THE IMPORTANCE OF SOUTHERN APPALACHIAN WETLANDS TO BREEDING BIRDS

A Thesis

by

JASON FREDERICK BULLUCK

Submitted to the Graduate School

Appalachian State University

in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

February 2003

Major Department: Biology

THE IMPORTANCE OF SOUTHERN APPALACHIAN WETLANDS TO BREEDING BIRDS

A Thesis

by

Jason Frederick Bulluck

May 2003

APPROVED BY:

Matthew P. Rowe

Matthew P. Rowe
Chairperson, Thesis Committee

Ray S Williams

Ray S. Williams
Member, Thesis Committee

Gary L. Walker

Gary L. Walker
Member, Thesis Committee

Vicki J. Martin

Vicki J. Martin
Chairperson, Department of Biology

Judith E. Domer

Judith E. Domer
Dean of Graduate Studies and Research

ABSTRACT

THE IMPORTANCE OF SOUTHERN APPALACHIAN WETLANDS TO BREEDING BIRDS

Jason Frederick Bulluck, B.S., Appalachian State University

Thesis Chairperson: Matthew P. Rowe

Southern Appalachian wetlands are among the scarcest wetlands in North America and face severe pressures from human land use practices. Though occurring in a region displaying high avifaunal diversity, no prior study has addressed the use of this habitat type by breeding birds. My research assessed avian habitat requirements in southern Appalachian wetlands through avian censusing and vegetation analysis of study sites. I censused 57 southern Appalachian wetlands with fixed-radius point counts in the breeding seasons of 1999, 2000 and 2001. In the field, I also measured 23 habitat variables including wetland size, beaver presence, livestock presence, edge type, and percent cover of nine vegetation layers. Using regression techniques, between-site differences in community-level species diversity, species richness, and total bird abundance were analyzed as a function of habitat and vegetation characters. At a finer level, species richness and abundance within 13 guilds based upon breeding habitat, migration habit, nesting location, and diet were also regressed with wetland characteristics. Some unexpected significant predictors of avian community structure were found. Presence of a gradual edge positively influenced breeding birds at the community level; gradual edge was the sole predictor of species diversity, and also predicted greater species richness. Wetlands impacted by beaver displayed greater community-level species richness and total avian abundance than non-beaver wetlands. Gradual edges and beaver impact also predicted higher species richness and abundance in several bird guilds, especially with Neotropical migrants and woodland breeding species. My findings suggest that when managing habitats that inherently contain edges, promotion of gradual over abrupt edges may result in ecotones that are more diverse in terms of breeding birds. Also, steps taken to allow the engineering of beaver in southern Appalachian wetlands may result in greater vegetational complexity and insect productivity in wetlands, thereby enhancing the breeding bird communities of these habitats. If the findings of this research concerning edge habitats and beaver activity, as well as other wetland characters, were employed in management plans for southern Appalachian wetlands, more successful conservation of many declining bird species may be achieved.

ACKNOWLEDGEMENTS

I would like to thank the U.S. Fish and Wildlife Service, the Wilderness Society, and the Explorers Club for providing funding for this project. Nora Murdock, now of the National Park Service, and Dave Lee, of the North Carolina Museum of Natural Sciences, provided the majority of financial assistance for my thesis data collection, through the USFWS. Chris Haney, now of The Nature Conservancy, also orchestrated a great deal of funding for my research through The Wilderness Society. I also thank the Cratis D. Williams Graduate School for financial assistance and the A.S.U. Biology Department for use of departmental facilities and equipment, especially vans, without which this research could not have been completed.

Many others have assisted in various ways with my thesis research at different stages. To list only a few, I thank Bob Cherry and Tom Davis of the National Park Service for their assistance with access to study sites on the Blue Ridge Parkway. Dennis Herman, of the N.C. Museum of Natural Sciences, provided essential assistance with locating and accessing many of my study sites, especially those that are privately owned. This project would not have been possible without Dennis' help throughout the duration of the study. Este Stifel and Beth Bockoven of The Nature Conservancy also helped greatly with access issues pertaining to study sites under TNC jurisdiction, and provided me with important natural history records for many TNC sites. I would also like to thank Reed Rossell, Research Associate at the UNCA Environmental Studies Department, for

his help with Tulula Bog in particular. Reed provided me with valuable historical information about Tulula Bog, and supplied bird data that he collected there in previous years, in order to familiarize me with this unique southern Appalachian wetland. I would also like to thank various landowners in the Appalachians of Virginia and North Carolina, who cooperated with me so that we may together better understand and manage the delicate wetlands contained on their properties. Without the collaboration with these landowners, the sample size of my study would have suffered greatly and this study would have thus been far less rigorous and thorough.

Most of all, I would like to thank my thesis chair, Matthew Rowe. Matt has been the most influential mentor of mine throughout my college education. He shows fervent passion for conservation and has helped to cultivate this passion in me via his instruction. I also thank Ray Williams and Gary Walker. Gary and Ray provided immeasurable assistance with experimental design considerations and shared their knowledge of the biogeography of southern Appalachian ecosystems in the classroom and in the field. Ray and Gary have also been of great help in the writing process of my thesis.

DEDICATION

To my family, Thomas W. Bulluck, Kaye O. Bulluck, and Travis Bulluck, for providing support through the years of studies that led up to this thesis. I also dedicate this to my fiancée, Lesley Penfield, for her undying support, and for endlessly sharing a love for the natural world, and the determination to contribute to conservation in the southern Appalachians.

TABLE OF CONTENTS

LIST OF TABLES

INTRODUCTION

I. Wetlands: a resource in peril - background

Wetlands are among the most studied ecosystems in the United States, due to their importance to humans and wildlife. For humans, wetlands provide invaluable ecosystem services. For example, wetlands prevent local flooding and purify water through the stabilization of sediment and nutrients (Woodward & Wui 2001). Additionally, due to the slow deposition of sediment, wetlands house a detailed fossil record of past floral and faunal communities. Thus, wetlands often provide the most accessible and accurate fossil record of inland habitats (Whitehouse et al. 1997). This record of past life on earth allows humans to better understand the mechanisms and results of climate change throughout the Holocene epoch (Whitehouse et al. 1997). Another service provided by wetlands is the habitat they afford wildlife. In general, wetlands occur in areas that display attributes of both terrestrial and aquatic ecosystems. Wetland ecotones thus afford habitat for both terrestrial and aquatic flora and fauna, generating localized hotspots of productivity and biodiversity (Woodward & Wui 2001).

Despite the obvious importance of wetlands to humans and wildlife, they have been severely degraded by humans. Since the arrival of Europeans in the continental U.S., wetlands have been so severely altered that less than half of the original wetland area remains today (Dahl 2000). In fact, it is estimated that one third of threatened and endangered animals and plants in the United States live in wetlands (Murdock 1994; Woodward & Wui 2001).

The southeastern U.S. bears the brunt of this loss. Between the mid-1970s and the mid-1980s, nearly 90% of all wetland losses in the U.S. were in the ten most southeastern states (Hefner et al. 1994). Today, approximately 50% of remaining U.S. wetland ecosystems occur in this ten state region (Hefner et al. 1994). In these ten states, 95% of these remaining wetlands are freshwater, or Palustrine wetlands (Dahl 2000).

Within the southeast, the wetland ecosystems of the southern Appalachians are perhaps the rarest and most threatened. Weakley and Shafale (1994) suggest that only one sixth of the original 2000 hectares of southern Appalachian bogs and fens remain today. While the fate of southern Appalachian wetlands seems altogether bleak, humans are working to protect those wetlands that remain. Groups such as The Nature Conservancy (TNC), the North Carolina Natural Heritage Program (NCNHP), the Wilderness Society (TWS), and the U.S. Fish and Wildlife Service (USFWS) fund wetland research to assess the value of southern Appalachian wetlands to rare, threatened and endangered flora and fauna.

However, most of these habitats are privately owned, and this poses a major obstacle to wetland conservation in the southern Appalachians (Murdock 1994; Weakley & Shafale 1994). Landowners are either unaware of the value of these habitats to wildlife, unconcerned about their value, or are hesitant to cooperate with state and federal agencies (Murdock 1994; Lee & Norden 1996). Therefore, efforts to save privately owned wetlands in the southern Appalachians are seldom successful (Lee & Norden 1996).

Southern Appalachian wetland origins

The origins of southern Appalachian wetlands can be traced to the end of the Pleistocene and the beginning of the Holocene epochs (Weakley & Shafale 1994; Lee & Norden 1996). Approximately 18,000 ybp, the Wisconsinin glaciation was at its maximum. The Laurentide ice sheet, extending as far south as Pennsylvania and Ohio, contained much of the water that makes up the Atlantic ocean and North American freshwater systems today (Delcourt & Delcourt 1981). The end of the Wisconsinin was brought about by an increase in global temperatures that caused the Laurentide ice sheet to melt. For the next 13,000 years, melt water increased sea level and water tables to their present levels (Delcourt & Delcourt 1981). During this period, wetlands formed in the southern Appalachians, though to a lesser extent than in the flat areas of the southeastern coastal plains. Due to their relatively high topographic relief, the southern Appalachians are not conducive to wetland formation when compared to the rest of the southeast (Weakley & Shafale 1994). Therefore, wetlands that were formed in the southern Appalachians were few and small (Weakley & Shafale 1994).

Today, only a small proportion of existing southern Appalachian wetlands are pristine. Some pristine southern Appalachian wetlands have been cored for dating of peat and for palynological analysis. These few wetlands were found to exceed 10,000-12,000 years old (Shafer 1984, 1986; Murdock, personal communication), dating back to the periods of heavy glacial runoff that marked the Pleistocene glacial retreat (Delcourt & Delcourt 1981). While scientists have only attempted to date pristine wetlands, many less pristine wetlands are thought to be just as old. These less pristine wetlands, representing the majority of southern Appalachian wetlands, have all been altered by humans for livestock grazing or commercial and residential development (Murdock 1994). In their disturbed states, these wetlands are considered southern Appalachian wetland remnants, as they are thought to have once been much like the few pristine wetlands that occur today (Lee, personal communication). In fact, many of these remnant wetlands are so severely altered that they are only recognizable due to small patches of wetland flora such as rushes (*Juncus* spp.) and sedges (*Carex* spp.) (Lee, personal communication).

The organisms that inhabit southern Appalachian wetlands provide further evidence that these habitats are results of Pleistocene glacial retreat (Weakley & Shafale 1994). As temperatures have risen with the melting of the Wisconsin glacial mass, vegetational communities and habitat types have migrated northward in latitude and higher in elevation, "following" areas of suitable climate. Therefore, southern Appalachian wetlands have been left in a patchy distribution wherever topography and elevation allow persistent standing water and wetland vegetational communities. As a result, the flora and fauna that depend on fens and bogs have become isolated in these patchy wetlands, and today contribute to a high degree of regional endemism. Some of these disjunct populations of organisms are thought to have been in place thousands of years before the populations in their main northern or eastern North American ranges (Weakley & Shafale 1994). Therefore many southern Appalachian populations of wetland organisms are the living ancestors of the populations that occur in their main ranges. For example, the Mountain Sweet Pitcher Plant (*Sarracenia rubra* spp. *jonesii*) is a federally endangered southern Appalachian wetland endemic. The next closest populations of this species complex are primarily in the coastal plain. Gray's Lily (*Lilium grayii*) is also a southern Appalachian endemic that may be soon listed as endangered (Murdock 1994). Swamp Pink Lily (*Helonias bullata*) is yet another threatened species with a distribution that extends into the more northern states. Similar to the pitcher plant, the federally endangered Bog Turtle (*Clemmys muhlenbergii*) has relict populations in southern Appalachian wetlands that are isolated from much larger populations in the wetlands of the northeast (Lee & Norden 1996).

Southern Appalachian wetlands – agents of maintenance and the impacts of humans

Following their post-glacial origins, southern Appalachian wetlands were maintained by a variety of abiotic and biotic forces, many of which have since been altered by humans. One of the forces, for example, that helped maintain wetlands, was fire. Delcourt and Delcourt (1997) found evidence of Native American use of fire, primarily for agriculture, in the southern Appalachians that predates the arrival of Europeans. By using fire, Native Americans may have helped maintain southern Appalachian wetlands by preventing woody vegetation from becoming established (Lee & Norden 1996). Woody vegetation encroachment dries out wetlands through evapotranspiration (Weakley & Shafale 1994). Today, large scale fires are actively suppressed and when natural fires do occur they are quickly extinguished (Weakley & Shafale 1994). Thus, with the arrival of Europeans, southern Appalachian wetlands are thought to have suffered from the loss of both Native American set and natural wildfires.

Other factors that may have maintained southern Appalachian wetlands were the Pleistocene megafauna and other herbivores which have been driven extinct, or were regionally extirpated by humans. Some researchers suspect that large Pleistocene herbivores, such as the Giant Beaver (*Castoroides ohioensis*), Mastadons (*Mammut americanum*), Mammoths (*Mammuthus columbi* and *M. primigenius*), and Giant Ground Sloths (*Megalonyx jeffersonii*) once played a part in maintaining open areas where wetlands exist(ed), by browsing woody vegetation (Weigl & Knowles 1995; Lee & Norden 1996). These herbivores were likely driven extinct from overhunting approximately 10,000 years ago, upon the first significant human movement into eastern North America (Martin 1984). More recently, it is thought that grazing by American Elk (*Cervus canadensis*) and American Bison (*Bison bison*) contributed to maintaining open areas conducive to wetland persistence (Lee & Norden 1996, but see Ward 1990). Abundant Elk were finally hunted to extirpation in the late 1700s in western North Carolina and Bison were known from the eastern U.S. up until 1765 (Lee & Norden 1996).

Another contributor to southern Appalachian wetland maintenance is the American Beaver (*Castor canadensis*). Beaver were probably one of the most dominant faunal components of North America after the last glacial maximum, with an estimated 60 to 400 million individuals occupying riparian areas from northern Mexico to the North American tundra (Naiman et al. 1988). Indeed, some authors suggest that 60 million beaver once inhabited North Carolina alone (McGrath & Summer 1992). Beavers, in such great numbers, are thought to have maintained southern Appalachian wetland habitats because they are ecosystem engineers (Naiman et al. 1988; Lee & Norden 1996).

Ecosystem engineers are "organisms that directly or indirectly modulate the availability of resources to other species, by causing physical state changes in biotic or abiotic materials" (Lawton & Jones 1995, p. 141). Through their engineering, beaver cause profound changes in the small area that surrounds their dam(s) (Johnston & Naiman 1990). Changes brought about by beaver engineering can create more hydric habitats for flora and fauna (Lawton & Jones 1995). More specifically, beaver can create conditions favorable for wetlands (Pollock et al. 1995).

When beavers construct a dam, the local hydrology is changed. Initially the area is flooded because the dam obstructs the water channel. Also, the water that is no longer taken up by trees, either flows down slope into the impoundment or percolates into the soil, raising the water table (Lawton & Jones 1995; Outwater 1996). The sediment and nutrient runoff from the upslope forest floor accumulates in the low-lying impoundment (Novak 1987; Outwater 1996).

This nutrient-rich impoundment produces a cascade effect in community structure as the habitat becomes more hydric. Beaver impoundments often increase local species richness of plants and intensify vegetation structure as wetland-associated species colonize the area (Snodgrass 1997). This increase in plant species composition and vegetational structure can, in turn, lead to an increase in faunal diversity; mammals and birds often use this vegetation for food and nesting cover (Lochmiller 1979; Medin & Clary 1991; Lawton & Jones 1995).

However, the increased complexity of beaver pond communities is not just a simple by-product of more species of plants and animals. The engineering by beaver shifts local communities creating higher productivity throughout the local food web. More phytoplankton, zooplankton, bacteria, fungi, and algae thrive in beaver impoundments than in non-beaver impacted areas (Outwater 1996). This increased productivity of microorganisms provides more food for aquatic invertebrates, many of which are early instars of insects. In fact, invertebrate biomass has been shown to be five times higher in beaver ponds than in non-beaver riparian areas (McDowell & Naiman 1986). Higher insect abundance is thought to lead to higher species richness and/or abundance of insectivorous mammals and birds in beaver-impacted habitats (Reese & Hair 1976; Medin & Clary 1991; Grover & Baldassarre 1995; Edwards & Otis 1999).

Superimposed on the structural modifications induced by beaver is a temporal pattern of successional change. Beaver create dramatic seral changes in habitats. When a beaver colony inhabits an area long enough to exhaust their food resources, they move on in search of more trees (Novak 1987). The abandoned beaver dam eventually breaks and the impoundment drains, exposing rich underlying sediments (Snodgrass 1997). Early successional grasses and forbs will then colonize the open sunny area, and "beaver meadows" result (Snodgrass 1997). Shrubs and saplings of riparian trees then may invade the area if succession continues (Snodgrass 1997), leading, eventually, to the return of a mature forest.

Successional change in beaver meadows, however, often truncates before forests again dominate (Snodgrass 1997). Beaver are known to recolonize formerly abandoned impoundments and re-build dams once woody vegetation (for forage) has returned. When beaver re-flood the area, succession begins again (Snodgrass 1997). After many successive beaver generations have colonized, abandoned, and re-colonized an area, the resulting wetland can become a stable fixture in the landscape (Webster et al. 1975). In fact, some beaver-created and maintained wetlands have been known to last for centuries (Webster et al. 1975; Jones & Lawton 1995). Indeed, beaver may have maintained many southern Appalachian wetlands for millennia (Weakley & Shafale 1994; Lee & Norden1996).

Upon the arrival of Europeans, maintenance of wetlands by beaver was abruptly halted. Between the 1600s and the 1800s, beaver were nearly extirpated from eastern North America by the burgeoning fur trade (Naiman et al. 1988; Lee & Norden 1996). As a result, by the 1900s, beaver trapping throughout North America virtually ceased (McGrath & Summer 1992). Overall, it is estimated that beaver populations were reduced from 200 million to 10 million (Outwater 1996). Consequently, this over-harvesting of beaver for the fur trade is partially to blame for southern Appalachian wetland losses in the past 400 years (Lee & Norden 1996).

Today, beaver populations are relatively safe from fur trappers on federal and state lands as laws regulate trapping on public lands (Naiman et al. 1988; Outwater 1996). However, beaver currently face a new threat: private landowners. Private landowners often trap beaver, not to sell their pelts but because they are considered nuisances; beaver often damage private property by flooding and felling trees. In North Carolina, landowners can legally kill beaver at any time if they are harming private property (NCWRC 2002). Indeed, beavers are usually trapped and killed very soon after they are noticed damming streams on privately owned land (Cherry, personal communication). As a result, most wetlands that are currently maintained by beavers occur on public lands, where the beavers are safe from disgruntled landowners.

Humans have also destroyed southern Appalachian wetlands with cattle grazing, a severe present-day threat to these ecosystems (Weakley & Shafale 1994). Southern Appalachian wetlands occupy some of the few, rare, small areas of flat ground in this region. Unfortunately, these flat parcels are optimal areas to pasture cattle, as it is usually easier to produce an open area for grazing through ditching a wetland (Buhlman et al. 1997; see Figure 30 p51 in Dahl 2000) than by clearing a forested slope. Indeed, the majority of wetlands remaining in the southern Appalachians have been converted to pasture or manipulated for other agricultural uses (Weakley & Shafale 1994).

Grazing can degrade even those wetlands that are not drained by way of its impacts upstream from wetlands. Cattle destroy stream banks, leading to increased erosion and sedimentation (Buhlmann et al. 1997). Indeed, the impact of cattle surpasses the disturbance intensities of all other agricultural activities in the southern Appalachians (Lenat 1998). Cattle also trample and eat some wetland flora, thereby destroying the habitats and resources upon which many southern Appalachian wetland organisms depend (Murdock 1994). Additionally, the nutrient deposition from cow dung alters wetland chemistry and kills sphagnum, the dominant floral component of many of these habitats (Murdock 1994; Weakley & Shafale 1994). The negative effects of sphagnum loss are often irreversible. As sphagnum dies, the water retention capacity of the wetland is severely decreased (Weakley & Shafale 1994), and the wetland may dry up.

Just as southern Appalachian wetlands are attractive to cattle farmers, they are likewise attractive to developers. Southern Appalachian wetlands are often drained for development; many have been converted into housing developments, golf courses and commercial properties (Weakley & Shafale 1994). Additionally, development upslope from wetlands can be detrimental. The forests that historically bordered southern Appalachian wetlands have mostly been cleared and the land altered for other uses (Weakley & Shafale 1994). Upslope runoff is thus faster on the more erosible slopes, resulting in a rapid introduction of greater sediment loads into small low-lying Appalachian wetlands (Weakley & Shafale 1994). Increased sedimentation may simply fill in a wetland, or, as with Appalachian streams, degrade macroinvertebrate communities in wetlands (Sidebottom 2000).

II. Neotropical Migratory Birds: taxa in peril-background

Neotropical migratory birds (NTMBs) spend at least part of their lives in the Neotropics (Rappole 1995). More specifically, NTMBs breed in the temperate regions north of the Tropic of Cancer and spend their non-breeding seasons south of this latitude, undertaking two annual migrations to do so (Rappole 1995). The majority of landbird NTMBs that breed in temperate North America belong to the families Parulidae (wood warblers), Thraupidae (tanagers), Icteridae (orioles and blackbirds), Tyrannidae (flycatchers), and Vireonidae (vireos) (Rappole 1995).

NTMBs dominate the avifauna of much of the Appalachians, where at least 75 species breed (Simons et al. 1999; Simons et al. 2000). In parts of the southern Appalachians, Neotropical migrants make up 80% of the breeding bird community (MacArthur 1972). NTMBs are abundant in the southern Appalachians because this region spans a broad elevational gradient within a relatively small geographic area (Simpson 1992). This elevational gradient generates many different temperature and moisture regimes, resulting in high plant diversity and, ultimately, a variety of niches to be exploited by NTMB species (Odum 1950; Simpson 1992; Simons et al. 2000)

Population declines in Neotropical migratory species

Nearly 71% of Neotropical migratory bird species have declined since 1978 (Robbins et al. 1989). NTMBs experiencing the greatest declines are area-sensitive species that breed in early successional or late successional habitats. These Neotropical bird species are in decline for two basic reasons: the loss of wintering habitats in the tropics and the loss of habitat in their North American breeding ranges (reviewed in Askins et al. 1990). Early and late successional breeding NTMBs show specific responses to habitat loss in their temperate breeding ranges relative to the types of breeding habitats they prefer.

Early successional species declines

Of the 128 early successional NTMBs that breed in North America, 76% are currently in decline (Hunter et al. 2001; Thompson & DeGraaf 2001). Furthermore, the southern Appalachian region is among the regions experiencing the greatest declines in early successional species, due primarily to losses in suitable breeding habitats (Franzreb

& Rosenberg 1997). Although there are different types of habitats classified as early successional, many of them share the common characteristic of being dependent on disturbances.

Indeed, Hunter et al. (2001) combine several types of early successional habitats into one category called "disturbance-dependent" habitats. These habitats are considered disturbance-dependent because they have an open canopy and are historically maintained by some form(s) of frequent disturbance, such as natural wildfire, native herbivores, or extensive beaver activity (Hunter et al. 2001). Disturbance-dependent habitats include habitats referred to as grasslands or prairies, scrub-shrub habitats, open woodlands and small forest openings (Hunter et al. 2001). Furthermore, each of these habitat types has a distinctive suite of bird species, many of which are declining alongside the habitats themselves (Hunter et al. 2001).

Grasslands, or prairie communities, are the scarcest of all early successional habitats in the eastern U.S., reduced to less than 1% of the area covered in the east prior to the arrival of Europeans (Noss et al. 1995; Hunter et al. 2001). As a result, 70% of grassland bird species in the U.S. are in decline (Hunter et al. 2001). In fact, over half of the federally endangered disturbance-dependent bird species in eastern North America breed in grasslands (Hunter et al. 2001). Both scrub-shrub habitats and open-canopy woodland habitats have also declined greatly; not surprisingly, 70% of bird species that require one of these early successional habitats are also declining (Hunter et al. 2001). Lastly, small forest openings are early successional habitats that result from small disturbances such as tree fall gaps, fire, beaver, or human impacts within a forest. Of the species associated with small forest openings in the eastern U.S., 45% are in decline (Hunter et al. 2001).

There are several contributors to losses in these types of early successional habitats, all of which were previously discussed in terms of southern Appalachian wetland losses. Wildfire, native herbivores, and beaver are three agents of large scale early successional habitat maintenance that are not as prevalent today as before human activity in North America. Human activity has also directly affected the amount of suitable habitats available to early successional bird species via land use practices (Askins 2001). Shortly after their arrival, European settlers created an abundance of early successional habitats for up to three hundred years via forest clearing (Hunter et al. 2001) in order to make land suitable for agriculture (Outwater 1996; Askins 2001; Trani et al. 2001). These large scale human disturbances may have increased the amount of early successional habitat in the NTMB breeding zone (Askins 2001), causing a subsequent expansion of many early successional species' ranges (Hunter et al. 2001). However, since the early 1900s, less forested area has been cleared, farm abandonment has steadily increased and early successional bird species have therefore declined (Hunter et al. 2001; Trani et al. 2001).

Today, some speculate that humans may be offsetting these historic losses in early successional habitats by clearcut timber harvesting (Thompson & DeGraaf 2001). However, the open habitats that remain after forests are razed are not true early successional habitats (Lorimer 2001). True early successional habitats are more stable, composed of pioneer species of grasses, vines and shrubs, and do not shift in late successional stages rapidly (Lorimer 2001). To the contrary, the early successional habitats that result from clearcuts, referred to as "young forests", are very ephemeral. Instead of pioneer species, saplings of the late successional tree species that were previously harvested dominate these habitats, and quickly mature to form a closed canopy forest (Lorimer 2001). Therefore, an early successional state only occurs briefly following a clearcut, and for this reason, clearcuts do not replace the early successional habitats that have been historically lost.

Late successional species declines

Across all regions of North America, an average of 26.6% of forest-breeding species are experiencing declines (Franzreb & Rosenberg 1997). The southern Appalachian region contains the Great Smoky Mountains National Park, which constitutes approximately 80% of the primary forests in eastern North America, and contains the greatest area of relatively undisturbed forests in the eastern U.S. (Davis 1993). Therefore, one may assume that forest-breeding bird populations would be stable or even increasing in this region. However, this is not the case. Relative to all of North America, the southern Appalachians display an even greater percentage (42.1%) of declining forest-breeding bird species (Franzreb & Rosenberg 1997). Late successional NTMB species are thought to be declining because humans are fragmenting forests for agricultural, residential and commercial development (Robbins et al. 1989; Askins et al. 1990; Faaborg et al. 1995). NTMB abundance has been found to be lower and their declines more severe in fragmented areas due to smaller habitat patch sizes (Freemark &

Collins 1992). In the southern Appalachians, area-sensitive NTMB declines are among the most severe in the U.S. (Franzreb & Rosenberg 1997).

In general, forest fragmentation in the NTMB breeding zone decreases both the quantity of suitable breeding habitat and the quality of the remaining patches (Faaborg et al. 1995). More specifically, NTMBs respond unfavorably to fragmentation of their breeding habitats for four major reasons: nest parasitism, nest predation, limited food supply in the small forest habitats that result from fragmentation and competition with non-area-sensitive bird species (Askins et al. 1990).

Forest-breeding NTMBs suffer greatly from brood parasitism, by the Brown-headed Cowbird (*Molothrus ater*) (BHCO) (Brittingham & Temple 1983). Since the early 1900s, the range of BHCO has been expanding eastward and their populations have been increasing, presumably due to forest habitat fragmentation (Brittingham & Temple 1983). Cowbirds prefer to breed in forest edge habitats that often result when forests tracts are razed (Ortega 1998). Research has shown that BHCO brood parasitism occurs more often in host nests that are located closer to a forest edge (Brittingham & Temple 1983). Furthermore, higher numbers of cowbird eggs have been found in nests closer to the forest edge (Brittingham & Temple 1983).

Birds that naturally breed in fragmented habitats have evolved with Brown-headed Cowbirds (Ortega 1998). These species typically eject cowbird eggs or abandon their parasitized nest to rebuild (Ortega1998). In contrast, late successional Neotropical migrant species have little or no evolutionary experience with Brown-headed Cowbirds (Wilcove 1985; Ortega 1998). Consequently, most NTMBs do not recognize parasite eggs in their nests, are not equipped with ejection or abandonment defenses, and thus are frequently parasitized by Brown-headed Cowbirds (Ortega 1998). Cowbird nestlings outcompete host nestlings and are raised by hosts at the expense of their own young's survival (Ortega 1998). This is especially detrimental to the reproductive success of NTMB hosts because most species only lay one, small brood per year (Robbins et al. 1989). Brittingham and Temple (1983) conclude that there has been a negative correlation between BHCO abundance and NTMB abundance in recent years, and this may be due to the negative impacts of cowbirds on NTMB species.

In fragmented forests, NTMBs also exhibit lower reproductive success due to higher rates of nest predation (Askins et al. 1990). Bird nests are often more densely distributed along forest edges than within the forest tracts (Gates & Gysel 1978), and thus attract more predators (Askins et al. 1990). Studies of natural and artificial nests have shown that nests closer to forest edges and nests in smaller patches of forest habitat suffer increased nest predation (Forsyth & Smith 1973; Wilcove 1985).

Habitat fragmentation may also be disadvantageous for NTMBs because of decreased food supply and increased competition for these decreased resources. It has been empirically shown that NTMB species may suffer from lower food abundance in small forest tracts (Zannette et al. 2000). In a study of the Eastern Yellow Robin (*Eopsaltria australis*), an area-sensitive species in Australia, food was found to be less abundant in small forested habitats where this species was nesting. As a result, the mother was forced to leave the nest more often in order to acquire food, leaving her nest and offspring vulnerable to brood parasites and predation (Zannette et al. 2000).

Furthermore, in small forest habitats, NTMBs must compete with area-insensitive species that are naturally abundant and more densely distributed in small habitats (Askins et al. 1990). Studies have suggested that NTMBs are at a disadvantage in competition with area-insensitive species (Askins et al. 1990). This is because area-insensitive species are better adapted for small habitats, and are thus better competitors for scarce resources than area-sensitive NTMBs (Askins et al. 1990).

III. The potential importance of a declining habitat to declining avian taxa

Perhaps not surprisingly, declines of early and late successional bird species in the southern Appalachians have occurred simultaneously with the loss of southern Appalachian wetlands (Hunter et al. 1999). This suggests that southern Appalachian wetland losses may be partially to blame for southern Appalachian bird declines. However, no study has systematically examined the importance wetlands to breeding avifauna throughout the southern Appalachians, and the value of these wetlands is poorly understood (Hunter et al. 1999).

Alhough no study has systematically examined the importance of southern Appalachian wetland habitats to breeding birds, two descriptive accounts imply that such habitats may be regionally important to birds. Rossell et at. (1995) censused birds at Tulula Bog (Graham Co., NC) during the breeding and non-breeding seasons of 1994, in a multi-taxa wetland survey of this site. Seventy-four bird species were recorded, many of which are species of songbirds that are in decline (Rossell et al. 1995). Bird surveys have also been conducted in three wetlands in Shady Valley, Tennessee for over five decades (Coffey & Shumate 1999). These surveys have shown that wetlands in this small southern Appalachian valley contribute greatly to bird diversity in the area (Coffey & Shumate 1999). Both of these pioneering surveys suggest that southern Appalachian wetlands may be a critical resource for more than just pitcher plants and bog turtles. Most importantly, both studies highlight the need for a rigorous hypothesis-based study of avian use of southern Appalachian wetlands.

Such a study would better explain the importance of these habitats to birds; it would not only ask if birds use wetlands, but it would also address the specific factors of certain southern Appalachian wetlands that are most important to birds. Indeed, all southern Appalachian wetlands are not alike. In short, these habitats display a chronological cross section of seral stages. They may be open early successional wetlands dominated by sedges (*Carex* spp.), rushes (*Juncus* spp.), and/or a variety of grasses and shrubs. They may be mid-successional meadows, characterized by thickets of sub-canopy level shrub cover. Or, they may be late successional wetlands beneath a closed canopy forest, dominated by hardwoods and/or evergreens.

For this reason, different wetland types or different characteristics of southern Appalachian wetlands may appeal to different suites of breeding birds, depending upon the habitats they prefer for breeding. With the current paucity of early successional habitat throughout North America, early successional southern Appalachian wetlands may provide valuable habitats for early successional birds (Hunter et al. 1999). Early successional southern Appalachian wetlands may provide habitat for open grassland or scrub-shrub bird species. Wetlands occurring as small patches in forest tracts (e.g. those created by beaver) may provide suitable habitat for early successional canopy gap species. These early successional wetlands may also provide highly productive habitat for forest-breeding NTMBs. Pagen et al. (2000) found that adult and fledgling forest interior NTMBs forage in early successional habitats during both their breeding and post-breeding periods because early successional habitats may be more productive than the late successional forests used for breeding. Forest-breeding birds may also use late successional southern Appalachian wetlands, as they occur under a closed forest canopy. Many late successional southern Appalachian wetlands, especially those that are created and maintained by beaver, may provide highly productive forest habitats for forest-breeding NTMBs due to the abundance of resources often associated with beaver activity.

Overall, southern Appalachian wetlands occur in a region known for high avian species diversity and species richness. Unfortunately, many of the species that constitute a large proportion of this regional avifauna are experiencing significant population declines. For this reason a better understanding of the importance of these habitats to the whole bird community, as well as to birds that prefer certain habitat types would be of value. Altogether, if southern Appalachian wetland habitats are important to breeding birds, certain characteristics of these habitats may predict high community-level species diversity, species richness, and total avian abundance. More specifically, birds that prefer different breeding habitats (i.e. early successional and late successional species) or that exhibit different migration strategies (i.e. Neotropical species, short-distance migrants and permanent residents) may use some wetland types more than others based on their different life history characteristics. In this study, I seek a better understanding of the potential importance of wetland habitats to birds at the community and guild levels. This knowledge may aid in developing management plans for these rare habitats and the birds that use them.

METHODS

I. Description of study sites

I collected data at 57 southern Appalachian wetlands totaling 794.63 hectares in western North Carolina (n=44), northeast Tennessee (n=3), and southwest Virginia (n=10). Study site elevations ranged from 442-m to 1,254-m and site areas ranged from ~0.40- to ~94.7-hectares. Wetlands were owned by Appalachian State University (ASU) (n=2), the Blue Ridge Parkway (BRP) (n=22), the Nature Conservancy (TNC) (n=6), the N.C. Department of Transportation (NCDOT) (n=2), the U.S. Forest Service (USFS) (n=3), and private landowners (n=22).

All study sites were classified using the Cowardin Classification (Cowardin et al. 1979) as Palustrine wetlands. Palustrine wetlands include freshwater wetlands that are often referred to as swamps, marshes, bogs, fens or prairies (Cowardin et al. 1979). In the southern Appalachians, Palustrine wetlands are typically found on lake edges, in river channels, or in river floodplains (Cowardin et al. 1979).

The Palustrine wetland category can be further divided into eight classes. All study wetlands belonged to only three of these classes (Cowardin et al. 1979): (1) the persistent-emergent class, (2) the scrub-shrub class, and (3) the forested class. These classes are designated based upon the uppermost structural vegetation level that occupies at least 30% of the wetland. Persistent-emergent wetlands are dominated by at least 30% cover of emergent vegetation (defined as vegetation with roots below the water surface) that remains standing for the majority of the year. Scrub-shrub wetlands have an uppermost vegetation sub-canopy level of shrubs covering at least 30% of the site. Forested wetlands have dominant canopy level vegetation (defined as woody vegetation at least 5m in height) comprising at least 30% of the wetland vegetation cover. I assigned each site to one of these three Palustrine classes by visually estimating the uppermost vegetation level that covered at least 30% of the wetland. Twenty-three study sites belonged to the persistent-emergent class, 21 to the scrub-shrub class, and 13 to the forested wetland class.

In addition to differences based on Palustrine class, sites also varied with respect to human land use in and around the wetlands. As a general rule, almost all privately owned sites had been subjected to some form of human disturbance. Such disturbances resulted from the impacts of logging, draining for agriculture, grazing, mowing, residential development, and/or commercial development. Sites not privately owned, hereafter operationally referred to as protected sites, were generally less vulnerable to these land use impacts. However, while Blue Ridge Parkway sites were considered protected, they often displayed the effects of disturbance similar to privately owned sites. Presumably due to their small sizes, BRP sites were often degraded by human disturbances on adjacent privately owned land. Moreover, because of the Parkway's mandate to preserve the human as well as the natural history of the mountains, the BRP's wetlands are often grazed to maintain these areas as the pasture they were historically.

Another major way in which the 57 sites varied was based on differences in their edges. Edges associated with wetlands were considered as either abrupt or gradual. An abrupt edge was defined operationally as a distinct change in vegetation structure between two vegetation types (Suarez et al. 1997). Abrupt edges in sites (n=29) usually resulted from persistent human disturbances. Beaver, a natural agent of disturbance, also created abrupt edges in study sites. Immediately after a beaver dam floods an area, there may be no transitional vegetation between the pond and canopy level vegetation (Snodgrass 1997). However, beaver-created edges become less abrupt as emergent vegetation grows, shrubs invade and succession takes place (Snodgrass 1997), unlike the abrupt edges that persist due to active human maintenance.

A comparable number of sites (n=28) were classified as having gradual edges, defined operationally as a gradient from one vegetation type to the next. The cross section of a gradual edge was often a successional ecotone; i.e., where the vegetation shifted fluidly from groundcover to shrub, then from shrubs to saplings, and then from saplings to forest (Suarez et al. 1997). Most of the beaver–impacted wetlands in this study had gradual edges, primarily because succession had been given sufficient time since beaver invasion to turn the abrupt edges between beaver ponds and forests into the transitional edges between beaver meadows and forest.

II. Avian Censusing

In the 1999 field season, I conducted a pilot study comparing spot mapping and 50-m fixed-radius point counts to determine the most suitable and efficient avian census technique for this project. Two of the pilot study sites were located in Avery County, NC and three in Watauga County, NC. I conducted eight point counts and eight spot-map censuses at each site between May 15 and August 1, 1999. Censuses were conducted between sunrise and 1000 when there was neither rain nor wind to potentially hinder my detection of singing birds.

Results of the two census techniques were compared in the five pilot study sites. For each study site, I graphically plotted the number of species detected with each additional minute of census time with point counts and with spot-mapping. Upon comparison of the graphs, I found that more species were consistently detected in a smaller amount of time using point counts, relative to using the spot-map censusing technique. Hence, these pilot data analyses indicated that fixed-radius point counts were superior for the objectives of this study. This technique enabled detection of more species in a smaller amount of time, so that I could spend less time at each site. By spending less time at each site, I could survey additional study sites, thereby increasing my sample size in the following field seasons. Other researchers have reported this advantage of point counts over spot-maps as well (e.g. Ralph et al. 1995).

Thus, in the field seasons of 2000 and 2001, I used 50-m fixed-radius point counts to census 57 wetland sites throughout the mountains of North Carolina, Tennessee, and Virginia. Following the suggestions of Ralph et al. (1997), three point counts were conducted per site between May 15 and June 30 of either 2000 or 2001; 32 sites were censused in 2000 and 25 in 2001. Counts were conducted from sunrise to 1000 on days when there was neither precipitation nor wind to interfere with my detection of birds.

On each of three visits to every site, I conducted one, ten-minute 50-m fixed-radius point count from the center of the core wetland area (Ralph et al. 1995). One point count was placed in each site, regardless of wetland size, in order to avoid biasing my results due to "passive sampling" (Johnson 2001). Passive sampling occurs when larger sites contain more census plots than smaller sites, leading to the incidental detection of more species (Johnson 2001). During each point count, I divided species detections into 1-3 minute, 3-5 minute, and 5-10 minute intervals. Bird detections were also divided spatially into <25.0-m, 25.1-50.0-m, and >50.0-m distances. All birds seen or heard within the ten-minute count were recorded (Ralph et al. 1995). Flyovers were also noted separately but not used in analyses. Only I conducted the sampling. I was very careful to avoid counting the same bird twice if it moved during the ten-minute point count. Therefore, I am confident that no birds were double counted and that my point counts accurately sampled birds in these wetlands.

The number of species and individuals recorded at point counts were used to calculate dependent variables for statistical analyses. Dependent variables were calculated at two levels, the community-level and the guild-level (Table 1).

Table 1. Description of dependent variables and how they were measured

Dependent Variable	*Method of Measurement*
Community-level	
Species Diversity	Calculated Shannon Index of diversity (H') (Cox 1996) values for each site using the mean numbers of individuals of each species observed across all three visits to each site.
Species Richness	Maximum number of species detected in each wetland across all visits.
Total Avian Abundance	Totaled the number of individual birds detected across all three visits to each site. This total was then divided by the number of visits (3), to obtain the average number of individuals observed on any visit in each wetland.
Guild-level	
Migration Status Permanent residents Short-distance migrants Neotropical migrants *Breeding Habitat* Grassland species Wetland-open-water species Scrub-shrub species Woodland species Urban-breeding species *Nesting substrate level* Ground-low-nesting species Mid-upper-nesting species Cavity-nesting species Woodpeckers Insectivores	For each guild, the maximum number of species detected in each wetland across all visits was used to calculate within-guild species richness. (e.g. I summed the number of permanent residents species across all 3 visits to a site for this guild's species richness at that site.) To calculate within-guild abundances, the number of individuals belonging to each guild, regardless of species, in each site was totaled. This abundance for each guild in each wetland was divided by the number of visits (3) to obtain the average number of individuals belonging to each guild that were detected during any visit to each site.

At the community-level, avian species diversity, species richness, and total avian abundance were calculated for every site. The Shannon index (H') of species diversity was used to assess the diversity of species with breeding territories including the wetland area (Table 1) (Cox 1996). As a conservative measure, I considered only those species present on all three point counts to have been breeding at that site. By restricting my analyses of H' to these common species, I avoided the inflation of species diversity values that can occur when rare species are included (Cox 1996). Species richness, in contrast, was used to assess *all* species that may be using a site. Aside from breeding per se, birds may have been using wetlands for foraging (Pagen et al. 2000) or for forays in search of extraterritorial copulations (Norris & Stutchbury 2001). Species richness was therefore calculated as the number of species detected in at least one of the three visits to each site, and thus includes rare species (Cox 1996) (Table 1). I used total avian abundance to assess how many birds, regardless of species, may be using each site on an average visit (Table 1).

The importance of wetlands to birds was also analyzed at the guild-level. I employed the same classifications as the Breeding Bird Survey (Sauer et al. 2001) in assigning bird species to guilds. All species detected were grouped in guilds based on migration status (permanent resident species, short-distance migrant species, or Neotropical migrant species), breeding habitat (grassland species, wetland-open-water species, scrub species, woodland species, or urban species), and nesting substrate level (ground-low-nesting species, mid-upper-nesting species or cavity-nesting species). Ground-shrub-nesting and midstory-canopy-nesting species are those that build their nests on the ground or in shrubs, and those that nest in midstory or canopy vegetation, respectively. Additionally, I analyzed woodpeckers (Family Picidae) in a grouping by themselves because woodpeckers have been shown to be good indicators of overall bird diversity in a habitat (Mikusinski et al. 2001). I also analyzed the diet-based guild of insectivorous species (Ehrlich et al. 1988), because others have found that insectivores may benefit from the high insect productivity of wetlands (Reese & Hair 1976, Medin &Clary 1990). Species richness and abundance were calculated for each of these 13 guilds, and were then used as dependent variables in the statistical analyses (see below).

III. Vegetation Analyses:

At each site, I recorded wetland class (Cowardin et al. 1979), presence or absence of grazers (cows or other livestock), edge type, presence or absence of beaver evidence, and whether or not the site was protected. These observations provided categorical variables that were employed as independent variables in statistical analyses (Table 2).

I also took quantitative vegetation data at each site. Vegetation sampling was conducted in the breeding season of the same year that the site was also censused for birds. My sampling protocol followed those suggested by Hamel et al. (1996). At each wetland, all site attributes and vegetation data were collected from the 11.28m radius circle surrounding the point from which birds were censused (James & Shugart 1970) (Table 2).

Table 2. Description of independent variable measurement methods.

Independent Variables	*Method of Measurement*
Wetland class	Every wetland was classified as persistent emergent, scrub-shrub or forested Palustrine wetland according to the guidelines of the Cowardin Classification of Wetland and Deepwater Habitats (Cowardin et al. 1979).
Grazing status	Presence or absence of livestock within the wetland proper was recorded at each site.

Table 2. continued

Edge type	In every site, the edge nearest the point count center was classified as gradual or abrupt (Suarez et al. 1997).
Beaver evidence	Evidence of recent beaver activity was recorded at each site on a presence/absence basis. Beaver evidence included actively maintained dams, freshly felled trees, or recently gnawed stumps.
Protected status	All sites that were owned by ASU, BRP, TNC, NCDOT, or USFS were considered protected. Privately owned sites were considered unprotected.
BRP status	Sites were categorized as being BRP- or non-BRP-owned sites.
Size of wetland	Wetland size was usually obtained from the entities that owned the sites (i.e. ASU, BRP, TNC, NCDOT, and USFS). For privately owned sites, I estimated wetland size visually and then refined my estimates using 1:24,000 USGS topographic maps. All wetland sizes were estimated to the nearest 0.1-hectare.
Percent cover of water	When present, percent cover of water was visually estimated using the ocular tube method described by Hamel et al. (1996). At distances of 2-m, 4-m, 6-m, 8-m, and 10-m from point center in each of the cardinal directions, I looked downward through a 2.0" by 2.0" tube. Presence or absence of water in the center of the ocular tube at each of these distances was recorded. This was then used to calculate the percent water cover.
Snag stem density	Snags greater than 10-cm dbh were counted within each 11.28- m vegetation plot.
Stem density	All trees greater than 10-cm dbh within the 11.28-m plot were counted to obtain an estimate of site stem density.
Basal area (cm^2)	The diameter of all trees exceeding 10-cm dbh within the 11.28-m plot were measured using a Biltmore stick. Diameter measurements were used to estimate site basal areas (Hamel et al. 1996).
Canopy closure	Canopy closure was calculated using the ocular tube method described by Hamel et al. (1996). At distances of 2-m, 4-m, 6-m, 8-m, and 10-m from point center in each of the cardinal directions, I looked upward through a 2.0" by 2.0" tube. Presence or absence of canopy vegetation in the center of the ocular tube at each of these distances was recorded. This was then used to estimate the percent canopy cover.

Table 2. continued

Midstory cover	Percent cover of total midstory vegetation was estimated via ocular tube readings (Hamel et al. (1996) within the 11.28-m radius vegetation plot.
Shrub cover	Percent cover of shrub vegetation was estimated in the 11.28-m plot using ocular tube readings (Hamel et al. 1996).
Groundcover	Percent cover of total groundcover was calculated using the ocular tube method of Hamel et al. (1996).
Forb cover	Percent cover of forb vegetation was estimated with the ocular tube technique throughout the 11.28-m plot.
Grass cover	Percent cover of grass vegetation was estimated visually in the 11.28-m plot with the ocular tube technique (Hamel et al. 1996).
Vegetation Profile	A vegetation profile board was constructed and used to assess the horizontal density of vegetation (Hamel et al. 1996). This method incorporates a 20" by 20" profile board that is divided into a grid of 25 equally sized squares. The board was placed approximately 2m from the ground and 10-m from the point center. The number of squares fully visible at 0-m, 2.5-m, 5-m, and 7-m from point center, in each of the cardinal directions was recorded. The total number of squares fully visible in all four directions was subtracted from the total possible squares so that the difference represented squares covered by vegetation. This number of obstructed squares was then divided by the total number of potentially visible squares in all four directions (400), and the product represented the percentage of the horizontal vegetation density in the wetland.

IV. Data Analysis

A stepwise multiple linear regression (SMLR) was used for among-site analyses of wetland use by breeding birds at the community and guild-levels using SAS Version 8.1 (SAS Institute 2000). Since species diversity, species richness, and total avian abundance are all continuous variables, a SMLR is the most suitable technique for analysis of these variables (Zar 1999). Thus SMLR was used to indicate if any of these community or guild-level dependent variables (Table 1) were significantly predicted by any independent habitat variables (Table 2).

Preliminary Analyses

Prior to final statistical analyses, all dependent and independent variable distributions were checked for outliers with SAS Version 8.1 (SAS Institute 2000). To search for outlying data points, I constructed box plots and normal probability plots for every variable. I detected outliers in the independent variables of site size (n=5), percent cover of site by water (n=3), site basal area (n=1), percent cover of midstory level vegetation (n=1), and percent cover of emergent vegetation (n=7). For each of these variables, I re-evaluated the raw data to make sure that outliers were not due to data entry error. No outliers appeared to have occurred due to a mistake in data entry. The outlying values were therefore retained for final regression analyses because they are thought to represent meaningful variation in characteristics of study sites.

In addition to preliminary outlier analyses, I also checked each dependent and independent variable for normality using residual scatterplots (Tabachnick & Fidell 1983; Zar 1999). Residual scatterplots were obtained by running preliminary multiple regression models for every dependent variable using all raw independent variables (SAS 2000). If the residual scatterplot for a given dependent variable was normal, I considered the regression model for that particular dependent variable valid. When the model for a dependent variable exhibited non-normal residual scatterplots, I examined the normal probability plots, box plots, and histograms of every variable in that model. Any non-normal independent variables that were significant predictors of the dependent variable in the model were transformed (Tabachnick & Fidell 1983). I then ran the model again with the transformed variables included and re-analyzed the residual scatterplots.

In two cases, transforming the independent variables resulted in normality. Percent cover of upperstory vegetation was not normally distributed in the regression model for short-distance migrant species richness. This variable was normalized with an arc-sin transformation (Tabachnick & Fidell 1983). Also, stem density displayed a non-normal residual distribution in the permanent resident species abundance model. Stem density was normalized with a square root transformation in this model (Tabachnick & Fidell 1983).

There were two non-normal dependent variables that could not be normalized through data transformation: species diversity and grassland species richness. For analysis of species diversity, I used a non-parametric Mann-Whitney test with a continuity correction (Zar 1999). This test was suitable because the only independent variable that entered the regression model as a significant predictor of higher species diversity was dichotomous (Zar 1999). My inability to normalize grassland-breeding species richness through transformation was probably because very few sites (n=6) contained any grassland-breeding species. In fact, in all six cases, only one grassland species, the Eastern Meadowlark (*Sturnella magna*), was observed. For this reason, the grassland-breeding guild was not analyzed.

RESULTS

During the breeding seasons of 1999, 2000 and 2001, a total of 171 point counts were conducted in the 57 study wetlands. A total of 2,266 individual birds, representing 92 species, were observed (see Appendix for species observed).

I. Community-level analyses

In terms of species diversity (H’), only the independent variable (IV) edge type entered the Mann-Whitney model as a significant predictor. Greater species diversity was found in sites associated with a gradual edge than in sites with an abrupt edge ($p<0.0002$, $z=3.520$) (Table 3).

Greater avian species richness was also significantly correlated with presence of a gradual edge ($p<0.0001$, $R^2=0.2995$) (Table 3). Two additional IVs entered the SMLR model for avian species richness. Beaver impact was positively correlated with higher site species richness ($p=0.0024$, $R^2=0.1108$) (Table 3). Also, sites that were unprotected tended to contain more bird species than protected sites ($p=0.0030$, $R^2=-0.0909$) (Table 3). Collectively, a site’s association with a gradual edge, evidence of beaver, and unprotected site status explained approximately 50% of the total variation in avian species richness among study sites ($p=0.0030$, $R^2=0.5012$).

Presence of beavers was the first IV to enter the SMLR model for total avian abundance. Sites impacted by beaver contained more birds than sites with no beaver activity ($p=0.0024$, $R^2=0.1552$) (Table 3). Total avian abundance was significantly lower in sites that occurred on the Blue Ridge Parkway (BRP) ($p=0.0104$, $R^2=0.0975$) (Table

3). Combined, these two site characteristics explained approximately 25% of the variation in total avian abundance across study sites ($p=0.0104$, $R^2=0.2528$).

Table 3. Wetland habitat characteristics that significantly predicted southern Appalachian wetland bird use at the community-level.

Avian Community DVs	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species diversity (H')	gradual edge	------	0.0206	$z=0.3520$	$p<0.0002$
Species richness	gradual edge	23.51	0.9629	$R^2=0.2995$	$p<0.0001$
	beaver evidence	10.15	1.1270	$R^2=0.1108$	$p=0.0024$
	protected status	9.66	0.9629	$R^2=-0.0909$	$p=0.0030$
Total avian abundance	beaver evidence	10.11	4.6397	$R^2=0.1552$	$p=0.0024$
	BRP status	7.05	3.9681	$R^2=-0.0975$	$p=0.0104$

II. Guild-level analyses

Permanent residents

Two IVs entered the SMLR model as significant predictors of permanent resident species richness. Fewer permanent resident species were detected in ungrazed sites than in grazed sites ($p=0.0055$, $R^2=0.1317$) (Table 4). Also, there were more permanent resident species in sites with a greater percent cover of midstory vegetation ($p=0.0500$, $R^2=0.0602$) (Table 4). Together, grazed status and percent cover of midstory vegetation account for approximately 19% of the total variability in permanent resident species richness ($p=0.0500$, $R^2=0.1919$).

Abundance of permanent residents was most strongly predicted by stem density. Sites with greater stem density values contained more permanent resident birds than sites with lower stem densities ($p<0.0001$, $R^2=0.3245$) (Table 4). Wetland class was also a significant predictor of permanent resident abundance. Forested wetlands contained significantly more permanent resident birds than non-forested wetlands ($p=0.0358$, $R^2=0.0534$) (Table 4). Permanent resident abundance was also positively correlated with percent canopy closure ($p=0.0168$, $R^2=0.0642$) and percent cover of midstory vegetation ($p=0.0376$, $R^2=0.0449$) (Table 4). Collectively, stem density, wetland class, canopy closure, and percent cover of midstory vegetation explained approximately 49% of the total variability in permanent resident abundance ($p=0.0376$, $R^2=0.4870$).

Table 4. Significant wetland habitat characteristics that predicted southern Appalachian wetland use by the permanent resident guild.

Permanent resident guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	grazing status	8.35	0.3841	$R^2=-0.1317$	$p=0.0055$
	% cover midstory vegetation	4.02	0.0121	$R^2=0.0612$	$p=0.0500$
Abundance	square root stem density	26.42	0.0566	$R^2=0.3245$	$p<0.0001$
	forested wetland class	4.63	1.8087	$R^2=0.0534$	$p=0.0358$
	% canopy closure	6.10	0.0459	$R^2=0.0642$	$p=0.0168$
	% cover midstory vegetation	4.55	0.0429	$R^2=0.0449$	$p=0.0376$

Short-distance migrants

There was a negative relationship between percent upperstory cover and short-distance migrant species richness ($p=0.0002$, $R^2= -0.2186$) (Table 5). Also, fewer short-distance migrant species occurred in protected sites than in unprotected sites ($p=0.0049$,

R^2=-0.1073) (Table 5). These two IVs explained approximately 33% of the total variation in short-distance migrant species richness (p=0.0049, R^2=0.3260).

Protected status also entered the SMLR model for short-distance migrant abundance. Significantly lower abundances of short-distance migrants were recorded in sites that were protected (p=0.0006, R^2= -0.1963) (Table 5). Sites that displayed lower stem density values contained significantly more short-distance migrants than sites with high stem density (p=0.0047, R^2= -0.1113) (Table 5). Likewise, more short-distance migrants were observed in sites with less percent cover of groundcover vegetation (p=0.0171, R^2= -0.0711) (Table 5). The weakest predictor of short-distance migrant abundance was percent cover of upperstory vegetation. Sites with less upperstory cover contained significantly fewer short-distance migrants (p=0.0411, R^2= -0.0484) (Table 5). As a group, these four variables described approximately 43% of the total variability in short-distance migrant abundance (p=0.0411, R^2=0.4271).

Table 5. Significant wetland habitat characteristics that predicted southern Appalachian wetland use by the short-distance migrant guild.

Short-distance migrant guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	arc sin % cover upperstory vegetation	15.39	0.9769	R^2=-0.2186	p=0.0002
	protected status	8.60	0.5737	R^2=-0.1073	p=0.0049
Abundance	protected status	13.43	2.9236	R^2=-0.1963	p=0.0006
	stem density	8.68	0.1136	R^2=-0.1113	p=0.0047
	% groundcover vegetation	6.07	0.1000	R^2=-0.0711	p=0.0171
	% cover upperstory vegetation	4.39	0.0928	R^2=-0.0484	p=0.0411

Neotropical migrants

Neotropical migrant species richness was significantly explained by four IVs. Presence of gradual edge ($p<0.0001$, $R^2=0.4246$) and beaver activity ($p=0.0068$, $R^2=0.0737$) both predicted significantly greater Neotropical migrant species richness (Table 6). Percent cover of both grass ($p=0.0030$, $R^2=0.0775$) and groundcover vegetation ($p=0.0416$, $R^2=0.0329$) were negatively correlated with the number of Neotropical migrant species (Table 6). Collectively, these four IVs predicted approximately 61% of the total variation in Neotropical migrant abundance ($p=0.0416$, $R^2=0.6086$).

As with Neotropical migrant species richness, edge type was the strongest predictor of Neotropical migrant abundance. Sites associated with a gradual edge contained more Neotropical migrant species than sites with an abrupt edge ($p<0.0001$, $R^2=0.2813$) (Table 6). The presence of beaver impacts in sites also predicted greater Neotropical migrant species richness ($p=0.0071$, $R^2=0.0910$) (Table 6). Lastly, percent cover of upperstory vegetation displayed a positive relationship with richness of this guild ($p=0.0370$, $R^2=0.0499$) (Table 6). Altogether, edge type, presence of beaver impacts, and percent cover upperstory vegetation explained approximately 42% of the variability in Neotropical migrant species abundance ($p=0.0370$, $R^2=0.4222$).

Table 6. Significant wetland habitat characteristics that predicted southern Appalachian wetland use by the Neotropical migrant guild.

Neotropical migrant guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	gradual edge	40.59	0.6218	R^2=0.4246	p<0.0001
	beaver evidence	7.93	0.8340	R^2=0.0737	p=0.0030
	% cover grass vegetation	9.68	0.0110	R^2=-0.0775	p=0.0068
	% cover groundcover vegetation	4.37	0.0209	R^2=-0.0329	p=0.0416
Abundance	gradual edge	21.53	2.3668	R^2=0.2813	p<0.0001
	beaver evidence	7.83	2.7974	R^2=0.0910	p=0.0071
	% cover upperstory vegetation	4.53	0.0569	R^2=0.0499	p=0.0370

Open-water breeding species

No IVs entered the models for either species richness or abundance of open-water species.

Scrub-breeding species

Of the three IVs that significantly predicted scrub-breeding species richness, basal area displayed the strongest relationship. As site basal area decreased, there were more scrub species present in study sites (p<0.0001, R^2=0.3520) (Table 7). There was a positive correlation between percent forb cover and species richness of this guild (p=0.0045, R^2=0.0906) (Table 7). Also, more scrub species were present in sites that were ungrazed (p=0.0111, R^2= -0.0644) (Table 7). Basal area, percent forb cover, and grazed status explained approximately 51% of the variability in the model for scrub-breeding species abundance (p=0.0111, R^2=0.5071).

Basal area was also the strongest predictor of the abundance of scrub-breeding species (p<0.0001, R^2= -0.2936) (Table 7). Presence of a gradual edge was also a significant predictor of higher scrub-breeding species abundance ($p=0.0173$, $R^2=0.0710$) (Table 7). Together, lower site basal area and presence of a gradual edge explained approximately 37% of the model variability for scrub-breeding species abundance ($p=0.0173$, $r^2=0.3646$).

Table 7. Significant wetland habitat characteristics that predicted southern Appalachian wetland use by the scrub-breeding guild.

Scrub-breeding guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	basal area	29.88	0.0033	$R^2=-0.3520$	$p<0.0001$
	% cover forb vegetation	8.78	0.0122	$R^2=0.0906$	$p=0.0045$
	grazed status	6.93	0.6544	$R^2=-0.0644$	$p=0.0111$
Abundance	basal area	22.86	0.0112	$R^2=-0.2936$	$p<0.0001$
	gradual edge	6.04	2.1231	$R^2=0.0710$	$p=0.0173$

Woodland-breeding species

Four IVs, basal area, edge type, beaver, and percent cover of midstory vegetation entered the model for woodland-breeding bird species richness. More species of woodland-breeding birds were observed in sites with higher basal area values ($p<0.0001$, $R^2=0.4918$) and in sites associated with a gradual edge ($p<0.0001$, $R^2=0.1686$) (Table 8). Also, study sites with beaver impacts were used by relatively more woodland bird species than sites with no evidence of beaver ($p=0.0008$, $R^2=0.0655$) (Table 8). Lastly, sites with higher percent cover of midstory vegetation contained significantly more woodland-breeding species ($p=0.0027$, $R^2=0.0438$) (Table 8). These four variables explained approximately 77% of the overall variation in woodland-breeding species abundance ($p=0.0027$, $R^2=0.7697$).

High basal area ($p<0.0001$, $R^2=0.4937$) and the presence of beaver impacts ($p=0.0002$, $R^2=0.1178$) also had strong positive effects on the abundance of woodland-breeding species (Table 8). Also, more woodland-breeding birds were observed in sites as percent cover of midstory vegetation increased ($p=0.0029$, $R^2=0.0604$) (Table 8). Collectively, basal area, beaver, and percent cover of midstory vegetation described approximately 67% of the total model variability for woodland-breeding species abundance ($p=0.0029$, $R^2=0.6719$).

Table 8. Significant wetland habitat characteristics that predicted southern Appalachian wetland use by the woodland-breeding guild

Woodland-breeding guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	basal area	53.23	0.0039	$R^2=0.4918$	$p<0.0001$
	gradual edge	26.80	0.6847	$R^2=0.1686$	$p<0.0001$
	beaver evidence	12.66	0.7623	$R^2=0.0655$	$p=0.0008$
	% cover midstory vegetation	9.90	0.0234	$R^2=0.0438$	$p=0.0027$
Abundance	basal area	53.36	0.0119	$R^2=0.4937$	$p<0.0001$
	beaver evidence	16.38	2.3394	$R^2=0.1178$	$p=0.0002$
	% cover midstory vegetation	9.76	0.0747	$R^2=0.0604$	$p=0.0029$

Urban-breeding species

Protected status was the only IV that significantly predicted urban-breeding species richness. More urban-breeding species were observed in unprotected than in protected sites ($p=0.0006$, $R^2=0.1937$) (Table 9).

Urban-breeding species abundance was predicted by two independent variables. Wetlands belonging to the persistent emergent class contained significantly more urban-breeding birds than other types of wetlands ($p=0.0006$, $R^2=0.1953$) (Table 9).

Unprotected sites also contained more urban-breeding birds than protected sites ($p=0.0314$, $R^2=0.0667$) (Table 9). These two variables, wetland class and protected status, accounted for nearly 26% of the model variability for urban-breeding species abundance ($p=0.0314$, $R^2=0.2620$).

Table 9. Significant wetland habitat characteristics that predicted southern Appalachian wetland use by the urban-breeding guild.

Urban-breeding guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	protected status	13.21	0.2915	R^2=-0.1937	p=0.0006
Abundance	persistent emergent class	13.35	0.7443	R^2=0.1953	p=0.0006
	protected status	4.88	0.7570	R^2=-0.0667	p=0.0314

Midstory-canopy-nesting species

Richness of species nesting in the upper layers of vegetation was greater when a gradual edge rather than an abrupt edge was associated with the wetland ($p=0.0004$, $R^2= -0.2073$) (Table 10). Also, midstory-canopy-nesting species richness increased as percent groundcover vegetation decreased across sites ($p=0.0096$, $R^2=0.0934$) (Table 10). Combined, these two IVs contributed 30% of the total model variability ($p=0.0096$, $R^2=0.3008$).

Three IVs entered the SMLR model for midstory-canopy-nesting species abundance. As with species richness of this guild, significantly more midstory-canopy-nesting birds were detected in sites with a gradual edge than in sites containing abrupt edges ($p=0.0004$, $R^2=0.2073$) (Table 10). Also, more midstory-canopy-nesting birds were observed in forested wetlands than in non-forested wetlands ($p=0.0018$, $R^2=0.1320$) (Table 10). Lastly, sites with beaver evidence contained significantly more mid-high nesters than sites with no evidence of beaver ($p=0.0374$, $R^2=0.0523$) (Table 10). Collectively, edge type, wetland class, and beaver significantly explained about 39% of the variability in mid-high level- nesting species abundances in wetland sites ($p=0.0374$, $R^2=0.3917$).

Table 10. Wetland habitat characteristics that significantly predicted southern Appalachian wetland use by the midstory-canopy-nesting guild.

Midstory-canopy-nesting guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	gradual edge	14.39	0.8267	$R^2=0.2073$	$p=0.0004$
	% groundcover vegetation	7.21	0.0260	$R^2=-0.0934$	$p=0.0096$
Abundance	gradual edge	14.39	0.8054	$R^2=0.2073$	$p=0.0004$
	forested wetland class	10.79	0.9147	$R^2=0.1320$	$p=0.0018$
	beaver evidence	4.56	0.9584	$R^2=0.0523$	$p=0.0374$

Ground-shrub-nesting species

Three variables entered the regression model as predictors of ground-shrub-nesting species richness. Species richness of this guild was higher in non-BRP sites than in BRP sites ($p=0.0088$, $R^2=0.1184$) (Table 11). Ground-shrub-nesting species richness was also greater in sites associated with a gradual edge ($p=0.0076$, $R^2=0.1001$) (Table 11). Lastly, as percent cover of midstory vegetation increased across study sites, fewer ground-shrub-nesting species were observed ($p=0.0143$, $R^2= -0.0936$) (Table 11). Altogether, these three habitat variables explained approximately 31% of the overall variation in the species richness of ground-shrub-nesting birds ($p=0.0076$, $R^2=0.3121$).

The strongest predictor of lower nesting species abundance was the presence of beaver evidence (p<0.0001, R^2=0.2580) (Table 11). Forested wetlands also contained more ground-shrub nesters than non-forested wetlands (p=0.0030, R^2=0.1127) (Table 11). Greater abundance of this guild also occurred in sites that were protected relative to unprotected sites (p=0.0144, R^2=0.0678) (Table 11). Also, there was a negative relationship between percent cover of standing water and the abundance of ground-shrub-nesting birds (p=0.0448, R^2=0.0422) (Table 11). The last variable to enter the model for ground-shrub nester abundance was grazed status. Grazed sites contained significantly fewer ground-shrub-nesting birds than ungrazed sites (p=0.0434, R^2=0.0403) (Table 11). These five wetland characteristics alone significantly explained over 52% of the model variability in abundance of this guild (p=0.0434, R^2=0.5210).

Table 11. Wetland habitat characteristics that significantly predicted southern Appalachian wetland use by the ground-shrub-nesting guild.

Ground-shrub-nesting guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	BRP status	7.39	0.6823	^{R2}=-0.1184	p=0.0088
	gradual edge	7.71	0.6752	R^2=0.1001	p=0.0076
	% cover midstory vegetation	6.41	0.0226	R^2=-0.0936	p=0.0143
Abundance	beaver evidence	19.12	2.6367	R^2=0.2580	p<0.0001
	forested wetland class	9.67	2.3928	R^2=0.1127	p=0.0030
	protected status	6.40	2.0357	R^2=0.0678	p=0.0144
	% water cover	4.23	0.0983	R^2=-0.0422	p=0.0448
	grazed status	4.29	2.1578	R^2=-0.0403	p=0.0434

Cavity-nesting species

The only IV to enter the SMLR model for cavity-nesting species richness was stem density. Cavity-nesting species richness increased with stem density (p=0.0388,

R^2=0.0755) (Table 12). Cavity-nesting species abundance was also predicted by a single habitat variable. More cavity-nesting birds were observed in forested wetlands as opposed to non-forested wetlands (p=0.0033, R^2=0.1467) (Table 12).

Table 12. Wetland habitat characteristics that significantly predicted southern Appalachian wetland use by the cavity-nesting guild.

Cavity-nesting guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	stem density	4.48	0.0153	R^2=0.0755	p=0.0388
Abundance	forested wetland class	9.45	1.4415	R^2=0.1467	p=0.0033

Woodpeckers

Not unlike cavity nesters, only stem density entered the regression model as a significant predictor of woodpecker species richness. As stem density increased in study sites, so did the species richness of woodpeckers (p=0.0003, R^2=0.2153) (Table 13). Woodpecker abundance was also only predicted by one habitat variable: percent cover of midstory vegetation. Sites with greater vegetational cover in the midstory contained significantly more woodpeckers than sites with less midstory cover (p=0.0003, R^2=0.2117) (Table 13).

Table 13. Wetland habitat characteristics that significantly predicted southern Appalachian wetland use by woodpeckers.

Woodpeckers	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	stem density	15.0900	0.0096	^{R2}=0.2153	p=0.0003
Abundance	% cover midstory vegetation	19.0583	0.0098	^{R2}=0.2117	p=0.0003

Insectivorous species

Four IVs, edge type, beaver, protected status and wetland class significantly predicted species richness of insectivores. Sites associated with a gradual edge contained more insectivorous species than sites associated with an abrupt edge ($p<0.0001$, $R^2=0.3851$) (Table 14). Also, when a wetland showed evidence of beaver activity there were more insectivorous species observed relative to sites with no beaver activity ($p=0.0132$, $R^2=0.0667$) (Table 14). Protected sites contained fewer insectivorous species than unprotected sites ($p=0.0255$, $R^2= -0.0497$) (Table 14). Lastly, insectivore richness was higher in forested wetlands than in non-forested wetlands ($p=0.0409$, $R^2=0.0389$) (Table 14). Collectively, these four IVs described approximately 54% of the variability in the insectivorous species richness model ($p=0.0409$, $R^2=0.5404$).

Three variables loaded into the regression model as significant predictors of insectivore abundance. The strongest relationship was greater insectivore abundance in sites impacted by beaver ($p=0.0030$, $R^2=0.1493$) (Table 14). Sites on the Blue Ridge Parkway contained significantly fewer insectivores than in sites not on the Parkway ($p=0.0313$, $R^2= -0.0706$) (Table 14). Lastly, insectivores decreased in abundance as a site's percent cover of grass increased ($p=0.0494$, $R^2= -0.0553$) (Table 14). Altogether, beaver, Blue Ridge Parkway status, and percent cover of grass explained about 28% of the total model variability for insectivore abundance ($p=0.0494$, $R^2=0.2752$).

Table 14. Wetland habitat characteristics that significantly predicted southern Appalachian wetland use by insectivores.

Insectivorous guild	*Predictors*	*F*	*S.E.*	*Influence*	*Significance*
Species richness	gradual edge	34.44	0.8523	$R^2=0.3851$	$p<0.0001$
	beaver presence	6.57	0.9981	$R^2=0.0667$	$p=0.0132$
	protected status	5.28	0.8777	$R^2=-0.0497$	$p=0.0255$
	forested wetland class	4.40	0.9767	$R^2=0.0389$	$p=0.0409$
Abundance	beaver evidence	9.65	16.3814	$R^2=0.1493$	$p=0.0030$
	BRP status	4.89	-7.5849	$R^2=-0.0706$	$p=0.0313$
	% cover grass vegetation	4.04	-0.1234	$R^2=-0.0553$	$p=0.0494$

DISCUSSION

Many of the breeding guild responses to wetland habitat variables were intuitive; i.e., they might have been expected when considering the breeding habitat requirements of birds that composed certain guilds. For instance, both species richness and abundance of scrub-breeding birds were negatively correlated with the basal area of trees at the site. Also, scrub-breeding birds were more abundant in sites as the percent of forb vegetation increased. One may expect basal area and percent forb cover, each representative of habitats with low tree cover, to correlate with species that breed in shrub dominated wetlands. On the other hand, higher basal area and greater percent cover of midstory vegetation predicted greater woodland-breeding species richness and abundance. This too was an expected outcome, as more basal area and midstory cover characterize wooded habitats, and thus should be attractive to woodland-breeding birds.

Results of the urban-breeding guild analyses were intuitive as well, though perhaps not as obvious as the results of scrub-breeding and woodland-breeding guilds. Urban-breeding species richness and abundance were both higher in unprotected sites relative to protected sites. Protected status may not be expected to enter the urban-breeding regression models based on the within-site variables that I measured, because mere protection may not produce consistent measurable differences in vegetation characteristics of wetlands. Therefore, the within-site characteristics I measured at the local scale, may not have differed markedly between protected and unprotected sites. As a result, local vegetation characteristics, probably unrelated to within-site vegetation, may not have demonstrated the factors related to protected status that were influencing urban-breeding species.

Rather, this relationship may indicate that influences on urban-breeding birds occurred beyond the scale of the variables measured in this study; specifically, the urban-breeding guild may have responded to landscape level attributes. The landscape surrounding protected wetlands was usually composed of intact forest expanses and/or large open pastures. In contrast, unprotected sites were often situated within patchier landscapes that were more likely to contain urban and residential areas. Therefore, one might have expected more urban species and individuals to use unprotected sites due to the different landscape compositions surrounding protected versus unprotected sites.

Results for the cavity-nesting and woodpecker guilds also were expected, as the species richness of both guilds displayed positive correlations with stem density. Furthermore, cavity nesters were more abundant in sites that were forested, and woodpeckers were more abundant in sites with greater midstory cover. The nesting and foraging requirements of species in both of these guilds, i.e. trees and forests respectively, explain the effects of these forest-related variables.

In addition to guilds constructed based on breeding habitat preferences, guilds reflecting migratory behavior also demonstrated expected responses to within-site habitat variables. These responses were almost certainly driven by the breeding habitat preferences of species that constituted migration guilds. For instance, four of the variables (% cover midstory, stem density, forested site type, % canopy closure) that predicted more species and individuals of permanent residents are characteristics of forested wetlands. Forest-related characteristics were probably valuable to the permanent resident guild because 11 of the 16 permanent resident species are woodland breeders. The strongest predictor of permanent resident species richness, ungrazed status, may have also appeared because most permanent residents are woodland breeders. Grazed wetlands were more often non-forested than forested. Therefore, grazed sites were probably less useful to a guild dominated by woodland-breeding species.

Short-distance migrant species richness and abundance displayed an expectedly negative relationship with percent cover of upperstory vegetation. This relationship was probably incidental due to the breeding habitat preferences of the short-distance migrants that were observed in study wetlands. Short-distance migrant species richness and abundance may have been greater in sites with less upperstory cover due to the fact that 22 of the 30 species observed in this guild prefer to breed in non-forested habitats.

While many of the guild-level and community-level results were expected, others were not. Interpretations of these anomalous results may prove even more valuable to our understanding of how breeding birds use southern Appalachian wetlands.

One of the most curious outcomes of this study was the paucity of wetland-breeding species. No truly obligate wetland species (i.e., waterfowl and wading birds) were observed on all three visits to any wetland. This was probably due to the small sizes of wetland study sites. Average wetland size was only 4.58 hectares. In fact, when wetland sizes were averaged excluding the four largest sites (those exceeding 20 hectares), average size was only 1.59 hectares. Studies of wetland bird habitat requirements have found that waterfowl and wading bird species richness are more positively influenced by wetland size and the area of open, shallow water than by vegetative characteristics (Kantrud & Stewart 1984; Brown et al. 1996, VanRees-Siewert & Dinsmore 1996; Edwards & Otis 1999). Thus, the small southern Appalachian wetlands associated with this study probably are not suitable for breeding waterfowl and wading birds.

In contrast, all of my study wetlands were characterized by an abundance of avian species that are not wetland obligates. These non-wetland species represented a diversity of breeding habitat preferences and migration strategies. In particular, Neotropical migrants (n=41 species) and woodland-breeding species (n=48) were especially well represented in southern Appalachian wetlands; these two guilds accounted for more than half of all species observed (n=92) across all 13 guilds. Southern Appalachian wetlands may be important to species in these two guilds, many of which are in decline. Therefore, the habitat variables that entered models for Neotropical migrants and woodland-breeding species are particularly interesting.

Specific habitat variables that predicted higher species richness and abundance of Neotropical migrants and woodland breeders may explain *why* certain attributes of southern Appalachian wetlands might benefit these species. Moreover, the results of these two guilds' regression models may also explain community-level responses to wetland characteristics, in that these two guilds compose the majority of species observed in this study. Two variables, edge type and beaver activity, entered models not only for these two guilds but also entered several community-level regression models. Therefore, Neotropical migrant and woodland-breeding guild responses to edge type and beaver may be driving avian community-level responses to these variables as well.

Edge type entered guild-level regression models more consistently than any other variable. In fact, presence of a gradual edge predicted higher richness and/or abundance of 8 of the 13 guilds analyzed. At the community-level, gradual edge in wetlands was also the sole predictor of greater species diversity and the strongest predictor of higher species richness.

The negative effects of edges on Neotropical migrants and woodland-breeding species have been well studied. In general, predator abundance (Temple & Cary 1988; Wilcove & Robinson 1990; Faaborg et al. 1995), predator species richness (Forsyth & Smith 1973; Heske 1995; Chalfoun et al. 2002a), and brood parasite abundance (brown-headed cowbirds) (Brittingham & Temple 1983; Johnson & Temple 1990) have been found to be greater at habitat edges. While these general edge effects have been well documented, the differential effects of edge types on breeding birds have received much less attention (i.e., Suarez et al. 1997; Lopez de Casenave et al. 1998).

Wetlands with gradual edges may have been better breeding habitats than those with abrupt edges due to lower rates of predation and parasitism in gradual edges. In a study of the indigo bunting (*Passerina cyanea*), a Neotropical migrant, Suarez et al. (1997) found that nesting success was lower near abrupt edges than gradual edges. This was attributed to increased cowbird parasitism and nest predation at abrupt edges (Suarez et al. 1997). Neither nest predation nor brood parasitism rates were measured in this study. However, I can speculate on parasitism rates in wetlands with different edge types based on cowbird abundance. Of the five brown-headed cowbirds that were observed across all sites, four of these occurred in grazed pastures with abrupt edges. This evidence, though anecdotal, suggests that cowbirds are less abundant in southern Appalachian wetlands with gradual edges. Predation rates may have also been lower in wetlands with gradual edges relative to wetlands with abrupt edges, as was the case for Suarez et al. (1997). If parasitism and predation rates are lower in wetland sites with gradual edges, then breeding birds may prefer wetlands with gradual edges over those associated with abrupt edges.

Greater avian use of sites with gradual edges may also be due to differences between gradual and abrupt edges in terms of vegetation. In a study comparing bird communities of "mature," or gradual, edges with the bird communities of surrounding habitat patches, Lopez de Casenave et al. (1998) found higher avian species richness and greater total bird abundance in the gradual edge habitats. Lopez de Casenave et al. (1998) suggested that more species were nesting in gradual edges due to more nesting strata in the well developed edge vegetation. They also attributed higher bird diversity and abundance to a greater diversity of foraging niches. More foraging niches resulted from greater vegetation structural complexity and higher fruit production by plants in gradual edges than in surrounding habitats (Lopez de Casenave et al. 1998). In light of these findings, wetlands with gradual edges may provide more nesting strata and foraging niches for breeding birds than those with abrupt edges.

Interestingly, sites containing gradual rather than abrupt edges were often the same wetlands that displayed beaver activity. Presence of beaver activity, much like presence of a gradual edge, predicted higher species richness and abundance of both Neotropical migrants and woodland-breeding species. Beaver evidence also predicted greater species richness and total abundance at the community-level. Gradual edges may result from activities of this ecosystem engineer in southern Appalachian wetlands, thereby enhancing the breeding bird communities associated with these habitats.

Beaver may have benefited breeding birds not only by creating gradual edges, but also through their effects on wetland vegetation structure. In a study comparing breeding birds of beaver wetlands versus wetlands with no beaver activity, Grover and Baldassarre (1995) found that beaver activity might benefit birds via their impacts on vegetational complexity. Grover and Baldassarre (1995) found that active beaver wetlands harbored more Neotropical migrants and woodland species (referred to as facultative wetland species) during the breeding season, than wetlands with past beaver activity or wetlands only potentially inhabited by beaver. Grover and Baldassarre (1995) suggest that more facultative species may have been found in active sites because beaver activity creates wetlands with structurally more diverse vegetation mosaics than non-beaver wetlands. In southern Appalachian wetlands, beaver may have the same positive impact on Neotropical migrants and woodland-breeding species by diversifying wetland vegetation structure.

Beaver activity may not only generate greater habitat complexity for Neotropical migrants and woodland-breeding birds, but also may enhance the foraging efficiency for these birds. At least two studies have documented higher avian use of beaver wetlands than non-beaver wetlands, and suggested that this may be due to increased insect productivity, as insects constitute at least part of many bird species' diets (Reese & Hair 1976; Medin & Clary 1990). Although I did not measure insect abundance directly, my results are consistent with these suggestions (Reese & Hair 1976; Medin & Clary 1990) as well as with the observations (McDowell & Naiman 1986) that beaver wetlands produce more insects than non-beaver wetlands. In my study, beaver-impacted wetlands displayed higher species richness and abundance of insectivorous bird species than non-beaver wetlands. Furthermore, this relationship may be driving the positive effect of beaver on Neotropical migrants and woodland-breeding species, since insects dominate the diets of most Neotropical migrants and woodland breeders in this study.

Neither Neotropical migrants nor woodland-breeding species were influenced by protected status or BRP status, yet these variables did show some interesting relationships with other guilds. Protected status and BRP status entered eight guild-level models as well as two community-level models, and in nine of these ten cases protected status or BRP status displayed a negative effect. The two community-level responses to these variables were most likely incidental and due to the fact that protected status and BRP status also entered eight guild-level models. In fact, the vast majority of species observed in this study (82 of the total 92 species) belonged to at least one of the guilds for which there was a negative effect of either protected status or BRP status. For this reason, the negative effects of protected status and BRP status may be best interpreted via their community-level effects.

The negative effect of protected status on overall species richness may be for similar reasons that protected status negatively affected the urban-breeding guild; i.e., community-level species richness may be lower in protected sites due to landscape level phenomena. Landscapes containing non-protected sites typically displayed high habitat diversity. The landscape surrounding non-protected sites was typically disturbed, consisting of a mosaic of forests, shrub habitats, pastures, agricultural lands, residential developments and/or urban areas. On the contrary, protected sites characteristically occurred within less fragmented landscapes, and were thus surrounded by fewer, but larger patches of forest or pasture. Therefore, fewer breeding habitat types may have been available in the vicinity of protected wetlands than in the landscape matrix of non-protected wetlands. It is probably for this reason that fewer species were observed using protected wetlands than non-protected wetlands.

The negative relationship of short-distance migrant richness and abundance with protected status may further suggest that landscape heterogeneity is driving the influence of protected status on birds. The short-distance migrant guild, in particular, consists of a suite of species that prefer a diversity of breeding habitats. Thus, the short-distance migrant guild would expectedly be more abundant in wetlands surrounded by more habitat types. Indeed, this interpretation agrees with earlier studies of relationships among habitat heterogeneity, habitat area, and bird communities (Whitcomb et al. 1981; Freemark & Merriam 1986). In these studies, short-distance migrant species were more abundant in sites situated within more patchy habitats, rather than in areas exhibiting more homogenous habitat dispersion (Whitcomb et al. 1981; Freemark & Merriam 1986).

One may expect BRP status to show similar effects in the same community-level models as protected status since BRP sites are really a subset of all protected sites. However, this is not the case. Rather, BRP sites showed a negative effect on total avian abundance rather than species richness. This implies that Blue Ridge Parkway sites share some common characteristic distinguishing them from protected sites.

Most Blue Ridge Parkway wetlands were disturbed via grazing or mowing, due to the Parkway's mandate to preserve historical human land use practices. These disturbances usually created a homogenous and simple vegetation structure of only grass. Therefore, BRP wetlands may have provided less forage substrata and cover than was available in sites with more developed vegetation. Also, disturbances in BRP sites usually created very abrupt edges at or near wetlands. This, especially considering the negative effect of abrupt edge on several guilds, overall species diversity, and overall species richness, may indicate that BRP wetlands harbored fewer individuals due to their abrupt edges, which result from human disturbance. The tendencies for BRP sites to display simple vegetation structure, as well as abrupt edges due to human disturbance, may collectively contribute to the overall negative effect of BRP status on total avian abundance.

Management Implications

Overall, this study indicates that southern Appalachian wetland avifaunas are unlike the bird communities typically found in larger wetlands. Rather than providing breeding habitat for wetland species, southern Appalachian wetlands are heavily used by non-wetland birds, notably breeding Neotropical migrants and woodland-breeding species. Although many of the species in these two guilds are relatively abundant in the southern Appalachian region, they are also in the most severe declines in this part of their breeding ranges (Rodriguez 2002). Results from this study, therefore provide a key general finding. Not only are southern Appalachian wetlands important to many threatened or endangered species of flora (e.g., Swamp Pink Lily, Mountain Sweet Pitcher Plant, Gray's Lily) and herpetofauna (e.g., the Bog Turtle, the Four-toed Salamander (*Hemidactylium scutatum*), the Mole Salamander (*Ambystoma talpoideum*)), but these habitats also appear to be important to even more species of breeding birds. In fact, many of these bird species are suffering declines throughout their breeding ranges (see Appendix). Furthermore, some of the species observed in this study are of high conservation concern in the southern Appalachian region in particular (see Appendix). Therefore, the findings of this study provide additional reasons why action should be taken to conserve existing southern Appalachian wetlands.

Beyond suggesting that southern Appalachian wetlands should be conserved, this study identifies the particular wetland characteristics that may promote their use by breeding birds, namely Neotropical migrants and woodland-breeding species. The wetland characteristics that predicted greater richness and abundance of these two guilds, as well as greater community-level species diversity, richness and abundance, indicate *how* these habitats may be best managed to facilitate conservation of breeding bird communities. In particular, this study suggests that management techniques involving wetland edges, grazing and beaver could benefit breeding birds. Since many southern Appalachian wetlands are publicly or federally owned, these sites offer great opportunities for management strategies to be tested.

In agreement with previous studies, my results suggest that maintenance of gradual edges must be promoted over abrupt edges in southern Appalachian wetlands to make these habitats more attractive to breeding birds. Given that many wetlands on the BRP have abrupt edges due to human disturbances, these wetlands may offer an opportunity to explore ways to "gradualize" southern Appalachian wetland edges. Furthermore, edge management could also be attempted in BRP pastures that do not contain wetlands.

Edge management techniques could be employed relatively easily on the Parkway. The National Park Service hires interns yearly to construct and maintain cattle fences on the Blue Ridge Parkway. Perhaps some of this effort could be directed towards shifting fences in several-year cycles to gradualize edges. This could be accomplished without repeated time- and labor-intensive destruction and reconstruction of livestock fences. For example, in a cow pasture that abuts a mature forest, one could simply install more fence posts; some inside the forest edge and some further into the pasture. Then, barbed wire could be easily moved from old posts to new posts. Overall, this would restrict cows from some recently grazed areas, while allowing them access to areas that were previously protected. As a result, some revegetation would begin to take place in the newly restricted pasture edge. After a certain number of years, the barbed wire could be moved back to the original posts, reversing the cattle's access to different areas. Grazing would then redisturb the area that was previously fenced off. After several cycles of this systematic cattle exclusion and inclusion, edges may be made less abrupt. Rather than a permanent abrupt edge marked by a fence line at a pasture-forest ecotone, this management technique might produce a persistent gradual edge of arrested succession around a cow pasture.

Likewise, mowing practices could be manipulated to facilitate the maintenance of gradual edges as well. For example, mowing could be relaxed at meadow-forest ecotones. This may allow more forbs and shrubby vegetation to become established. Eventually, this could lead to more gradual edges around mowed meadows than those that currently occur where there is more intense mowing up to fence lines.

Furthermore, based on the effect of BRP status on total avian abundance, it seems that grazing and mowing practices throughout the wetlands, not just at edges, might be detrimental to many birds that use these habitats. Therefore, it might be worthwhile to reduce the continual disturbances wrought by year-after-year grazing and mowing, by periodically excluding cattle from, and ceasing mowing in, BRP wetlands. A periodicity of these disturbances might enhance vegetational structural diversity, by promoting wetland floral development, while maintaining stable early successional vegetation stages in wetlands.

In fact, the lack of stability of early successional habitats is partially to blame for many early successional NTMB declines, as unmanaged early successional habitats often revert to late successional stages rapidly (Thompson & DeGraaf 2001). Managing Parkway wetlands temporally, by making grazing and mowing disturbances more periodic, could create a spatial mosaic of early successional stages throughout the landscape. In this way different wetlands could be simultaneously managed in favor of several early successional species that prefer different stages of early succession. For instance, Eastern Meadowlarks, one of the most imperiled species in the southern Appalachians (Hunter et al. 2001), could benefit from habitats managed as stable open meadows. Indeed, Eastern Meadowlarks were only observed in grazed wetlands in this study. For Eastern Meadowlarks, cattle could be grazed *more* often in some wetlands to maintain open, grassy habitats. Grazing or mowing could be allowed *less* often in other wetlands, leading to a more structurally complex habitat, which other species prefer for breeding. This less frequent disturbance would maintain suitable breeding habitat for another species of high conservation concern, the Golden-winged Warbler (Hunter et al. 2001). Golden-winged Warblers require large early successional habitats that contain some shrub cover (Hunter et al. 2001) and high singing perches (Rossell 2001). Ultimately, if grazing and mowing disturbances were managed temporally in wetlands, the Parkway could effectively satisfy both their mandates to preserve historical land use as well as regional avifaunal diversity.

Results from this study also highlight the importance of healthy beaver populations to Neotropical migrants, woodland-breeding birds, and to community species richness and total avian abundance in southern Appalachian wetlands. However, beaver preservation is certain to be a sensitive issue with private landowners. This ecosystem engineer is typically considered a nuisance, and landowners would thus translate their preservation into an invitation for unnecessary problems (e.g., flooding) that would decrease property values. For this reason, the most realistic way that beaver could be protected is by promoting populations that already inhabit state and federally owned lands.

However, current beaver management strategies differ between state and federally owned lands. Beaver preservation is already taking place on federally owned BRP lands included in this study; e.g., the Blue Ridge Parkway is the only agency that prohibits beaver trapping without specific waivers granted to the trapper(s). In contrast, beaver can be trapped legally at certain times of the year in National Forests. This study suggests that beaver management on BRP lands, via promoting beaver activity, may be more beneficial to breeding birds than in beaver management in National Forests. Perhaps trapping seasons in National Forests could be shortened, only opened in alternating years, or closed altogether, to potentially allow beaver activity to increase in these areas.

If beaver populations are allowed to expand, perhaps bird communities of southern Appalachian wetlands, as well as the avifauna of the entire southern Appalachian region, may benefit. First of all, beaver may provide a free ecosystem service by facilitating wetland persistence in the southern Appalachians via repeated invasion and abandonment of wetlands. Small southern Appalachian wetlands, occurring in a region of such high topographic relief, are inherently ephemeral. If beaver activity stabilized these habitats, or consistently created them, returning migratory species that breed in these wetlands could rely on their presence in the landscape every year. Additionally, beaver activity would create gradual edges in these wetlands, another habitat quality that enhanced breeding bird communities in this study.

Also, more beaver in the Appalachian landscape would likely result in more of the highly productive habitats, in terms of insects and vegetation, that birds find important. Specifically, late successional insectivorous birds may benefit from increased insect productivity whenever beaver flood a mature forest. Furthermore, open beaver impoundments that remain after beaver have harvested canopy vegetation in an area may also display high insect productivity, thereby benefiting early successional insectivorous species that might use these canopy gaps. By increasing insect abundance, more beaver activity might produce more avian hotspots for a wide variety of bird species in the southern Appalachians.

Moreover, beaver could presumably benefit early successional bird species in the southern Appalachians via the habitats that result from beaver activity. Beaver could create early successional beaver meadows, which result when beaver abandon a wetland and the impoundment drains. Beaver activity would thus benefit threatened Neotropical migrants that prefer early successional seral stages. Ultimately, if management actions were invested in promoting this ecosystem engineer, it may pay off in more and improved breeding bird habitats. Allowing beaver to create and maintain habitats for declining species might very well be easier and less costly than trying it ourselves.

Further research suggestions

Many of the conclusions generated by this study could benefit from more detailed investigations. Research on the effects of gradual edges on avian reproductive success in the southern Appalachians, via nesting success studies, would be an informative next step. Nest success studies could, for example, more directly assess the effects of brood parasitism on birds breeding in habitats with different types of edges. Nest monitoring studies might also provide a better understanding of predator activity in southern Appalachian wetlands. In a recent study, predator diversity was found to vary greatly across different ecotone types (Chalfoun et al. 2002a), and Chalfoun et al. (2002b) subsequently suggested that more site- and predator species-specific studies of nest predation across different types of ecotones should be conducted. It would be interesting to compare predator effects on birds among different wetland types as well as among wetlands with different edge types. Furthermore, a comparison of predator effects as well as the predator community in southern Appalachian wetlands with other southern Appalachian habitats may reveal that these wetlands have unique predator communities altogether.

A more detailed study of beaver impacts on avian reproductive success would also provide a better understanding of how beaver promote higher richness and abundance of certain bird guilds in southern Appalachian wetlands. A comparative study of nesting success in beaver impacted wetlands with wetlands that are not impacted by beaver would indicate if these habitats offer better nesting habitats in terms of gradual edges and overall vegetation structure. For instance, if beaver were creating more complex vegetation structure and gradual edges in wetlands to the benefit of breeding birds, then nest success would be higher in beaver wetlands. Also, comparative studies of insect diversity, abundance and biomass, coupled with nesting success studies in beaver impacted versus non-beaver wetlands could show whether or not beaver wetlands are actually better foraging habitats for birds. If beaver wetlands are more productive than non-beaver wetlands, in terms of insects, then nestling and fledgling growth rates should be higher in wetlands with beaver.

It would also be useful to examine the potential importance of southern Appalachian wetlands to birds outside the breeding season; studies conducted during migration, the post-breeding season, and winter could be informative of year-round importance of southern Appalachian wetlands to birds. More open wetlands may receive more solar radiation and thus be warmer than later successional habitats in the southern Appalachians. This difference might be important to birds, especially during migration, as the more highly insolated wetlands may be more productive than forests during this time of year. On many occasions, I have witnessed migratory warblers (e.g. Yellow Warblers (*Dendroica petechia*), Yellow-rumped Warblers (*Dendroica coronata*)) foraging in southern Appalachian wetlands well before the breeding season. These observations may be evidence that southern Appalachian wetlands are critical stopovers for Neotropical migrants.

Southern Appalachian wetlands may also serve as feeding stopovers for obligate wetland species during migration, including migratory waterfowl and wading birds. Indeed, during visits to my study sites during spring and fall migrations, I observed Blue-winged Teal (*Anas discors*), Green-winged Teal (*Anas crecca*), Lesser Scaup (*Aythya affinis*), Greater Yellowlegs (*Tringa melanoleuca*), and Solitary Sandpipers (*Tringa solitaria*).

Southern Appalachian wetlands may also be important during the post-breeding season to juvenile and adult songbirds that typically breed in late successional habitats. For example, Pagen et al. (2000) found that many forest-breeding songbirds were more often found in early successional than late successional habitats after their broods had fledged. They (Pagen et al. 2000) suggested that these post-breeding adults and young-of-the-year were foraging in early successional habitats because these habitats are highly productive. Therefore, foraging in these habitats enabled birds to build up fat reserves in preparation for fall migration. Pagen et al. (2000) also conclude that early successional habitats may have offered greater cover from predators than the late successional forests where these species breed. Highly productive early sucessional southern Appalachian wetlands may be important to post-breeding adult and juvenile forest interior bird species for exactly the same reasons.

In the winter, many permanent resident species could use these habitats for foraging. During the cold winter months, the openness of early successional wetlands may provide warmer microclimates relative to other Appalachian habitats. These slightly warmer wetlands may thaw more often than other Appalachian habitat types which would remain more consistently snow covered or frozen. Resources may therefore be more readily available in wetlands than in forests during winter months. Permanent residents may take advantage of southern Appalachian wetlands at these times.

It would also be useful to expand on my study in a spatial context. A landscape level analysis, for example, could offer an understanding of how birds may differentially use wetlands based on the make-up of the surrounding landscape matrix. A GIS-based analysis of the vegetation classifications and land use practices that surround southern Appalachian wetlands using the SAMAB (Southern Appalachian Man and Biosphere Project) (SAMAB 1996) database would achieve a spatially explicit view of breeding bird use of southern Appalachian wetlands. Because southern Appalachian wetlands are typically small, the landscape matrix may be more important to avian community structure than the within-site wetland characteristics that I measured.

Specifically, a landscape level study could be used to test my hypotheses regarding how a site's protected status affects breeding birds in these wetlands. A landscape-focused study would compare within-site bird census data from this study with landscape variables such as percent cover of different landcover types and habitat diversity. For example, short-distance migrants may be found to be more abundant in wetlands that are surrounded by a high diversity of habitats, since this guild consists of birds preferring a variety of breeding habitats. Likewise, community-level species richness would be expected to be higher in wetlands situated in more diverse landscapes.

On the other hand, Neotropical migrants and woodland-breeding species may be found to correlate negatively with landscape level habitat diversity. These area-sensitive species, whether early or late successional, typically prefer large undisturbed expanses of suitable breeding habitat, as opposed to patchy landscapes (Askins et al. 1990). Therefore, a landscape level study of these guilds in southern Appalachian wetlands may show that they are more abundant in wetlands surrounded by less fragmented breeding habitat.

If landscape scale effects on bird communities related to landcover and habitat patch sizes are found, one may presume that conservation of certain habitats near these small wetlands, in addition to the wetlands themselves, would be important management considerations. Also, a landscape level understanding of breeding bird use of southern Appalachian wetlands may help to prioritize which wetlands we attempt to conserve. For instance, wetlands surrounded by greater expanses of forest may be more important to forest interior species. Ultimately, knowledge of both within-site and landscape level predictors will be required if we hope to develop effective conservation strategies for birds using southern Appalachian wetlands.

BIBLIOGRAPHY

Askins, R. A. 2001. Sustaining biological diversity in early successional communities: the challenge of managing unpopular habitats. Wildlife Society Bulletin **29:** 407-412.

Askins, R. A., J. F. Lynch, and R. Greenberg. 1990. Population declines in migratory birds in eastern North America. Pages 1-57 in D. M. Power, editor. Current Ornithology, Volume 7. Plenum Press: New York.

Brittingham, M. C. and S. A. Temple. 1983. Have cowbirds caused forest songbirds to decline? Bioscience **33:** 31-35.

Brown, D. J., W. A.. Hubert, and S. H. Anderson. 1996. Beaver ponds create wetland habitat in mountains of southeastern Wyoming. Wetlands **16:** 127-133.

Buhlman, K. A., J. C. Mitchell, and M. G. Rollins. 1997. New approaches for the conservation of bog turtles, *Clemmys muhlenbergii*, in Virginia. Pages 359-363 in J. Van Abbema, editor. Proceedings: Conservation, Restoration, and management of Tortoises and Turtles – An International Conference. New York Turtle and Tortoise Society: New York.

Carter, M. F., W. C. Hunter, D. N. Pashley, and K. V. Rosenberg. 2000. Setting conservation priorities for landbirds in the United States: The Partners in Flight approach. Auk **117:** 541-548.

Chalfoun, A. D., M. J. Ratnaswamy, and F. R. Thompson III. 2002a. Songbird nest predators in forest-pasture edge and forest interior in a fragmented landscape. Ecological Applications **12:** 858-867.

Chalfoun, A. D., F. R. Thompson, and M. J. Ratnaswamy. 2002b. Nest predators and fragmentation: a review and meta-analysis. Conservation Biology **16:** 306-318.

Coffey, J. W. and J. L. Shumate, Jr. 1999. Bird study in Shady Valley, Tennessee. Universal Printing: Bristol, VA.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. The Cowardin Classification of Wetland and Deepwater Habitats of the United States. U.S. Fish and Wildlife Service Report FWS-OBS 70-31.

Cox, G. W. 1996. Laboratory Manual of General Ecology. William C. Brown Publishers: Debuque, IA.

Dahl, T. E. 2000. Status and trends of wetlands in the conterminous United States 1986 to 1997. U.S. Fish and Wildlife Service, Washington, D.C.

Davis, M. B. 1993. Old growth in the east. The Cenozoic Society: Richmond, Va.

Delcourt, H. R.and P. A. Delcourt. 1997. Pre-Columbian Native American use of fire on southern Appalachian landscapes. Conservation Biology **11:** 1010-1014.

Delcourt, P. A. and H. R. Delcourt. 1981. Vegetation maps for eastern North America: 40,000 yr b.p. to the present. Pages 123-165 in R. C. Romans, editor. Geobotany II, Plenum Press: NY.

Edwards, N. T. and D. L. Otis. 1999. Avian communities and habitat relationships in South Carolina piedmont beaver ponds. American Midland Naturalist **141:** 158-171.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1988. The Birder's Handbook: a field guide to the natural history of North American Birds. Simon and Schuster Inc.: New York, NY.

Faaborg, J., M. Brittingham, T. Donovan, and J. Blake. 1995. Habitat fragmentation in the temperate zone. Pages 357- 380 in T. E. Martin and D. M. Finch, editors. Ecology and Management of Neotropical Migratory Birds. Oxford University Press: New York.

Forsyth, D. J., and D. A. Smith. 1973. Temporal variation in home ranges of eastern chipmunks (*Tamias striatus*) in a southeastern Ontario woodlot. American Midland Naturalist **90:** 107-117.

Franzreb, K. E. and K. V. Rosenberg. 1997. Are forest songbirds declining? Status assessment from the southern Appalachians and northeastern forests. Transactions. 62nd North American Wildlife and Natural Resources Conference **62:** 264-278.

Freemark, E. F. and H. G. Merriam. 1986. Importance of area and habitat heterogeneity to bird assemblages in temperate forest fragments. Biological Conservation **36:** 115-141.

Freemark, K., and B. Collins. 1992. Landscape ecology of birds breeding in temperate forest fragments. Pages 443-454 in J. M. Hagan III and D. W. Johnston, editors. Ecology and Conservation of Neotropical Migrant Landbirds. Smithsonian Institute Press: Washington, D.C.

Gates, J. E. and L. W. Gysel. 1978. Avian nest dispersion and fledging success in field-forest ecotones. Ecology **59:** 871-883.

Grover, A. M. and G. A. Baldassarre. 1995. Bird species richness within beaver ponds in south-central New York. Wetlands **15:** 108-118.

Hamel, P. B., W. P. Smith,, D. J. Twedlt, J. R. Woehr, E. Morris, R. B. Hamilton, and R. J. Cooper. 1996. A Land Manager's Guide to Point Counts of Birds in the Southeast. General Technical Report SO-120. U. S. Forest Service, Southern Research Station, Asheville, NC.

Hefner, J. M., B. O. Wilen, T. E. Dahl, and W. E Frayer. 1994. Southeast Wetlands; Status and Trends, Mid 1970's to Mid 1980's. U.S. Department of the Interior, Fish and Wildlife Service, Atlanta, Georgia.

Heske, E. J. 1995. Mammalian abundances in forest-farm edges versus forest interiors in southern Illinois: is there an edge effect? Journal of Mammology **76:** 562-568.

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.

Hunter, W. C., R. Katz, D. N. Pahley, and R. P. Ford. 1999. Partners In Flight southern Blue Ridge bird conservation plan (physiographic area 23): version 1.0. In Press.

Hunter, W. C., M. F. Carter, D. N. Pashley, and K. Barker. 1993. Neotropical migratory landbird species and habitats of special concern within the southeast region. Pages 159-171 in D. M. Finch and P. W. Stangel, editors. Status and Management of Neotropical Migratory Birds; September 21-25, 1992, Estes Park, Colorado. General Technical Report RM-229. U. S. Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

James, F. C. and H. H. Shugart Jr. 1970. A quantitative method of habitat description. Audubon Field Notes **24:** 727-736.

Johnson, D. H. 2001. Habitat fragmentation effects on birds in grasslands and wetlands: a critique of our knowledge. Great Plains Research **11:** 211-230.

Johnson, R. G. and S. A. Temple. 1990. Nest predation and brood parasitism of tall grass prairie birds. Journal of Wildlife Management **54:** 106-111.

Johnston, C. A. and R. J. Naiman. 1990. Browse selection by beaver: effects of riparian forest composition. Canadian Journal of Forestry **20:** 1036-1043.

Kantrud, H. A. and R. E. Stewart. 1984. Ecological distribution and crude density of breeding birds on prairie wetlands. Journal of Wildlife Management **48:** 426-437.

Lawton, J. H. and C. G. Jones. 1995. Linking species and ecosystems: organisms as ecosystem engineers. Pages 141-150 in C. G. Jones and J. H. Lawton, editors. Linking species & ecosystems. Chapman & Hall: New York.

Lee, D. S. and A. W. Norden. 1996. The distribution, ecology and conservation needs of bog turtles, with special emphasis on Maryland. The Maryland Naturalist **40:** 7-46.

Lee, D. S. and B. Browning. 1998. Conservation concerns related to avian endemism in the southern Appalachians. *In Press.*

Legrand, H. E. Jr., S. P. Hall, and J. T. Finnegan. 2001. Natural Heritage Program list of the rare animal species of North Carolina. North Carolina Natural Heritage Program, Division of Parks and Recreation, N.C. Department of the Environment and Natural Resources.

Lenat, D. 1998. Evaluation of Christmas tree farming and cattle grazing on water quality in the New River basin, Ashe and Alleghany counties. Internal report to N.C. Department of Environmental Health and Natural Resources.

Lochmiller, R. L. 1979. Use of beaver ponds by southeastern woodpeckers in winter. Journal of Wildlife Management **43:** 263-266.

Lopez de Casenave, J., J. P. Pelotto, S. M. Caziani, M. Mermoz, and J. Protomastro. 1998. Responses of avian assemblages to a natural edge in a Chaco semiarid forest in Argentina. Auk **115:** 425-435.

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.

MacArthur, R. H. 1972. Geographical Ecology. Harper and Row: New York.

Martin, P S. 1984. Prehistoric overkill: the global model. Pages 354-403 in P.S. Martin and R. G. Klein, editors. Quaternary Extinctions. University of Arizona Press: Tucson, AZ.

McDowell, D. M. and R. J. Naiman. 1986. Structure and function of a benthic invertebrate stream community as influenced by beaver *(Castor canadensis)*. Oecologia **68:** 481-489.

McGrath, C. and P. Summer. 1992. Beaver: *Castor Canadensis*. Wildlife Profiles. Division of Conservation Education, North Carolina Wildlife Resources Commission.

Medin, D. E., and W. P. Clary. 1990. Bird populations in and adjacent to a beaver pond ecosystem in Idaho. U.S. Forest Service Research Paper INT-432. U.S. Forest Service, Intermountian Research Station, Orgden, Utah.

Medin, D. E., and W. P. Clary. 1991. Small mammals of a beaver pond ecosystem and adjacent riparian habitat in Idaho. U.S. Forest Service Research Paper INT-445. U.S. Forest Service, Intermountian Research Station, Orgden, Utah.

Mikusinski G., M. Gromadzki, and P. Chylarecki. 2001. Woodpeckers as indicators of forest bird diversity. Conservation Biology **15:** 208-217.

Murdock, N. A. 1994. Rare and endangered plants and animals of southern Appalachian wetlands. Water, Air and Soil Pollution **77:** 385-405.

Naiman R. J., Johnston C A, and Kelley J. C. 1988. Alteration of North American streams by beaver. Bioscience **38:** 753-762.

Norris, D. R., B. J. M. Stutchbury. 2001. Extraterritorial movements of a forest songbird in a fragmented landscape. Conservation Biology **15:** 729-736.

North Carolina Wildlife Resources Commission. 2002. Hunting and Trapping Regulations.

Noss, R. F., E. T. LaRoe III, and J. M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biological Report 28, USDI National Biological Service, Washington, D. C., USA.

Novak, M. 1987. Beaver. Pages 283-312 in M. Novak, J. A. Baker, M. E. Obbard, and B. Mallock, editors. Wild furbearer management and conservation in North America. Minister of Natural Resources: Ontario.

Odum, E. P. 1950. Bird populations of the highlands (North Carolina) plateau in relation to plant succession and avian invasion. Ecology **31:** 587-605.

Ortega, K. P. 1998. Cowbirds and Other Brood Parasites. University of Arizona Press: Tucson.

Outwater, A. 1996. Water: a natural history. Harper Collins Publishers, Inc.: New York.

Pagen, R. W., F. R. Thompson III, and D. E. Burhans. 2000. Breeding and post-breeding habitat use by forest migrant songbirds in the Missouri Ozarks. The Condor **102:** 738-747.

Pollock, M. M., R. J. Naiman, H. E. Erickson, C. A. Johnston, J. Pastor, and G. Pinay. 1995. Beaver as engineers: influences on biotic and abiotic characteristics of drainage basins. Pages117-126 in C. G. Jones and J. H. Lawton, editors. Linking species & ecosystems. Chapman & Hall: New York.

Ralph, C.J., S. Droege, and J.R. Sauer. 1995. Managing and monitoring birds using point counts: standards and applications. In C.J. Ralph, J.R. Sauer, S. Droege, eds. 1995. Monitoring Bird Populations by Point Counts. Gen. Tech. Rep. PSW-GTR-149. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

Rappole, J. H. 1995. The ecology of Migrant birds: a Neotropical perspective. Smithsonian Institution Press: Washington, D. C.

Reese, K. P. and J. D. Hair. 1976. Avian species diversity in relation to beaver pond habitats in the piedmont region of South Carolina. Proc. Annu. Conf. Southeast Assoc. of Fish and Wildl. Agencies **30:** 437-447.

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

Robinson, S. K., F. R. Thompson III, T. M. Donovan, D. R. Whitehead, and J. Faaborg. 1995. Regional forest fragmentations and the nesting success of migratory birds. Science **267:** 1987-1990.

Rodriguez, J. P. 2002. Range contraction in declining North American bird populations. Ecological Applications **12:** 238-248.

Rossell Jr., C R. 2001. Song perch characteristics of Golden-winged warblers in a mountain wetland. Wilson Bulletin **113:** 246-248.

Rossell, C. R. Jr., I. M. Rossell, J. W. Petranka, and K. K. Moorhead. 1995. Characteristics of a partially disturbed southern Appalachian forest-gap bog complex. Virginia Museum of Natural History **3:** 81-89.

SAMAB (Southern Appalachian Man and Biosphere). 1996. The Southern Appalachian Assessment (SAA) Terrestrial Report: Report 5 of 5. U. S. Forest Service, Southern Region, Atlanta, GA.

SAS Institute. 2000. SAS Version 8.1. SAS Institute Inc. Cary, NC.

Sauer, J. R., J. E. Hines, G. Gough, I. Thomas, and B. G. Peterjohn. 2001. [database on the internet] The North American Breeding Bird Survey, Results and Analysis 1966-2000. Version 2001.2, USGS Patuxent Wildlife Research Center, Laurel Maryland. *http://www.mbr-pwrc.usgs.gov/bbs/bbs.html*. Accessed 2001 July 12.

Shafer, D. S. 1984. Late-Quaternary paleoecologic, geomorphic, and climatic history of Flat Laurel Gap, Blue Ridge Mountains, North Carolina. Masters Thesis. The University of Tennessee, Knoxville.

Shafer, D. S. 1986. Flat Laruel Gap bog, Pisgah Ridge, North Carolina: Late Holocene development of a high-elevation heath bald. Castanea **51:** 1-10.

Sidebottom, J. R. 2000. Evaluation of the effect of Christmas tree production in western North Carolina on surface water quality. Report to the North Carolina Cooperative Extension Service and U.S. Environmental Protection Agency.

Simons, T. S., J. Lichstein, K. Weeks, and K. E. Franzreb. 2000. The effects of landscape pattern, core areas, and the forest management practices of avian communities in the southern Appalachians: 1999 Annual Report. Annual Report to the National Park Service.

Simons, T. S., K. N. Rabenold, D. A. Buehler, J. A. Collazo, and K. E. Franzreb. 1999. The role of indicator species: Neotropical migratory birds. Pages 187-208 in J. D. Peine, editor. Ecosystem Management for Sustainability: Principles and Practices Illustrated by a Regional Biosphere Reserve Cooperative. Lewis Publishers: New York, NY.

Simpson, M. B. Jr. 1992. Birds of the Blue Ridge Mountains. University of North Carolina Press: Chapel Hill, NC.

Snodgrass, J. W. 1997. Temporal and spatial dynamics of beaver-created patches as influence by management practices in a south-eastern North American landscape. Journal of Applied Ecology **34:** 1043-1056.

Suarez A. V., K. S. Pfennig, and S. K. Robinson. 1997. Nesting success of a disturbance-dependent songbird on different kinds of edges. Conservation Biology **11:** 928-935.

Tabachnick B G and Fidell L S. 1983. Using Multivariate Statistics. Harper and Row Publishers: New York, NY.

Temple, S. A. and J. R. Cary. 1988. Modeling dynamics of habitat-interior bird populations in fragmented landscapes. Conservation Biology **2:** 340-347.

Thompson, F. R. III and R. M. DeGraaf. 2001. Conservation approaches for woody, early successional communities in the eastern United States. Wildlife Society Bulletin **29:** 483-494.

Trani, M. K., R. T. Brooks, T. L. Schmidt, V. A. Rudis, and C. M. Gabbord. 2001. Patterns and trends of early successional forests in the eastern United States. Wildlife Society Bulletin **29:** 413-424.

VanRees-Siewert, K. L. and J. J. Dinsmore. 1996. Influence of wetland age on bird use of restored wetlands in Iowa. Wetlands **16:** 577-582.

Ward, H. T. 1990. The bull in the North Carolina buffalo. Southern Indian Studies **39:** 19-30.

Weakley, A. S. and M. P. Shafale. 1994. Non-alluvial wetlands of the southern Blue Ridge - diversity in a threatened ecosystem. Water, Air and Soil Pollution **77:** 359-383.

Webster, J. R., J. B. Waide, and B. C. Patten. 1975. Nutrient recycling and the stability of ecosystems. Pages 1-27 in F. G. Howell, J. B. Gentry, and M. H. Smith, editors. Mineral cycling in the southeastern ecosystems. Energy Research and Development Administration Symposium Series CONF-740513, Oak Ridge, TN.

Weigl, P. D. and T. W. Knowles. 1995. Megaherbivores and southern Appalachian grass balds. Growth and Change **26:** 365-382.

Whitcomb, R. F., C. S. Robbins, J. F. Lynch, B. L. Whitcomb, M. K. Klimkiewicz and D. Bystrak. 1981. Effects of forest fragmentation on avifauna of the eastern deciduous forest. Pages 125-205 in R. L. Burgess and D. M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag: New York, NY.

Whitehouse, N. J., G. B. Boswijk, and P.C. Buckland. 1997. Peatlands, Past, Present and Future; Some Comments from the Fossil Record. Pages 54-64 in L. Parkyn, R. E. Stoneman, and H. P. Ingram, editors. Conserving Peatlands. CAB International: New York, NY.

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

Wilcove, D. S. and S. K. Robinson. 1990. The impact of forest fragmentation on bird communities in eastern North America. Pages 319-331 in A. Keast, editor. Biogeography and ecology of forest bird communities. SPB Academic Publishing: The Hague, The Netherlands.

Woodward, R. T. and Y. Wui. 2001. The economic value of wetland services: a meta-analysis. Ecological Economics **37:** 257-270.

Zannette, L., P. Doyle, and S. M. Tremont. 2000. Food shortage in small fragments: evidence from an area-sensitive passerine. Ecology **81:** 1654-1666.

Zar, J.H. 1999. Biostatistical Analysis. Upper Saddle River, NJ: Prentice Hall, Inc.

APPENDIX

Bird species observed in all southern Appalachian wetland study sites

Common name	*Scientific name*	*Number of sites where observed (%)*
American Bittern [a]	*Botaurus lentiginosus*	1 (1.75)
Green Heron [a]	*Butorides virescens*	1 (1.75)
Turkey Vulture	*Cathartes Aura*	4 (7.02)
Wood Duck	*Aix sponsa*	2 (3.51)
Mallard	*Anas platyrhynchos*	1 (1.75)
Red-tailed Hawk	*Buteo jamaicensis*	1 (1.75)
Ruffed Grouse	*Bonasa umbellus*	1 (1.75)
Northern Bobwhite [ab]	*Colinus virginianus*	1 (1.75)
Killdeer [a]	*Charadrius vociferus*	1 (1.75)
Mourning Dove [a]	*Zenaida macroura*	6 (10.53)
Chimney Swift [ab]	*Chaetura pelagica*	1 (1.75)
Ruby-throated Hummingbird	*Archilocus colubris*	11 (19.30)
Belted Kingfisher [a]	*Ceryle alcyon*	10 (17.54)
Red-bellied Woodpecker	*Melanerpes carolinus*	6 (10.53)
Yellow-bellied Sapsucker [c]	*Sphyrapicus varius*	3 (5.26)
Downy Woodpecker [b]	*Picoides pubescens*	12 (21.05)
Hairy Woodpecker	*Picoides villosus*	11 (19.29)
Northern Flicker [ab]	*Colaptes auratus*	2 (3.51)
Pileated Woodpecker	*Dryocopus pileatus*	3 (5.26)
Eastern Wood-pewee [abc]	*Contopus virens*	12 (21.05)
Acadian Flycatcher [bc]	*Empidonax virescens*	8 (14.04)
Alder Flycatcher [c]	*Empidonas alnorum*	14 (24.56)
Willow Flycatcher [a]	*Empidonax traillii*	9 (15.79)
Least Flycatcher [a]	*Empidonax minimus*	9 (15.79)
Eastern Phoebe [b]	*Sayornis phoebe*	21 (36.84)
Great-crested Flycatcher	*Myiarchus tyrannulus*	2 (3.51)
White-eyed Vireo	*Vireo griseus*	4 (7.02)
Blue-headed Vireo [c]	*Vireo solitarius*	16 (28.07)
Red-eyed Vireo	*Vireo olivaceous*	35 (61.40)
Blue Jay [ab]	*Cyanocitta cristata*	11 (19.30)
American Crow	*Corvus brachyrhynchos*	21 (36.84)
Tree Swallow	*Tachycineta bicolor*	1 (1.75)
Bank Swallow	*Riparia riparia*	1 (1.75)
Barn Swallow	*Hirundo rustica*	1 (1.75)
Carolina Chickadee [a]	*Poecile carolinensis*	32 (56.14)
Tufted Titmouse	*Baeolophus bicolor*	28 (49.12)
Red-breasted Nuthatch [c]	*Sitta canadensis*	2 (3.51)
White-breasted Nuthatch	*Sitta carolinensis*	10 (17.54)
Brown Creeper [c]	*Certhia americana*	1 (1.75)
Carolina Wren [b]	*Thryothorus ludovicianus*	17 (29.82)

House Wren	*Troglodytes aedon*	9 (15.79)
Winter Wren [c]	*Troglodytes toglodytes*	1 (1.75)
Golden-crowned Kinglet [c]	*Regulus satrapa*	7 (12.28)
Blue-gray Gnatcatcher [b]	*Polioptila caerulea*	6 (10.53)
Eastern Bluebird	*Sialia sialius*	22 (38.60)
Veery [a]	*Catharus fuscescens*	4 (7.02)
Wood Thrush [abc]	*Hylocichlia mustelina*	12 (21.05)
American Robin [b]	*Turdus migratorius*	25 (43.86)
Gray Catbird [bc]	*Dumetella carolinensis*	40 (70.18)
Northern Mockingbird [a]	*Mimus polygottos*	2 (3.51)
Brown Thrasher [a]	*Toxostoma rufum*	8 (14.04)
European Starling [a]	*Sturnus vulgaris*	9 (15.79)
Cedar Waxwing	*Bombbycilla cedrorum*	31 (54.39)
Golden-winged Warbler [ac]	*Vermivora chrysoptera*	3 (5.26)
Northern Parula [c]	*Parula americana*	21 (36.84)
Yellow Warbler	*Dendroica petechia*	8 (14.04)
Chestnut-sided Warbler [ac]	*Dendroica pensylvanica*	17 (29.82)
Black-throated Blue Warbler [c]	*Dendroica caerulescens*	14 (24.56)
Yellow-rumped Warbler	*Dendroica coronata*	1 (1.75)
Black-throated Green Warbler	*Dendroica virens*	2 (3.51)
Blackburnian Warbler [c]	*Dendroica fusca*	2 (3.51)
Pine Warbler	*Dendroica pinus*	3 (5.26)
Prairie Warbler [ab]	*Dendroica discolor*	3 (5.26)
Black-and-white Warbler [b]	*Mniotilta varia*	10 (17.54)
American Redstart	*Setophaga ruticilla*	5 (8.77)
Worm-eating Warbler [c]	*Helmitheros vermivora*	2 (3.50)
Swainson's Warbler [bc]	*Limnothylpis swainsonii*	1 (1.75)
Ovenbird [b]	*Seiurus aurocapillus*	17 (29.82)
Louisiana Waterthrush [c]	*Seiurus motacilla*	7 (12.28)
Common Yellowthroat [ab]	*Geothylpis trichas*	36 (63.16)
Hooded Warbler [c]	*Wilsonia citrina*	22 (38.60)
Canada Warbler [ac]	*Wilsonia canadensis*	3 (5.26)
Yellow-breasted Chat [b]	*Icteria virens*	7 (12.28)
Scarlet Tanager [abc]	*Piranga olivacea*	15(26.32)
Eastern Towhee [ab]	*Pipilo erythrophthalmus*	38 (66.67)
Chipping Sparrow [b]	*Spizella passerina*	10 (17.54)
Field Sparrow [ab]	*Spizella pusilla*	13 (22.81)
Song Sparrow [ab]	*Melospiza melodia*	41 (71.93)
White-throated Sparrow [a]	*Zonotrichia albicollis*	2 (3.51)
Dark-eyed Junco [ac]	*Junco hyemalis*	16 (28.07)
Northern Cardinal	*Cardinalis cardinalis*	37 (64.91)
Rose-breasted Grosbeak [a]	*Pheucticus ludovicianus*	1 (1.75)
Indigo Bunting [ab]	*Passerina cyanea*	42 (73.68)
Red-winged Blackbird [a]	*Agelaius phoeniceus*	21 (36.84)
Eastern Meadowlark [ab]	*Sturnella magna*	5 (8.77)

Common Grackle	*Quiscalus quiscula*	4 (7.02)
Brown-headed Cowbird [a]	*Molothrus ater*	3 (5.26)
House Finch	*Carpodacus mesicanus*	6 (10.53)
American Goldfinch	*Carduelis tristis*	36 (63.16)
House Sparrow [ab]	*Passer domesticus*	1 (1.75)

[a] Displaying significant declining population trends throughout the species' breeding range by the Breeding Bird Survey (BBS) (Sauer et al. 2001).

[b] Displaying a moderate or significant decline in the Partners in Flight (PIF) southern Blue Ridge (Physiographic Region 23) (Carter et al. 2000, Hunter et al. 1999) or displaying significant declining population trends in the BBS Blue Ridge region (Sauer et al. 2001).

[c] Considered a PIF priority species in the southern Blue Ridge (Physiographic Region 23) (Carter et al. 2000, Hunter et al. 1999), or a species of local concern in the southern Appalachians by the North Carolina Natural Heritage Program (NCNHP) (LeGrand et al. 2001), Hunter et al. (1993) or Lee and Browning (1998).

VITA

Jason Frederick Bulluck was born on July 9, 1976 in Rocky Mount, North Carolina. He attended Edgecombe County public schools, graduating in May of 1994, from Southwest Edgecombe high school.

Entering Appalachian State University in the following September, Jason began undergraduate study in Biology. During this time, he gained a fervent interest in conservation biology. While taking a field course in Ornithology, Dr. Matthew Rowe inspired Jason to *do* science in order to achieve conservation. Jason decided, in short, that avian ecology studies were an invaluable tool for both understanding the impact of humans on the natural world, as well as establishing conservation priorities for habitats throughout the world.

Immediately after earning his Bachelors of Science in Biology/Environmental Studies from A.S.U. in Spring of 1999, Jason began graduate research towards a Masters of Science in Biology at Appalachian that focused conservation of breeding bird habitats. During this time, Jason was awarded several grants for his thesis research. He also became a member of the Association of Field Ornithologists, the American Ornithologists Union, and the Society of Wetland Scientists, and presented his research at meetings of various research groups. Also during his Masters studies, Jason conducted contract GIS-related research assessing effects of human land use on bird populations in Ohio and used GIS to further explore his thesis topic from a landscape perspective. Jason was awarded his M.S. in Biology in May, 2003.

Jason Bulluck currently resides in Knoxville, TN. His parents are Mr. Thomas Bulluck and Mrs. Kaye Bulluck of Rocky Mount, North Carolina. His wife-to-be is Miss Lesley Penfield, also a graduate of the Appalachian State University Biology program.