.16^{ab}	1.07 ± 2.62^{ab}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}
Shrub-scrub birds	8.25 ± 7.91^{a}	2.92 ± 4.58^{b}	0.17 ± 0.41^{c}	0.00 ± 0.00^{c}	0.00 ± 0.00^{c}	0.23 ± 0.57^{c}	0.17 ± 0.41^{c}
Disturbed-woodland birds	1.98 ± 3.19	0.67 ± 0.75	0.78 ± 1.91	0.65 ± 1.01	1.97 ± 4.83	2.58 ± 2.45	0.00 ± 0.00
Forest-disturbance birds	0.57 ± 0.65^{bc}	0.00 ± 0.00^{c}	0.34 ± 0.83^{bc}	0.47 ± 0.76^{bc}	0.00 ± 0.00^{c}	2.14 ± 1.98^{a}	1.77 ± 2.54^{ab}
Indices							
Richness	5.00 ± 2.89^{a}	4.33 ± 3.38^{a}	3.50 ± 1.52^{a}	2.00 ± 1.26^{ab}	1.17 ± 1.17^{b}	5.67 ± 4.22^{a}	2.5 ± 2.35^{ab}
Diversity	3.75 ± 2.12	3.73 ± 2.50	2.76 ± 1.42	2.15 ± 0.96	1.29 ± 0.64	4.62 ± 4.23	2.51 ± 1.66
TCV	41.00 ± 32.43	22.83 ± 23.81	27.67 ± 20.43	4.33 ± 2.73	28.17 ± 42.64	18.50 ± 18.04	11.83 ± 14.03

Table 4.9. Pearson correlation coefficeints (r) of grass cover (within study plot) and forest cover within 500m and 1km buffers around the plot with avian community metrics for study plots in central Gerogia, USA for Jan-March 2003-2004 (n=42 for each comparison). TCV (total conservation value) is defined within the text.

Community metric	Grass cover	Forest cover (500m)	Forest cover (1km)
Abundance			
All birds	-0.17	0.20	-0.17
Grass-herb birds	-0.22	0.05	-0.42[b]
Shrub-scrub birds	-0.18	0.10	-0.55[c]
Disturbed-woodland birds	-0.17	0.04	-0.09
Forest-disturbance birds	-0.39[b]	0.15	0.28
Indices			
Richness	-0.36[a]	0.22	-0.05
Diversity	-0.32[a]	0.18	0.002
TCV	-0.11	0.16	-0.15

[a]$P \leq 0.05$,[b]$P \leq 0.01$,[c]$P \leq 0.001$

Table 4.10. P-values from Monte-Carlo forward selection permutation test of envrionmental variables used in canonical correspondence analysis (CCA) of winter avian community data from seven land management strategies for disturbance dependent birds in central Georgia (2003-2004). Significant p-values are in **bold** indicating that variable contributes significantly to the model.

Code	Definition	2003	2004
EGrass	Average exotic grass cover (%) in plot	**0.001**	
NGrass	Average native grass cover (%) in plot		0.57
Forb	Average forb cover (%) in plot	**0.001**	0.21
Woody/Shrub	Average shrub cover (%) in plot	**0.04**	0.40
Bare	Average amount of bare ground (%)		
Canopy	Canopy cover (%)		**0.01**
SLitter	Average litter depth (cm) during the previous summer		
WVegDen	Vegetation density 0.0-0.5m above ground during winter	0.07	
SVegDen	Vegetation density (%) 0.0-0.5m above ground during the previous summer		
FC500m	Forest cover in 500-m buffer around plot		
FC1km	Forest cover in 1-km buffer around plot		

CHAPTER 5

CONCLUSIONS

STATUS OF EARLY SUCCESSIONAL BIRDS

Early successional birds in the Southeastern United States have received little attention in the literature (Plentovich et al. 1998, 1999; Marcus et al. 2000). Priorities for grassland bird research include multiple-species management, replacement of exotic grasses with native grasses, and wintering ecology (Brennan and Kuvlesky 2005). The purpose of our research was to fill in some of the gaps in current knowledge about early successional bird species in the Southeastern United States. We used community level analyses to look at bird response to current management alternatives in the Southeast as well as a novel technique: native grass restoration. We examined the impact of these management alternatives on both breeding and wintering birds. Finally, we looked at differences in arthropod resources during the breeding season in the different management alternatives.

We hypothesized that native warm season grasses (NWSG) could be established successfully in old field habitat consisting primarily of exotic pasture grasses. Further, we believed that restored habitat would differ in vegetative community indices and structure from old field habitat. We hypothesized that the differences in vegetation community and structure in restored grasslands would positively impact the bird community by 1) supporting a more diverse community, 2) supporting more birds, and 3) providing more conservation benefit.

Eastern early successional birds exist across a spectrum ranging from open grasslands to gaps in mature forests (Hunter et al. 2001a). To address the impact of a tree component of early successional habitat on the avian community, we looked at sites ranging from open fields to the grass understory of a pine forest. Contrary to research in the Midwest, we hypothesized that abundance of breeding populations of southeastern early successional birds would not be negatively impacted by forest cover in the landscape. We hypothesized short-distance migrants that winter in the Southeast would show a negative relationship with forest cover.

Because breeding birds are heavily dependent on arthropod food, we examined arthropod community responses to the same management practices. We hypothesized that arthropod richness and abundance would be greatest in treatments with the highest native grass and forb cover. We also hypothesized that fire ant (*Solenopsis invicta*) abundance would negatively impact the richness and abundance of other arthropods.

NATIVE GRASS RESTORATION

The Partners in Flight migratory bird program set habitat objectives for the South Atlantic Coastal Plain region that include restoring or converting cool-season grass pastures to native warm-season grasslands on more than 4 million ha (Hunter et al. 2001b). Eastern grasslands are fire dependent systems and most management has focused on re-introducing fire. Reintroduction of the necessary disturbance may not be enough to restore native vegetation if the local seed bank is impoverished and seed dispersal is limited by a fragmented landscape (Kindscher and Tieszen 1998, Bakker and Berendse 1999). We restored native warm season grasses on approximately 20 ha of old field habitat in central Georgia. We found that native warm season grasses can be successfully restored in old field habitat in central Georgia. The low species richness of grasses in planted plots in 2002 indicated that initial site preparation was effective in suppressing exotic pasture grasses. After the first growing season, planted plots were characterized by sparse native grass cover and low vegetation density.

The structure of grasslands can have an impact on avian occupancy (Pons et al. 2003), reproductive success (Dale et al. 1997, Shochat et al. 2005), and food resource availability (Bock et al. 1996, McCraken et al. 2004). As we hypothesized, control and planted plots differed in vegetative structure. Planted plots had tall, distinct bunches of grass. Grass in control plots was low to the ground and dense, forming a thick, ground-level mat of vegetation.

The extensive forest cover within the landscape of our study areas poses management challenges for maintaining early successional habitat. Increasing woody plant cover in grasslands decreases the suitability of habitat for grassland birds (O'Leary and Nyberg 2000, Coppedge et al. 2001). Although planted and control plots had a similar number of woody species throughout the study, the preparation required to establish native grasses effectively reduced woody vegetation in planted plots. We expect that the lack of shrubs and other woody vegetation in planted plots make these areas especially attractive to grassland associated birds, as was found in southern pine parklands, where reducing woody ground cover was beneficial to grassland birds of concern (Conner et al. 2002).

Even though field size was not a predictor of bird density in other studies of grassland birds (Ribic and Sample 2001), our study plots (1.2-4.0 ha) may have been too small to support large populations of grassland birds masking any subtle differences between treatments. Why were differences not detected in Georgia? There are several possible explanations, but I suspect that the composition of the breeding bird community in Georgia is partially responsible. Many of the grassland obligate birds that winter in Georgia head north to breeding grounds in the Midwest, the northern U.S., and southern Canada, leaving far fewer grassland obligate bird species in the Southeast during the spring and summer (Rising and Beadle 1996, Sibley 2000). Except for the Bachman's sparrow (Hammerson 1994), Georgia is not the core breeding habitat for any grassland bird, but does support breeding populations of grasshopper sparrow, eastern kingbird, northern bobwhite, and eastern meadowlark.

WINTER BIRD REPONSE

The Southeast provides both breeding and wintering habitat to early successional birds. Many early successional birds are short distance migrants that winter in the southern United States and northern Mexico (Root 1988). Winter is an often neglected part of the annual cycle of birds. Little research has focused on wintering grassland birds within the United States (Grzybowski 1982, 1983; Gordon 2000a, 2000b; Tucker and Robinson 2003). In the Southeast, wintering ecology of the declining Henslow's sparrow has received some attention (Plentovich et al. 1998, 1999; Tucker and Robinson 2003), but large gaps still exist in our knowledge of wintering ecology.

Research on winter ecology and habitat use are research priorities for grassland bird conservation (Vickery and Herkert 2001). Our research contributes to the small, but growing body of knowledge about wintering grassland birds. Unlike during the breeding season, NWSG restoration positively impacted the wintering birds in Georgia.

In 2003, we found the highest density of wintering birds and sparrows in the study plots we planted with NWSGs. Because the planted study plots had only been planted the previous spring, these plots had low vertical vegetation density that may have been attractive to wintering sparrows. Wintering sparrows may be attracted to recent disturbances. The highest density of over-wintering sparrows in Southern Texas was found in shrub-grasslands (Igl and Ballard 1999). In Oregon, sparrow diversity was highest in areas with low vegetation complexity and recent disturbance (Patterson 2002). Vesper (*Pooecetes gramineus*) and savannah sparrows (*Passerculus sandwichensis*), which were commonly observed in our plots, responded positively to fire disturbance in Arizona (Gordon 2000a).

Birds can use their mobility to track fluctuating resources and distribute themselves according to resource abundance (Hurlbert and Haskell 2003). In our study, wintering sparrows may have been attracted to planted plots due to increased seed availability. Wintering grassland birds select sites that have high seed density (Robinson et al. 2004). Annual plants dominate restored prairie areas for the first 5 years (Kindscher and Tieszen 1998) producing large quantities of seed. Fields enrolled in the conservation reserve program (CRP) with a similar species composition to our planted fields, had significantly more seed availability than nearby pastures (Klute et al. 1997). In addition to being more abundant, seeds may have been easier to acquire in planted fields because the low vegetation density during 2003 left large areas of bare earth. Ten of 14 bird species wintering in agricultural grasslands in England were associated with bare earth suggesting that avian foraging opportunities were better in those fields (Perkins et al. 2000).

Wintering bird abundance in our study was highly variable from year to year. Abundance in 2004 was a quarter of what it had been in 2003. The distribution and abundance of wintering sparrows in Southeastern Arizona also varied significantly between years across the region (Gordon 2000a, 2000b). Wintering sparrows are short distance migrants (Root 1988) moving only as far as necessary to get the resources needed. Wintering bird distribution is driven by weather in Scotland (French and Picozzi 2002). The second winter of this study was very mild across large parts of the United States. Birds may not have had to move as far south to find food resources and escape inclement weather. The high annual variability we detected indicates that long-term studies of wintering bird communities are probably required to fully understand winter bird ecology and habitat use.

ARTHROPOD RESPONSE

Invertebrate abundance and diversity should be considered a top priority in avian management (Robel et al. 1995). Arthropods are a key component of avian diets during the spring and summer. Both chicks and older birds rely on this protein rich food source to fuel energetic demands of rapid growth and reproduction, respectively. If food resources are limited, adult birds may face a trade-off between annual survival and reproductive success (Martin 1995). Availability of arthropods to provision nestlings may directly limit nestling survival (Moreby 2004).

Different management strategies could have varying impacts on arthropod abundance, influencing the availability, abundance, and size of arthropods (McCraken et al. 2004). Our results supported our hypothesis that arthropod richness and abundance would be greatest in treatments with high grass and forb cover. Plots with less canopy cover and high forb and grass cover had high arthropod abundance. Forb cover was an important determinant of the arthropod community. Diversity and richness of arthropods in Kansas were positively correlated to plant diversity and richness (Jonas et al. 2002). Abundance and diversity of invertebrates on British farmland declined on farms with management practices that decreased the number and diversity of forbs (Atkinson et al. 2004).

Canopy cover was also an influential habitat variable in 2004. Treatments with high canopy cover had low abundance and diversity of arthropods. High canopy cover probably reduced light penetration and development of a forb and grass rich understory. The current forest management plan for Oconee National Forest, one of our study sites, calls for thinning upland pine and reducing the hardwood understory (USDA Forest Service 2004). Our data suggest that implementation of the management plan likely will increase the abundance of understory arthropods.

Fire ants (*Solenopsis invicta*) have been shown to negatively impact invertebrates through direct mortality and competition for food and habitat (Wojcik et al. 2001). In central Texas, areas infested with fire ants had 40% fewer arthropod species and 75% fewer individuals than areas without fire ants (Porter and Savignano 1990). Contrary to our hypothesis, fire ant abundance was positively correlated with arthropod abundance. The positive correlation between arthropod abundance and fire ants detected in our study may have more to do with landscape level factors than an actual positive impact of fire ants on arthropods. Several studies have found a negative impact of shaded conditions on fire ant abundance. In Florida, fire ants were less abundant in shaded conditions (Porter and Tschinkel 1987). Fire ant density in South

Carolina was positively related to open canopy and disturbance suggesting that direct sunlight may be a requirement for fire ant colonization (Stiles and Jones 1998). In our study, plots with low fire ant and low arthropod abundance were shaded. Low arthropod abundance was likely a result of shading and not the absence of fire ants. Shading likely reduced the cover of forbs and grass in the understory, thereby reducing the numbers of arthropods sampled by sweep netting.

Orthopterans, lepidopterans, and araneae are important food resources for breeding birds (McIntyre and Thompson 2003) and are thus an important consideration for land management decisions. Orthopteran abundance was reduced in both of our treatments maintained with annual mowing. Lepidopteran abundance and richness did not differ among treatments likely because sweep netting is not the best method for sampling larval or adult lepidopterans. Further work using other arthropod sampling techniques would need to be completed to determine the response of lepidopterons to management treatments such as those in our study. Spiders were abundant in all but the most dense forest treatment in 2003 and were most abundant in the open woodland treatment in 2004. The presence of trees in early successional habitat did not seem to negatively impact spider abundance.

Arthropod communities can reflect differences in landscape level factors as well as plant communities (Jonas et al. 2002). Our CCA data suggested that both local (forb cover, grass cover, litter depth) and landscape level features (canopy cover, forest cover) were influencing the structure of arthropod communities in our study. Arthropod abundance was highest in our treatments planted with native grasses and lowest in treatments with the highest landscape level forest cover. A recent study of Texas rangelands found higher arthropod abundance in areas with native grasses (Flanders et al. 2006). Native prairie had the highest arthropod abundance when compared to native and non-native conservation reserve program (CRP) fields (McIntyre and Thompson 2003). Managing habitat for native grass and forb cover appears to increase avian insect availability.

AVIAN COMMUNITY RESPONSE

In our study, forest cover only ranged from 70-95%, but seemed to have an impact on the bird community contrary to what we had hypothosized. Forest cover within 500 m of our plots was not correlated with any breeding season or winter community metric, but forest cover with a 1 km buffer was negatively correlated with the abundance of shrub-scrub birds during the summer and winter. Shrub-scrub birds in Idaho also responded negatively to forest cover in the landscape during the summer (Knick and Rotenberry 2000).

Grass-herb bird abundance was positively correlated to grass cover within the study plot and showed no relationship with forest cover at either scale (500 m or 1 km) during the summer, but was negatively associated with forest cover during the winter. Our data suggest that the local characteristics of the understory may be more important than the surrounding landscape during the breeding season, but landscape characteristics may come into play during the winter. This change in driving environmental variables between seasons probably reflects changes in the bird community composition. The Bachman's sparrow (*Aimphila aestivalis*) was the most common breeding grass-herb bird in our study, but the grass-herb community was dominated by savannah and vesper sparrows during the winter.

During both summer and winter, open woodlands had one of the highest total conservation values (TCVs), but the open treatments also had high conservation value in the summer and winter. The differences in TCV were not significant in either season suggesting that all treatments are contributing conservation value to the system. Accommodating the habitat needs of all disturbance-dependent species in the eastern United States will require a mosaic of successional stages in the landscape. Managing for early successional habitat does not have to be a trade-off between mature forest birds and disturbance-dependent birds. Patches of early successional habitat within a forested landscape may actually benefit forest birds, especially immediately following breeding, and early successional species. Post breeding juvenile and adult mature forest species used regenerating clear cuts in Ohio (Vitz and Rodewald 2006) and patches of early successional habitat in Virginia (Childers et al. 1986). Shrub-scrub birds occurred in both group cuts and clear cuts with equal nesting success (King et al. 2001). Forest management might be able to benefit both mature forest birds and early successional species.

MANAGEMENT CONCLUSIONS

Except for restoration of native grasses, most of the land management strategies investigated in this study have been used for extended periods of time in the Southeast. Only recently has there been an interest in restoring native grass species to benefit early successional bird species. Restoration is increasingly important in regions with scarce quantities of quality habitat (Vickery et al. 1999) or where habitat change can not be remedied with less intense management alternatives (Briggs et al. 2005). Native grass restoration did not stand out as being superior to all other treatments during the breeding season, but NWSG study plots did support substantial numbers of shrub-scrub and forest disturbance birds. In the winter, open planted study plots had large numbers of grass-herb and shrub-scrub birds. Native warm season grass restoration has the potential to benefit disturbance-dependent species and should be added to the land manger’s toolbox.

Habitat preferences of individual species influence the spatial structure of grassland communities (Mikami and Kawata 2002, Pons et al. 2003). Grassland birds select different types and ages of grasslands. Eastern meadowlarks require older grasslands without recent severe disturbance (Wiens 1969). Grasshopper sparrows prefer sparse vegetation with bare earth patches (Arguedas-Negrini 2001). No single management prescription will provide habitat for both of these species or for all declining grassland-associated species. Landscape-level management of grassland resources is recommended for providing habitat for the variety of declining species.

A mixture of grassland types is desirable in the landscape. The eastern meadowlark and grasshopper sparrow are two species of concern in Georgia, but they have very different habitat requirements. Grasshopper sparrows prefer light to intermediate litter layer showing up 1-2 years after a burn while eastern meadowlarks are found in areas with heavy litter and no recent disturbance (Granfors et al. 1996, Kobal et al. 1999). Modification to farmland may allow us to incorporate beneficial early successional habitat into the eastern United States. Increasing heterogeneity in the farmland landscape through edges, fallow areas, and different types of fields can make the landscape more hospitable to breeding (Benton et al. 2002) and wintering (Smith et al. 2005) early successional birds.

Land managers need to provide vegetative communities that are currently scarce (Wood et al. 2004). In the forest dominated landscape of central Georgia, early successional habitat is in short supply. The remaining habitat should be managed to maximize utility for breeding and wintering grassland birds. Our study determined that native grass restoration provided benefits to both breeding and wintering communities of grassland birds.

LITERATURE CITED

Arguedas-Negrini, N. 2001. Distribution, habitat and behavior of grasshopper sparrows, *Ammodramus savannarum* (Passeriformes: Emberizidae) in northwestern Nicaragua. Revista de Biologia Tropical 49:703-708.

Atkinson, P.W., D. Buckingham, and A.J. Morris. 2004. What factors determine where invertebrate-feeding birds forage in dry agricultural grasslands? Ibis 146:99-107.

Bakker, J. P. and F. Berendse. 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. Tree 14:63-67.

Benton, T. G., D. M. Bryant, L. Cole, and H. Q. P. Crick. 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. Journal of Applied Ecology 39:673-687.

Brennan, L. A., and W. P. Kuvlesky. 2005. North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1-13.

Briggs, J. N., A. K. Knapp, J. M. Blair, J. L. Heisler, G. A. Hoch, M. S. Lett, and J. K. McCarron. 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55:243-254.

Childers, E. L., T. L. Sharik, and C. S. Adkisson. 1986. Effects of loblolly pine plantations on songbird dynamics in the Virginia Piedmont. Journal of Wildlife Management 50:406-413.

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 114:324-352.

Coppedge, B. R., D. Engle, M., S. D. Fuhlendorf, R. E. Masters, and M. S. Gregory. 2001. Landscape cover type and pattern dynamics in fragmented southern Great Plains grasslands, USA. Landscape Ecology 16:677-690.

Dale, B. C., P. A. Martin, and P. S. Taylor. 1997. Effects of hay management on grassland songbirds in Saskatchewan. Wildlife Society Bulletin 25:616-626.

Flanders, A. A., W. P. Kuvlesky, D. C. Ruthven, R. E. Zaiglin, R. L. Bingham, T. E. Fulbright, F. Hernandez, and L. A. Brennan. 2006. Effects of invasive exotic grasses on South Texas rangeland breeding birds. The Auk 123:171-182.

French, D. D., and N. Picozzi. 2002. 'Functional groups' of bird species, biodiversity and landscapes in Scotland. Journal of Biogeography 29:231-259.

Gordon, C. E. 2000a. Fire and cattle grazing on wintering sparrows in Arizona grasslands. Journal of Range Management 53:384-389.

Gordon, C.E. 2000b. Movement patterns of wintering grassland sparrows in Arizona. The Auk 117:748-759.

Granfors, D. A., K. E. Church, and L. M. Smith. 1996. Eastern Meadowlarks nesting in rangelands and Conservation Reserve Program fields in Kansas. Journal of Field Ornithology 67:222-235.

Grzybowski, J. A. 1982. Population structure in grassland bird communities during winter. The Condor 84:137-152.

Grzybowski, J. A. 1983. Patterns of space use in grassland bird communities during winter. The Wilson Bulletin 95:591-602.

Hammerson, G. 1994. Bachman's sparrow (*Aimophila aestivalis*) species management abstract. Arlington, VA: The Nature Conservancy.

Hunter, W. C., D. A. Buehler, R. A. Canterbury, J. L. Confer, and P. B. Hamel. 2001a. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin 29:440-455.

Hunter, W. C., L. H. Peoples, and J. A. Collazo. 2001b. Partners in Flight conservation plan for the South Atlantic Coastal Plain. US Fish and Wildlife Service.

Hurlbert, A. H., and J. P. Haskell. 2003. The effect of energy and seasonality on avian species richness and community composition. The American Naturalist 161:83-97.

Igl, L. D., and B. M. Ballard. 1999 . Habitat associations of migrating and overwintering grassland birds in Southern Texas. The Condor 101:771-782.

Jonas, J. L., M. R. Whiles, and R. E. Charlton. 2002. Aboveground invertebrate responses to land management differences in a central Kansas grassland. Environmental Entomology 31:1142-1152.

Kindscher, K., and L. L. Tieszen. 1998. Floristic and soil organic matter changes after five and thirty-five years of native tallgrass prairie restoration. Restoration Ecology 6:181-196.

King, D. I., R. M. Degraaf, and C. R. Griffin. 2001. Productivity of early successional shrubland birds in clearcuts and groupcuts in an eastern deciduous forest. Journal of Wildlife Management 65:345-350.

Klute, D. S., R. J. Robel, and K. E. Kemp. 1997. Seed availability in grazed pastures and Conservation Reserve Program fields during winter in Kansas. Journal of Field Ornithology 68:253-258.

Knick, S. T., and J. T. Rotenberry. 2000. Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. Ecology 8:220-227.

Kobal, S. N., N. F. Payne, and D. R. Ludwig. 1999. Habitat/area relationships, abundance, and composition of bird communities in 3 grassland types. Transactions of the Illinois State Academy of Science 92:109-131.

Marcus, J. F., W. E. Palmer, and P. T. Bromley. 2000. The effects of farm field borders on overwintering sparrow densities. Wilson Bulletin 112:517-523.

Martin, T. E. 1995. Avian life history evolution in relation to nest sites, nest predation, and food. Ecological Monographs 65:101-127.

McCraken, D. I., B. E. M., S. Blake, and G. N. Foster. 2004. Productivity and profitability: the effects of farming practices on the prey of insectivorous birds. Pages 75-87 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

McIntyre, N. E., and T. R. Thompson. 2003. A comparison of conservation reserve program habitat plantings with respect to arthropod prey for grassland birds. American Midland Naturalist 150:291-301.

Mikami, O. K., and M. Kawata. 2002. The effects of individual interactions and habitat preferences on spatial structure in a grassland bird community. Ecography 25:200-214.

Moreby, S. J. 2004. Birds of lowland arable farmland: the importance and identification of invertebrate diversity in the diet of chicks. Pages 21-35 *in* H. van Emden, and M. Rothschild, editors. Insect and Bird Interactions. Andover, Hampshire: Intercept Publishers.

O'Leary, C. H., and D. W. Nyberg. 2000. Treelines between fields reduce the density of grassland birds. Natural Areas Journal 20:243-249.

Patterson, M. 2002. Analysis of bird point count data for the Mill Pond Site, Neawanna Ecological Observatory, Seaside, Clatsop Co., Oregon. Oregon Birds 28:168-171.

Perkins, A. J., M. J. Whittingham, R. B. Bradbury, J. D. Wilson, A. J. Morris, and P. R. Barnett. 2000. Habitat characteristics affecting use of lowland agricultural grassland by birds in winter. Biological Conservation 95:279-294.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1998. Site fidelity of wintering Henslow's Sparrows. Journal of Field Ornithology 69:486-490.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1999. Habitat requirements of Henslow's Sparrows wintering in silvicultural lands of the Gulf Coastal Plain. The Auk 116:109-115.

Pons, P., B. Lambert, E. Rigolot, and R. Prodon. 2003. The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape. Biodiversity and Conservation 12:1843-1860.

Porter, S. D., and D. A. Savignano. 1990. Invasion of polygyne fire ants decimates native ants and disrupts arthropod community. Ecology 71:2095-2106.

Porter, S. D., and W. R. Tschinkel. 1987. Foraging in *Solenopsis invicta* (Hymenoptera: Formicidae): effects of weather and season. Environmental Entomology 16:802-808.

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

Rising, J. D., and D. D. Beadle. 1996. A guide to the indentification and natural history of the sparrows of the United States and Canada. San Diego: Academic Press.

Robel, R. J., B. M. Press, B. L. Henning, K. W. Johnson, H. D. Blocker, and K. E. Kemp. 1995. Nutrient and energetic characteristics of sweepnet collected invertebrates. Journal of Field Ornithology 66:44-53.

Robinson, R. A., J. D. Hart, J. M. Holland, and D. Parrott. 2004. Habitat use by seed-eating birds: a scale dependent approach. Ibis 146:87-98.

Root, T. 1988. Atlas of wintering North American birds: an analysis of Christmas Bird Count data. Chicago: The University of Chicago Press.

Shochat, E., M. A. Patten, D. W. Morris, D. L. Reinking, D. H. Wolfe, and S. K. Sherrod. 2005. Ecological traps in isodars: effects of tallgrass prairie management on bird nest success. Oikos 111:159-169.

Sibley, D. A. 2000. The Sibley guide to birds. New York: Chanticleer Press, Inc.

Smith, M. D., P. J. Barbour, W. Burger, Jr., and S. J. Dinsmore. 2005. Density and diversity of overwintering birds in managed field borders in Mississippi. Wilson Bulletin 117:258-269.

Stiles, J. H., and R. H. Jones. 1998. Distributuion of the red imported fire ant, *Solenopsis invicta*, in road and powerline habitats. Landscape Ecology 13:335-346.

Tucker, J. W., Jr., and W. D. Robinson. 2003. Influence of season and frequency of fire on Henslow's sparrows (Ammodramus Henslowii) wintering on Gulf Coast pitcher plant bogs. The Auk 120:96-106.

USDA Forest Service. 2004. Chattahoochee-Oconee National Forests land and resource management plan. U.S. Department of Agriculture, Forest Service, Southern Region. Report Management Bulletin R8-MB 113 A.

Vickery, P. D., and J. R. Herkert. 2001. Recent advances in grassland bird research: where do we go from here? The Auk 118:11-15.

Vickery, P. D., P. L. Tubaro, J. M. Cardoso da Silva, B. G. Peterjohn, J. Herkert, R., and R. B. Cavalcanti. 1999. Conservation of grassland birds in the western hemisphere. Studies in Avian Biology 19:2-26.

Vitz, A. C., and A. D. Rodewald. 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127:477-486.

Wiens, J. A. 1969. An approach to the study of ecological relationships among grassland birds. Ornithological Monographs 8:1-93.

Wojcik, D. P., C. R. Allen, R. J. Brenner, E. A. Forys, D. P. Jouvenaz, and R. S. Lutz. 2001. Red imported fire ants: impact on biodiversity. American Entomologist 47:16-23.

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

LITERATURE CITED

Anderson, B. W., R. D. Ohmart, and J. Rice. 1983. Avian and vegetation community structure and their season relationships in the lower Colorado River valley. Condor 85:392-405.

Anderson, R.C., T. Leahy, and S.S. Dhillion. 1989. Numbers and biomass of selected insect groups on burned and unburned sand prairie. American Midland Naturalist 122:151-162.

Arguedas-Negrini, N. 2001. Distribution, habitat and behavior of grasshopper sparrows, *Ammodramus savannarum* (Passeriformes: Emberizidae) in northwestern Nicaragua. Revista de Biologia Tropical 49:703-708.

Askins, R. A. 1999. History of grassland birds in Eastern North America. Studies in Avian Biology 19:60-71.

Askins, R.A. 2000. Restoring North America's birds: lessons from landscape ecology. Second edition. New Haven: Yale University Press.

Atkinson, P.W., D. Buckingham, and A.J. Morris. 2004. What factors determine where invertebrate-feeding birds forage in dry agricultural grasslands? Ibis 146:99-107.

Bajema, R. A., T. L. DeVault, P. E. Scott, and S. L. Lima. 2001. Reclaimed coal mine grasslands and their significance for Henslow's Sparrows in the American Midwest. The Auk 118:422-431.

Bajema, R. A., and S. L. Lima. 2001. Landscape-level analyses of Henslow's Sparrow (*Ammodramus henslowii*) abundance in reclaimed coal mine grasslands. American Midland Naturalist 145:288-298.

Bakker, J. P. and F. Berendse. 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. Tree 14:63-67.

Bakker, K. K., D. E. Naugle, and K. F. Higgins. 2002. Incorporating landscape attributes into models for migratory grassland bird conservation. Conservation Biology 6:1638-1646.

Barker, A.M. 2004. Insects as food for farmland birds: is there a problem? Pages 37-50 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

Beissinger, S. B., J. M. Reed, J. M. Wunderle, Jr., S. K. Robinson, and D. M. Finch. 2000. Report of the AOU conservation committee on the Partners in Flight species prioritization plan. The Auk 117:549-561.

Benton, T.G., D.M. Bryant, L. Cole, and H.Q.P. Crick. 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. Journal of Applied Ecology 39:673-687.

Berg, A. 2002. Composition and diverstiy of bird communities in Swedish farmland-forest mosaic landscapes. British Trust for Ornitholgy 49:153-165.

Berthelsen, P. S., and L. M. Smith. 1995. Nongame bird nesting on CRP lands in the Texas Southern High Plains. Journal of Soil and Water Conservation 50:672-675.

Bland, R.G. and H.E. Jaques. 1978. How to know the insects. Third edition. Boston: McGraw-Hill.

Bolton, B. 1994. Identification guide to the ant genera of the world. Cambridge, MA: Harvard University Press.

Borror, D.J., C.A. Triplehorn, and N.F. Johnson. 1989. An introduction to the study of insects. Sixth edition. Fort Worth: Harcourt Brace College Publishers.

Borror, D.J. and R.E. White. 1970. A field guide to the insects: America north of Mexico. Boston: Houghton Mifflin Company.

Braun-Blanquet, J. 1932. Plant sociology: the study of plant communities. New York: McGraw-Hill Book Company, Inc.

Brawn, J.D., S.K. Robinson, and F.R. Thompson, III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32:251-276.

Brennan, L. A., and W. P. Kuvlesky. 2005. North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1-13.

Briggs, J. N., A. K. Knapp, J. M. Blair, J. L. Heisler, G. A. Hoch, M. S. Lett, and J. K. McCarron. 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55:243-254.

Brinkman, M.A., W.A. Gardern, R.M. Ipser, and S.K. Diffie. 2001. Ground-dwelling ant species attracted to four food baits in Georgia. Journal of Entomological Science 36:461-463.

Brotons, L., A. Wolff, G. Paulus, and J.L. Martin. 2005. Effect of adjacent agricultural habitat on the distribution of passerines in natural grasslands. Biological Conservation 124:407-414.

Broyer, J. 2003. Unmown refuge areas and their influence on the survival of grassland birds in the Saone valley (France). Biodiversity and Conservation 12:1219-1237.

Butler, S.J. and S. Gillings. 2004. Quantifying the effects of habitat structure on prey detectability and accessibility to farmland birds. Ibis 146:123-130.

Castano-Meneses, G. and J.G. Palacios-Vargas. 2003. Effects of fire and agricultural practices on neotropical ant communities. Biodiversity and Conservation 12:1913-1919.

Childers, E. L., T. L. Sharik, and C. S. Adkisson. 1986. Effects of loblolly pine plantations on songbird dynamics in the Virginia Piedmont. Journal of Wildlife Management 1986:406-413

Chu, H.F. and L.K. Cutkomp. 1992. How to know the immature insects. Second edition. Boston: McGraw-Hill Publishers.

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 114:324-352.

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 114:324-352.

Cooper, R. J. 1999. Partners in Flight bird conservation plan for the southern Piedmont (physiographic area 11). The Plains, VA: American Bird Conservancy.

Cooper, R.J. and R.C. Whitmore. 1990. Arthropod sampling methods in ornithology. Studies in Avian Biology 13:29-37.

Coppedge, B.R., D.M. Engle, S.D. Fuhlendorf, R. E. Masters, and M. S. Gregory. 2001a. Landscape cover type and pattern dynamics in fragmented southern Great Plains grasslands, USA. Landscape Ecology 16:677-690.

Coppedge, B. R., D.M. Engle, R. E. Masters, and M. S. Gregory. 2001b. Avian response to landscape change in fragmented southern Great Plains grasslands. Ecological Applications 11:47-59.

Cowell, M. C. 1998. Historical change in vegetation and disturbance on the Georgia Piedmont. American Midland Naturalist 140:78-89.

Dale, B. C., P. A. Martin, and P. S. Taylor. 1997. Effects of hay management on grassland songbirds in Saskatchewan. Wildlife Society Bulletin 25:616-626.

Denevan, W. M. 1992. The pristine myth: the landscape of the Americas in 1492. Annals of the Association of American Geographers 82:369-385.

DeVault, T.L., P.E. Scott, R.A. Bajema, and S.L. Lima. 2002. Breeding bird communities in reclaimed coal-mine grasslands in the American Midwest. Journal of Field Ornithology 73:268-275.

DiGiulio, M., P.J. Edwards, and E. Meister. 2001. Enhancing insect diversity in agricultural grasslands: the roles of management and landscape structure. Journal of Applied Ecology 38:310-319.

Doolittle, W. E. 1992. Agriculture in North America on the eve of contact: a reassessment. Annals of the Association of American Geographers 82:386-401.

Dunn, C. P., F. Stearns, G. R. Guntenspergen, and D. M. Sharpe. 1993. Ecological benefits of the Conservation Reserve Program. Conservation Biology 7:132-139.

Flanders, A. A., W. P. Kuvlesky, D. C. Ruthven, R. E. Zaiglin, R. L. Bingham, T. E. Fulbright, F. Hernandez, and L. A. Brennan. 2006. Effects of invasive exotic grasses on South Texas rangeland breeding birds. The Auk 123:171-182.

French, D. D., and N. Picozzi. 2002. 'Functional groups' of bird species, biodiversity and landscapes in Scotland. Journal of Biogeography 29:231-259.

Giuliano, W. M., and S. E. Daves. 2002. Avian response to warm-season grass use in pasture and hayfield management. Biological Conservation 106:1-9.

Gordon, C. E. 2000a. Fire and cattle grazing on wintering sparrows in Arizona grasslands. Journal of Range Management 53:384-389.

Gordon, C.E. 2000b. Movement patterns of wintering grassland sparrows in Arizona. The Auk 117:748-759.

Grand, J. and S. A. Cushman. 2003. A multi-scale analysis of species-environment relationships: breeding birds in a pitch pine-scrub oak (*Pinus rigida-Quercus ilicifolia*) community. Biological Conservation 112:307-317.

Granfors, D. A., K. E. Church, and L. M. Smith. 1996. Eastern Meadowlarks nesting in rangelands and Conservation Reserve Program fields in Kansas. Journal of Field Ornithology 67:222-235.

Gray, L. C. 1933. History of agriculture in the southern United States to 1860. Volume II. Baltimore: Waverly Press, Inc.

Grzybowski, J. A. 1982. Population structure in grassland bird communities during winter. The Condor 84:137-152.

Grzybowski, J. A. 1983. Patterns of space use in grassland bird communities during winter. The Wilson Bulletin 95:591-602.

Hall, D. L. and M. R. Willig. 1994. Mammalian species composition, diversity, and succession in Conservation Reserve Program grasslands. The Southwestern Naturalist 39:1-10.

Hamel, P. B., W. P. Smith, D. J. Twedt, J. R. Woehr, E. Morris, R. D. Hamilton, and R. J. Cooper. 1996. A land manager's guide to point counts of birds of the Southeast. USDA Forest Service General Technical Report SO-120.

Hammerson, G. 1994. Bachman's sparrow (*Aimophila aestivalis*) species management abstract. The Nature Conservancy.

Harper, F., editor. 1998. The travels of William Bartram. Athens, GA: The University of Georgia Press.

Hays, R. L. and A. H. Farmer. 1990. Effects of the CRP on wildlife habitat: emergency haying in the Midwest and pine plantings in the Southeast. Transactions of the North American Wildlife and Natural Resources Conference 55:30-39.

Herkert, J. R. 1994. The effects of habitat fragmentation on midwestern grassland bird communities. Ecological Applications 4:461-471.

Horn, D. J. and R. R. Koford. 2000. Relation of grassland bird abundance to mowing of Conservation Reserve Program fields in North Dakota. Wildlife Society Bulletin 28:653-659.

Howe, F.P., R.L. Knight, L.C. McEwen, and T.L. George. 2000. Diet switching and food delivery by shrubsteppe passerines in response to an experimental reduction in food. Western North American Naturalist 60:139-154.

Hunter, W.C., D.A. Buehler, R.A. Canterbury, J.L. Confer, and P.B. Hamel. 2001a. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin 29:440-455.

Hunter, W. C., L. H. Peoples, and J. A. Collazo. 2001b. Partners in Flight conservation plan for the South Atlantic Coastal Plain. U.S. Fish and Wildlife Service.

Hurlbert, A. H., and J. P. Haskell. 2003. The effect of energy and seasonality on avian species richness and community composition. The American Naturalist 161:83-97.

Hutto, R. L., S. M. Pletschet, and P. Hendricks. 1986. A fixed-radius point count method for nonbreeding and breeding-season use. The Auk 103:593-602.

Igl, L. D., and B. M. Ballard. 1999. Habitat associations of migrating and overwintering grassland birds in Southern Texas. The Condor 101:771-782.

Ipser, R.M., M.A. Brinkman, W.A. Gardner, and H.B. Peeler. 2004. A survey of ground-dwelling ants (Hymenopter: Formicidae) in Georgia. Florida Entomologist 87:253-260.

Izhaki, I., D.J. Levey, and W.R. Silva. 2003. Effects of prescribed fire on an ant community in Florida pine savanna. Ecological Entomology 28:439-448.

Jansen, R., R. M. Little, and T. M. Crowe. 1999. Implications of grazing and burning of grasslands on the sustainable use of francolins (*Francolinus* spp.) and on overall bird conservation in the highlands of Mpumalanga province, South Africa. Biodiversity and Conservation 8:587-602.

Jonas, J.L., M.R. Whiles, and R.E. Charlton. 2002. Aboveground invertebrate responses to land management differences in a central Kansas grassland. Environmental Entomology. 31:1142-1152.

Jusino-Atresino, R. and S.A. Philips, Jr. 1994 Impact of red imported fire ants on the ant fauna of central Texas. Pages 258-269 In D.F. Williams, editor. Exotic ants: biology, impact, and control of introduced species. Boulder, CO: Westview Press.

Kempton, R. A., and L. R. Taylor. 1976. Models and statistics for species diversity. Nature 262:818-820.

Kindscher, K., and L. L. Tieszen. 1998. Floristic and soil organic matter changes after five and thirty-five years of native tallgrass prairie restoration. Restoration Ecology 6:181-196.

King, D. I., R. M. Degraaf, and C. R. Griffin. 2001. Productivity of early successional shrubland birds in clearcuts and groupcuts in an eastern deciduous forest. Journal of Wildlife Management 65:345-350.

Klute, D. S., R. J. Robel, and K. E. Kemp. 1997. Seed availability in grazed pastures and Conservation Reserve Program fields during winter in Kansas. Journal of Field Ornithology 68:253-258.

Knick, S. T., and J. T. Rotenberry. 1995. Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. Conservation Biology 9:1059-1071.

Knick, S. T., and J. T. Rotenberry. 2000. Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. Ecology 8:220-227.

Knopf, F. L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15:247-257.

Kobal, S. N., N. F. Payne, and D. R. Ludwig. 1999. Habitat/area relationships, abundance, and composition of bird communities in 3 grassland types. Transactions of the Illinois State Academy of Science 92:109-131.

Komarek, E. V., Sr., editor. 1962. Proceedings of the First Tall Timbers Fire Ecology Conference. Tall Timbers Research Station, Tallahassee, FL.

Lorimer, C. G. 2001. Historical and ecological roles of disturbance in Eastern North American forests: 9,000 years of change. Wildlife Society Bulletin 29:425-439.

Mackay, W. and E. Mackay. 2003. The ants of North America. Centennial Museum, Laboratory for Environmental Biology. The University of Texas, El Paso, TX.

Marcus, J. F., W. E. Palmer, and P. T. Bromley. 2000. The effects of farm field borders on overwintering sparrow densities. Wilson Bulletin 112:517-523.

Martin, S. 1969. A technique for capturing nesting grassland birds with mist nets. Bird Banding 40:233-237.

Martin, T. E. 1995. Avian life history evolution in relation to nest sites, nest predation, and food. Ecological Monographs 65:101-127.

Mayer, A.L. and G.N. Cameron. 2003. Landscape characteristics, spatial extent, and breeding bird diversity in Ohio, USA. Diversity and Distributions 9:297-311.

McCraken, D.I., E.M. Bigmal, S. Blake, and G.N. Foster. 2004. Productivity and profitability: the effects of farming practices on the prey of insectivorous birds. Pages 75-87 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Publishers.

McIntyre, N.E. and T.R. Thompson. 2003. A comparison of conservation reserve program habitat plantings with respect to arthropod prey for grassland birds. American Midland Naturalist 150:291-301.

Mikami, O. K., and M. Kawata. 2002. The effects of individual interactions and habitat preferences on spatial structure in a grassland bird community. Ecography 25:200-214.

Moreby, S.J. 2004. Birds of lowland arable farmland: the importance and indentification of invertebrate diversity in the diet of chicks. Pages 21-35 *in* H. van Emden and M. Rothschild, editors. Insect and bird interactions. Andover, Hampshire: Intercept Pubishers.

Morimoto, D. C. and F. E. Wasserman. 1991. Dispersion pattersn and habitat associations of rufous-sided towhees, common yellowthroats, and prairie warblers in the southeastern Massachusetts pine barres. The Auk 108:264-276.

Natural Resource Spatial Analysis Laboratory. 1998. 1998 land cover map of Georgia. Institute of Ecology, University of Georgia, Athens, GA.

National Resources Conservation Service. 2005. Soil Data Mart. http://soildatamart.nrcs.usda.gov/ Accessed 05/05/2006.

Nudds, T.D. 1977. Quantifying the vegetation structure of wildlife cover. Wildlife Society Bulletin 5:113-117.

Nur, N., S. L. Jones, and G. R. Geupel. 1999. A statistical guide to data analysis of avian monitoring programs. U.S. Department of the Interior, Fish and Wildlife Service. Report BTP-R6001-1999.

Nuttle, T., A. Leidolf, and L. W. Burger, Jr. 2003. Assessing conservation value of bird communities with Partners in Flight-based ranks. The Auk 120:541-549.

O'Leary, C. H., and D. W. Nyberg. 2000. Treelines between fields reduce the density of grassland birds. Natural Areas Journal 20:243-249.

Patterson, M. 2002. Analysis of bird point count data for the Mill Pond Site, Neawanna Ecological Observatory, Seaside, Clatsop Co., Oregon. Oregon Birds 28:168-171.

Patterson, M. P. and L. B. Best. 1996. Bird abundance and nesting success in Iowa CRP fields: the importance of vegetaion structure and composition. The American Midland Naturalist 135:153-167.

Perkins, A. J., M. J. Whittingham, R. B. Bradbury, J. D. Wilson, A. J. Morris, and P. R. Barnett. 2000. Habitat characteristics affecting use of lowland agricultural grassland by birds in winter. Biological Conservation 95:279-294.

Peterjohn, B. G., and J. R. Sauer. 1999. Population status of North American grassland birds from the North American Breeding Bird Survey, 1966-1996. Studies in Avian Biology 19:27-44.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1998. Site fidelity of wintering Henslow's Sparrows. Journal of Field Ornithology 69:486-490.

Plentovich, S. M., N. R. Holler, and G. E. Hill. 1999. Habitat requirements of Henslow's Sparrows wintering in silvicultural lands of the Gulf Coastal Plain. The Auk 116:109-115.

Pons, P., B. Lambert, E. Rigolot, and R. Prodon. 2003. The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape. Biodiversity and Conservation 12:1843-1860.

Porter, S.D. and D.A. Savignano. 1990. Invasion of polygyne fire ants decimates native ants and disrupts arthropod community. Ecology 71:2095-2106.

Porter, S.D. and W.R. Tschinkel. 1987. Foraging in *Solenopsis invicta* (Hymenoptera: Formicidae): effects of weather and season. Environmental Entomology 16:802-808.

Powell, R. A. 2000. Animal home ranges and territories and home range estimators. Pages 65-110 *in* L. F. Boitani and T.K. Fuller, editors. Research Techniques in Animal Ecology. New York: Columbia University Press.

Rasnake, M. 1992. Management of warm season bunch grasses. Proceedings of the Southern Pasture and Forage Crop Improvement Conference 48:38-39.

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

Rising, J. D., and D. D. Beadle. 1996. A guide to the indentification and natural history of the sparrows of the United States and Canada. San Diego: Academic Press.

Robel, R.J., B.M. Press, B.L. Henning, K.W. Johnson, H.D. Blocker, and K.E. Kemp. 1995. Nutrient and energetic characteristics of sweepnet collected invertebrates. Journal of Field Ornithology 66:44-53.

Robinson, R. A., J. D. Hart, J. M. Holland, and D. Parrott. 2004. Habitat use by seed-eating birds: a scale dependent approach. Ibis 146:87-98.

Robinson, R. A., J. D. Wilson, and H.Q.P. Crick. 2001. The importance of arable habitat for farmland birds in grassland landscapes. Journal of Applied Ecology 38:1059-1069.

Rodewald, A.D. and A.C. Vitz. 2005. Edge- and area-sensitivity of shrubland birds. Journal of Wildlife Management 69:135-151.

Root, T. 1988. Atlas of wintering North American birds: an analysis of Christmas Bird Count data. Chicago: The University of Chicago Press.

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

Samson, F. and E.L. Knopf. 1994. Prairie conservation in North America. Bioscience 44:418-422.

SAS Institute 2005. SAS OnlineDoc version 9.1.3. Cary, NC: SAS Institute, Inc.

Schotzko, D.J. and L.E. O'Keeffe. 1986. Comparison of sweepnet, d-vac, and absolute sampling for *Lygus Hesperus* (Heteroptera: Miridae) inlentils. Journal of Economic Entomology 79:224-228.

Schotzko, D.J. and L.E. O'Keeffe. 1989. Comparison of sweepnet, d-vac, and absolute sampling, and diel variation of sweepnet sampling estimates in lentils for pea aphid (Homoptera: Aphididae), nabids (Hemiptera: Nabidae), lady beetles (Coleoptera: Coccinellidae), and lacewings (Neuroptera: Chrysopidae). Journal of Economic Entomology 82:491-506.

Shochat, E., M.A. Patten, D.W. Morris, D.L. Reinking, D.H. Wolfe, and S.K. Sherrod. 2005a. Ecological traps in isodars: effects of tallgrass prairie management on bird nest success. Oikos 111:159-169.

Shochat, E., D.H. Wolfe, M.A. Patten, D.L. Reinking, and S.K. Sherrod. 2005b. Tallgrass prairie management and bird nest success along roadsides. Biological Conservation 121:399-407.

Shriver, W. G., T. P. Hodgman, J. P. Gibbs, and P. D. Vickery. 2004. Landscape context influences salt marsh bird diversity and area requirements in New England. Biological Conservation 119:545-553.

Sibley, D. A. 2000. The Sibley guide to birds. New York: Chanticleer Press, Inc.

Skowno, A. L. and W. J. Bond. 2003. Bird community composition in an actively managed savanna reserve, importance of vegetation structure and vegetation composition. Biodiversity and Conservation 12:2279-2294.

Smith, M. D., P. J. Barbour, W. Burger, Jr., and S. J. Dinsmore. 2005. Density and diversity of overwintering birds in managed field borders in Mississippi. Wilson Bulletin 117:258-269.

Soderstrom, B., T. Part, and E. Linnarsson. 2001. Grazing effects on between-year variation of farmland bird communities. Ecological Applications 11:1141-1150.

Standen, V. 2000. The adequacy of collecting techniques for estimating species richness of grassland invertebrates. Journal of Applied Ecology 37:884-893.

Stiles, J.H. and R.H. Jones. 1998. Distribution of the red imported fire ant, *Solenopsis invicta*, in road and powerline habitats. Landscape Ecology 13335-346.

ter Braak, C.J.F. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. Ecology 67:1167-1179.

ter Braak, C. J. F., and P. Šmilauer. 2002 CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 4.5). Ithaca, NY: Microcomputer Power.

Titeux, N., M. Dufrene, J.P. Jacob, M. Paquay, and P. Defourny. 2004. Multivariate analysis of a fine-scale breeding bird atlas using a geographical information system and partial canonical correspondence analysis: environmental and spatial effects. Journal of Biogeography 31:1841-1856.

Tucker, J. W., Jr., and W. D. Robinson. 2003. Influence of season and frequency of fire on Henslow's sparrows (*Ammodramus henslowii*) wintering on Gulf Coast pitcher plant bogs. The Auk 120:96-106.

USDA Forest Service. 2004. Chattahoochee-Oconee National Forests land and resource management plan. U.S. Department of Agriculture, Forest Service, Southern Region. Report Management Bulletin R8-MB 113 A.

Van Lear, D. H., W. D. Carroll, P. R. Kapeluck, and R. Johnson. 2005. History and restoration of the longleaf pine-grassland ecosystem: implications for species at risk. Forest Ecology and Management 211:150-165.

Vickery, P. D., and J. R. Herkert. 2001. Recent advances in grassland bird research: where do we go from here? The Auk 118:11-15.

Vickery, P. D., J. Herkert, R., F. L. Knopf, J. Ruth, and C. E. Keller. 2000. Grassland birds: an overview of threats and recommended management strategies. Pages 74–77 in R. Bonney, D. N. Pashley, R. J. Cooper, and L. Niles, editors.

Strategies for bird conservation: the Partners in Flight planning process. Proceedings of the 3rd Partners in Flight workshop (1995). Proceedings RMRS-P-16. U.S. Forest Service, Rocky Mountain Research Station, Ogden, Utah.

Vickery, P. D., P. L. Tubaro, J. M. Cardoso da Silva, B. G. Peterjohn, J.R. Herkert, and R. B. Cavalcanti. 1999. Conservation of grassland birds in the western hemisphere. Studies in Avian Biology 19:2-26.

Vitz, A. C., and A. D. Rodewald. 2006. Can regenerating clearcuts benefit mature-forest songbirds? An examination of post-breeding ecology. Biological Conservation 127:477-486.

Vogl, R.J. 1973. Fire in the Southeastern grasslands. Proceedings of the Tall Timbers Fire Ecology Conference 12:178-193.

Vogt, J.T., R.A. Grantham, E. Corbett, S.A. Rice, and R.E. Wright. 2002. Dietary habitats of *Solenopsis invicta* (Hymenoptera: Formicidae) in four Oklahoma habitats. Environmental Entomology 31:47-53.

Walk, J. W. and R. E. Warner. 2000. Grassland management for the conservation of songbirds in the Midwestern USA. Biological Conservation 94:165-172.

Whiles, M.R. and R.E. Charlton. 2006. The ecological significance of tallgrass prairie arthropods. Annual Review of Entomology 51:387-412.

Wiens, J. A. 1969. An approach to the study of ecological relationships among grassland birds. Ornithological Monographs 8:1-93.

Wiens, J.A. 1973. Pattern and process in grassland bird communities. Ecological Monographs 43:237-270.

Wiens, J.A. 1974. Habitat heterogeneity and avian community structure in North American grasslands. American Midland Naturalist 91:195-213.

Wojcik, D.P. 1994. Impact of the red imported fire ants on native ant species in Florida. Pages 269-281 *in* D.F. Williams, editor. Exotic ants: biology, impact, and control of introduced species. Boulder, CO: Westview Press.

Wojcik, D.P., C.R. Allen, R.J. Brenner, E.A. Forys, D.P. Jouvenaz, and R.S. Lutz. 2001. Red imported fire ants: impact on biodiversity. American Entomologist 47:16-23.

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

Zimmerman, J.L. 1992. Density-independent factors affecting the avian diversity of the tallgrass prairie community. Wilson Bulletin 104:85-94.