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Status of wintering grassland birds in a post-hurricane, salvage-logged forest

Matthew Edward Brooks

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STATUS OF WINTERING GRASSLAND BIRDS IN A POST-HURRICANE, SALVAGE-
LOGGED FOREST

A Thesis

Submitted to the Graduate Faculty of
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements of the degree of
Master of Science

in

The School of Renewable Natural Resources

by

Matthew Edward Brooks
B.S., University of North Carolina Asheville, 2002
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ABSTRACT

North American grassland birds have been declining at an alarming rate. Winter habitat for grassland species in the southeastern U.S. generally occurs within forests subject to management. I studied wintering grassland bird communities in De Soto National Forest in southern Mississippi. My objectives were to: 1) Assess bird communities and vegetation structure in upland forest stands that were salvage-logged following Hurricane Katrina, in stands managed for Red-cockaded Woodpeckers (*Picoides borealis*, RCW), and in stands containing bogs; and 2) Determine the extent of the association between vegetation and occurrence of the common wintering grassland birds. I conducted bird and vegetation surveys in 27 stands over two winters. The wintering grassland bird community included Bachman's Sparrow (*Aimophila aestivalis*), Henslow's Sparrow (*Ammodramus henslowii*), and Sedge Wren (*Cistothorus platensis*). Forest-stand associations varied among species. Bachman's Sparrows occurred only in upland and RCW-managed stands, Henslow's Sparrows occurred only in bogs and RCW-managed stands, and Sedge Wren occurred in all stand types. None of these species' densities was statistically different between salvaged and unsalvaged stands. There were no statistically significant differences in total grassland bird density or species richness among stand types. Henslow's Sparrow use of RCW stands was mostly ephemeral, but our data were consistent with previous studies suggesting that RCW management benefits grassland birds. Spatially uniform, dense herbaceous cover, and cover of *Scleria muhlenbergii*, a preferred food item, best predicted Henslow's Sparrow stand occupancy. Increased woody understorey vegetation and decreased tree density best predicted Sedge Wren occupancy. I recommend management practices focusing on small-scale herbaceous ground-layer restoration in bogs, an increase in the number of RCW clusters, and the thinning of trees in dense stands.

CHAPTER 1. WINTERING GRASSLAND BIRD COMMUNITIES IN THREE MANAGED-FOREST STAND TYPES FOLLOWING POST-HURRICANE KATRINA SALVAGE LOGGING IN DE SOTO NATIONAL FOREST, MS, USA

INTRODUCTION

North American grassland bird populations have declined over the last four decades as a result of habitat loss (Herkert 1994a). Grassland ecosystems in North America have been reduced by approximately 80% since the 1800s, mostly because of conversion to agricultural land and forests following suppression of natural fire regimes and native grazers (Knopf 1994, Noss et al. 1995, Askins 2000). These changes have altered grassland bird communities and decreased regional bird abundances. This dramatic decline in grassland bird abundance has sparked conservation concern and promulgated the need for research on grassland birds on both their breeding and wintering grounds.

The endangered longleaf pine (*Pinus palustris*) ecosystem was once the primary grassland habitat for much of the southeastern United States. This habitat type historically covered 37 million ha, about 2% of the land area of North America (Wahlenberg 1946), and ranged from Virginia to east Texas. A fire-dependent ecosystem, it has been reduced to less than 3% of the original area, and much of this consists of degraded, shrub-encroached lands lacking the diverse herbaceous layer that is an integral part of the system (Outcalt and Sheffield 1996, Gilliam and Platt 2006). It is this unique herbaceous layer, comprised of numerous species of grasses, sedges, and forbs, that provides habitat for multiple species of grassland birds during winter. Most remaining longleaf pine stands are small, isolated, management-dependent fragments—mere glimpses of this once wide-spread ecosystem (Frost 2006).

Some of the larger remaining longleaf pine tracts occur in the De Soto Ranger District of De Soto National Forest (DSNF) in southeastern Mississippi. De Soto National Forest covers approximately 153,780 ha and is the largest national forest in the state. It is a mosaic of upland

longleaf and slash pine (*Pinus elliottii*) forests and savannas and other forest types. Longleaf and slash pine stands make up 44% and 23% of the forest, respectively (Windham 2005); however, much of the forest has suffered from severe shrub encroachment and unnatural tree densities, which have led to a loss of the herbaceous layer. The forest is subject to multiple management practices aimed at forest restoration, timber production, and the protection of endangered species. Examples include the prescribed fire program, logging, and Red-cockaded Woodpecker (*Picoides borealis*; “RCW” hereafter) management.

The majority of grassland habitats in DSNF can be divided into three distinct stand types based on differences in natural geography, topography, and management practices. These are: 1) upland longleaf and slash pine stands; 2) upland longleaf pine stands managed for RCWs; and 3) hillside seepage pitcher plant bogs. I will refer to these three stand types as simply upland, RCW, and bog stands. Red-cockaded Woodpecker clusters are an artificially designated stand type, while upland and bog stand types, although subject to multiple management practices, are naturally occurring and well documented in the literature (Means and Moler 1979, Clewell 1986, Brooks et al. 1993, Olson and Platt 1995). The majority of DSNF consists of upland pine stands or non-savanna forest types; bogs and RCW clusters constitute only a small portion of the total grassland habitat. Although I use the term ‘grassland’ to refer to areas with a well-developed herbaceous layer, regardless of canopy, the longleaf pine habitat in DSNF is actually longleaf–slash pine savannas or forests, depending on the dominant tree species and density. Here, I use ‘grassland’ or ‘pine savanna’ to refer to all habitat types with a substantial herbaceous layer, not just treeless areas.

Hurricane Katrina, a category four storm, passed over DSNF on August 29, 2005 and caused damage to most of the mature upland pine stands in the forest. An estimated 117,000 ha of forest stands were damaged from sustained winds exceeding 230 km/hr (Meeker et al. 2005).

The most extensive damage occurred along roads, streams, and power line right-of-ways where trees were exposed to the most wind (Lee and Smith 2005). De Soto N.F. was opened up to widespread salvage-logging operations during the fall and winter following Hurricane Katrina because of the risk of tree-pests and tree-disease outbreaks in damaged trees, the danger of catastrophic fire from increased fuel loads, and the disruption of the prescribed fire program essential for restoring and maintaining pine savannas (Bradford 2005). Most of the mature upland and RCW stands were salvaged; bog stands were not opened to salvage operations (Gainey and James 2005, D. L. Tyron, USFS biologist, De Soto N.F., pers. comm.).

There are many ecological repercussions that can affect grassland bird habitats following logging in pine savannas. Noticeable results of the timber destruction in DSNF are changes in canopy closure and tree density. Numerous studies have shown the relationship among canopy closure, tree density, and herbaceous plant diversity in longleaf pine savannas. Lower tree densities lead to increased herbaceous plant diversity and biomass by reducing resource competition and increasing sunlight availability (Brewer 1998, Harrington and Edwards 1999, McGuire et al. 2001, Harrington et al. 2003, Gilliam et al. 2006, Platt et al. 2006). Lower tree densities may also indirectly discourage the establishment of undesirable bird-dispersed woody shrubs by decreasing the number of perches for seed-dispersing birds (Brewer 1998, 2002, Hinman et al. 2008). An open canopy, however, may also encourage growth of some woody plant species by reducing competition for light and moisture (Harrington and Edwards 1999). Another observable result of the downed timber and subsequent logging in DSNF was the temporary reduction in shrubs (D. L. Tyron, pers. comm.). These shrubs established in the absence of fire and grazing and are very difficult to remove even with resumed fire, persisting year after year from roots (Boyer 1992, Olson and Platt 1995, Drewa et al. 2002b). Fire will reduce the shrubs for a growing season, but the shrubs tend to return as dense or denser than

before within two growing seasons, particularly when dormant season fires are used (Hodgkins 1958, Olson and Platt 1995, Drewa et al. 2002b). Soil compaction from large machines used to remove salvaged timber can affect herbaceous plant species composition by prohibiting the growth of certain species while encouraging the growth of disturbance-tolerant annuals like *Panicum verrucosum* (warty panicgrass; Plentovich et al. 1999). Brewer (2002) showed that disturbance removing standing dead vegetation increases seedling emergence of *Ilex glabra* (gallberry), a native but invasive shrub common in pine savannas. All of these potential changes in the herbaceous layer can negatively affect grassland birds.

Red-cockaded Woodpecker breeding clusters are managed in ways that create or improve grassland bird habitat. Concern over the impact of single-species management on non-target species has sparked interest in the effects of RCW management on other organisms (Hunter et al. 1994, Brennan et al. 1995, Provencher et al. 2002). Several studies have shown that stands managed for RCWs contain different bird communities than unmanaged stands and have higher densities of grassland-dependent birds (Wilson et al. 1995, Conner et al. 2002, Provencher et al. 2002, Wood et al. 2004). Although many of these studies have assessed the use of RCW stands by other bird species in winter, they all relied upon point count sampling methods. Point counts are appropriate in the breeding season, but most species of wintering grassland birds in the southeastern U.S. exhibit silent, inconspicuous behavior over winter. Red-cockaded Woodpeckers are sensitive to midstorey woody vegetation, and the removal of this vegetation—via mowing, herbicide application, prescribed fire, or a combination of these treatments—is one of the primary management tools for the woodpecker (Sparks et al. 1999, Beaty 2003, Rudolph et al. 2004). The removal of woody plants, particularly when combined with repeated growing season prescribed fires, tends to create patches of open grasslands characterized by low densities of ground-level and midstorey woody vegetation (Conner et al. 2002). Masters et al. (1996)

showed that sites managed for RCWs in pine–oak forests in Arkansas had on average 5, 7, 2–10, and 4–9 times higher standing crops of grasses, sedges, forbs, and woody vines, respectively, than non-managed stands. Red-cockaded Woodpecker stands in Mississippi had higher grass and forb cover than traditionally-managed pine stands (Wood et al. 2004).

Southeastern pitcher plant bogs, the third distinct type of grassland habitat in DSNF, are floristically and structurally different from upland pine savannas (Walker and Peet 1983, Platt et al. 1988, Drewa et al. 2002a). Bogs have higher plant species richness than drier, upland savannas. On a small scale (1 m²), southeastern pitcher plant bogs have the highest plant diversity of any ecosystem in the temperate zone (Peet and Allard 1993, Varner and Kush 2004). Topographic gradients create soil moisture and soil-type heterogeneity, leading to a change in plant species composition, richness, and biomass that is more obvious with herbaceous than woody species (Platt et al. 1988, Bridges and Orzell 1989, Kirkman et al. 2001, Drewa et al. 2002a).

Despite the vegetation and geological differences between bogs and upland longleaf pine savannas, many grassland and scrub bird species occur in both habitat types. For example, in Louisiana, Swamp Sparrows (*Melospiza georgiana*), Henslow's Sparrows (*Ammodramus henslowii*), Sedge Wrens (*Cistothorus platensis*), and Common Yellowthroats (*Geothlypis trichas*) are commonly found in boggy flatwoods, seepage bogs, and upland pine savannas (pers. obs.). To the best of my knowledge, the differences in grassland bird communities and bird abundances between these two distinct and often adjacent habitats have not been reported.

I examined the wintering grassland bird communities and vegetation in three different grassland stand types in DSNF over two winters. The main objectives of my study were to assess the differences in wintering grassland bird communities, vegetation structure, and plant species composition among upland, RCW, and bog stands and between salvaged and unsalvaged

stands. Several side objectives of my study were to compare two bird-sampling methods and to test for significant declines in bird abundances over winter. Many grassland birds are sensitive, declining species that are dependent on endangered and rare ecosystems requiring human management. It is important that we understand how various management practices affect grassland birds both directly and indirectly. I hope that this information will provide further information on how pine savannas can be managed for grassland birds.

METHODS

Study Site

De Soto N.F. (De Soto Ranger District) is located between 31°15' and 30°30' latitude, 88°45' and 89°24' longitude, and occupies seven counties in Mississippi (Fig. 1.1). De Soto N.F. falls within the East Gulf Coastal Plain ecoregion and the Coastal Flatlands and Coastal Plain Rolling Hills physiographic provinces (Peet and Allard 1993, Dorfman 2000). The forest consists of a mosaic of habitats including titi swamps, deciduous woodlands, baygalls, upland longleaf pine and slash pine savannas, and hillside seepage bogs. Many of these bogs have a dense overstorey of slash pine initially planted by the Civilian Conservation Corps and the Forest Service. These pines subsequently invaded many areas in the first half of the 20th century following fire suppression and the demise of the longleaf pine tree (Peet and Allard 1993, Brewer 1998, Hinman et al. 2008, C. J. Boykin, USDA Forest Service, De Soto N.F., pers. comm.).

De Soto N.F. contains numerous ecological communities. According to the Terrestrial Ecological Classification system, the ecological communities that make up DSNF are East Gulf Coastal Plain Interior Upland Longleaf Pine Woodland, East Gulf Coastal Plain Floodplain Forest, Southern Coastal Plain Seepage Swamp and Baygall, and East Gulf Coastal Plain Near Coast Pine Flatwoods (NatureServe 2004, 2009). The Forest Service terms these same regions

southern region R8 code 21, 22 or 31, laurel oak-willow oak R8 code 64, hardwood forest R8 code 64, and slash pine R8 code 22, respectively (Windham 2005). The pine communities are often referred to as dry, mesic, or wet longleaf/slash pine savannas/forests or pitcher plant bogs, depending on hydrology, soil, elevation, and plant species composition, and are collectively classified as Southern Longleaf Savanna (Peet and Allard 1993, Brewer 1998, Mississippi Natural Heritage Program 2006). According to Peet's (2006) classification system, the longleaf types in DSNF are xeric sand barrens and uplands, subxeric sandy uplands, and seeps. The majority of pine savanna understories in DSNF are dominated by the native, invasive shrubs *Ilex coriacea* (large gallberry), *I. glabra*, *I. vomitoria* (yaupon), and *Gaylussacia mosieri*; the dominant graminoids in these habitats are *Andropogon* spp. (broom sedge), *Ctenium aromaticum* (toothache grass), *Dichanthelium* spp. (rosette grass), *Muhlenbergia expansa* (cutover muhly), *Panicum* spp. (panic grass), *Schizachyrium* spp. (bluestem), and in wetter sites, *Rhynchospora* spp. (beaksedge). Typically, in pine savannas, the pyrogenic grasses *C. aromaticum* and *M. expansa* are abundant after fire, but are gradually replaced by *Andropogon* spp. and *Schizachyrium* spp. as time since fire increases (Bechtoldt and Stouffer 2005, Johnson 2006). In DSNF, at sites where shrub intrusion is a problem, the herbaceous layer may be almost completely replaced by woody shrubs within three years following fire (K. Coursey, pers. comm; pers. obs.).

The soils in DSNF are described as sandy loam, are generally acidic and nutrient poor, and developed from a mixture of loamy, clayey, and sandy coastal plain material (Pessin 1933, Pettry 1977). Many upland longleaf stands may appear dry with sandy soils, but are fairly moist due to the clay and loam content of the soil (Moore 1997).

The climate is classified as Humid Mesothermal (Murray 1961). Summers are hot and humid and winters are mild and wet with occasional, brief periods of freezing temperatures. The mean annual temperature is 18–20°C; the mean annual rainfall is 1422–1575 mm (Curtis 2005).

De Soto N.F. is subject to multiple forest management practices, most notably the prescribed fire program. In the early 1960s, after 30-plus years of fire suppression, the Forest Service began using prescribed fire in an attempt to restore the land to a historical condition and to enhance habitat for the many rare, threatened, and endangered species found in the area (C. J. Boykin and K. Coursey, pers. comm.). Currently, DSNF is subject to an approximate three-year fire rotation with 38,040–52,600 ha burned annually. Most prescribed fires (60–70% of fires) are applied in the non-growing season, which ranges approximately from the first frost in autumn through February. Only a few prescribed fires were set during 2006–2007 following Hurricane Katrina because of the increased fuel load and clogged fire breaks (Jarvis 2005, Bryant and Boykin 2007). Livestock grazing also played an important role in shaping the current DSNF landscape. Grazing has occurred in the region since the arrival of the earliest European settlers and was prevalent in DSNF until the last several decades; there is still one compartment in the forest open to cattle. Many of the more open areas of the forest today are places might have experienced more recent (< 20 years) cattle grazing, and although perhaps not completely ecologically sound, livestock grazing played an important role historically in limiting the intrusion of woody plants into savannas (C. J. Boykin, pers. comm.).

De Soto N.F. experienced wide-scale salvage logging following Hurricane Katrina. The loss of trees following the hurricane thinned stands with an efficiency that could not be matched by anthropogenic methods; an estimated 30% of the forest's saw-timber-size trees were damaged. Around 40,060 ha of forests were salvaged within a year after Katrina (Hurricane Katrina Tree Removal and Hazardous Fuels Treatment Project), totaling a removal of 218

million board feet of timber (Bryant and Boykin 2007). Although the salvage operations caused an initial reduction in shrub density—a benefit to grassland habitat maintenance—by the third growing season after salvaging, shrubs had returned into many salvaged stands. Soil compaction, an ecological concern resulting from timber harvest operations, may have occurred, but the majority of salvaged stands in DSNF show few signs of major soil disturbance (*e.g.*, log-skidder tracks; pers. obs.). The Forest Service did not allow salvage logging in bogs and encouraged logging companies to fill machine-made indentions in the soil (Gainey 2005, Gainey and James 2005, D. L. Tyron, pers. comm.). Moreover, there was less-than-average precipitation during the bulk of the salvage operations, and this may have reduced the sensitivity of the soil to mechanical disturbances (D. L. Tyron, pers. comm.).

Management in DSNF also includes habitat restoration for rare, threatened, and endangered species. Management plans are currently in effect for the federally endangered RCW and gopher tortoise (*Gopherus polyphemus*). These plans are focused primarily on the restoration of specific habitats and locations used by these animals. Management in RCW stands is primarily mowing and prescribing fire to reduce woody midstorey vegetation. Before Hurricane Katrina, there were 41 active RCW clusters in DSNF, but about half of the 150 RCW cavity trees were lost during Hurricane Katrina (Bryant and Boykin 2007), and all breeding clusters were salvage logged, although not during the breeding season (Bradford 2005). In DSNF, RCW patches range approximately $< 0.2\text{--}1$ ha.

De Soto N.F. has 805 documented pitcher plant bogs. The largest bog in the forest is 117 ha (Gainey 2005). These bogs contain numerous rare and regionally-endemic species of plants (Walker and Peet 1983). Some large bog stands are managed via growing season prescribed fire and hand removal of intrusive slash pines. Before the salvage operations began, 19,500 ha of forests were evaluated for proposed, endangered, threatened, and sensitive species (PETS) for

multiple pre-salvaging biological evaluations (Gainey and James 2005, Bryant and Boykin 2007). The Forest Service determined after a pre-salvage impact assessment that salvage operations would have no significant ecological impact on the forest (Bradford 2005).

Site Selection

I selected 16 study sites in August–November 2007 and 11 additional sites in September–November 2008, but only surveyed five sites both years. Only one bog stand was salvage logged, all RCW stands were salvaged, and all but one of the upland stands were salvaged (Table 1.1). The elevations of my study sites ranged from approximately 20–60 m. All of my study sites except one were located in the southern portion of the National Forest in Stone, Harrison, and Jackson counties (Fig. 1.1); this area had considerably more grassland habitat, including more managed RCW stands, than did the northern portion of the forest. My criteria for establishing sites were that a habitat patch must have > 50% herbaceous cover, < 50% shrub cover, and enough area to contain ≥ 100 m of transect. Stands that did not meet these criteria were typically dominated by an understorey of *Ilex* spp. and a dense overstorey of young pines, contained considerable bare ground, and were unlikely to contain grassland birds (pers. obs.). At all sites located in bog and upland stands, I established 100–400 m of 20-m-wide fixed-width transects, in proportion to the patch size; thus, the area sampled among these sites ranged from 0.2–0.8 ha. I oriented transects in random directions within the range of bearings that allowed for adequate transect length. Most sites consisted of one continuous transect; six sites had several shorter transects spaced 100 m apart. In RCW stands, I marked four corners around the managed clusters, making four-sided plots ranging from 0.2–0.96 ha. All sites sampled within the same year were > 500 m apart to assure independence among sites; sites < 500 m apart were not sampled the same year.

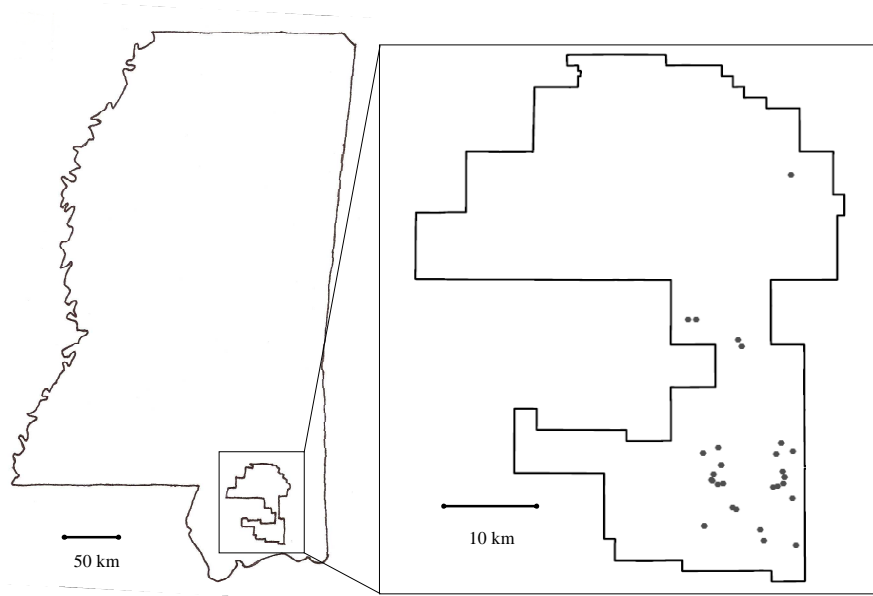


Figure 1.1. Map of Mississippi showing the location of DSNF and the 27 study sites used in this study.

Grassland Bird Sampling

I surveyed sites for wintering grassland birds from November 29th through February 23rd during the winters of 2007–2008 and 2008–2009. Sites were sampled between sunrise and sunset; no sampling was done in the rain. The first year I sampled all ten sites located in upland stands and five sites located in bog stands (plus one site in a stand I classified as “other”). The second year I sampled all six RCW stands, five additional bog stands, and the five bog stands from the previous year. In total, I sampled 27 different sites over two winters. I did not sample the upland sites the second year because of shrub encroachment following the first winter of sampling. By the second year of sampling, the upland sites no longer fit my herbaceous-cover > 50% and shrub-cover < 50% criteria for sampling. At most of the upland sites, the shrubs *Ilex coriacea* and *Ilex vomitoria* grew approximately 1.5 m between sampling years. Both of these species are native but invasive in DSNF (Brewer 2002), and are two of the dominant understorey shrubs in the forest. The bog stands sampled the first year also suffered from shrub

Table 1.1. Study sites in DSNF, MS and pertinent information. Information includes forest service stand and compartment classification, area sampled (ha), salvage status, stand type classification used in this study, the last year of prescribed fire treatment (all dormant season fires; year refers to before January 1st, so a winter fire in January 2008 is still assigned a fire year of 2007), the dominant tree species, and the stand age (years since harvest). The county, stand/compartment, fire year, dominant tree species, and stand age information is from GIS metadata compiled by the U.S. Forest Service.

Site code	County	Stand/ Compartment	Area sampled	Salvaged	Stand type	Fire year	Dominant tree species	Stand age
BAC	Jackson	20/525, 19/525	0.8	No	Other	2006	<i>P. elliotii</i>	56
BMX	Harrison	8/613	0.64	Yes	Upland	2008	<i>P. palustris</i>	71
BOG	Harrison	20/550	0.67	No	Bog	2007	<i>P. elliotii</i>	55
BOO	Jackson	1/510	0.8	Yes	Bog	2007	<i>P. palustris</i>	83
BYR	Harrison	7/520, 10/520	0.6	Yes	Upland	2004	<i>P. elliotii</i>	51
CAR	Stone	16/569, 26/529	0.44	No	Bog	2007	None	80
DAN	Stone	13/532	0.6	No	Bog	2007	<i>P. palustris</i>	88
GOB	Harrison	13/532, 12/532	0.28	No	Bog	2008	<i>P. palustris</i>	58
KAT	Harrison	1/567	0.6	Yes	Upland	2007	<i>P. palustris</i>	12
KES	Jackson	1/606	0.6	Yes	Upland	2007	<i>P. palustris</i>	79
LUV	Stone	13/628	0.28	No	Bog	2008	<i>P. elliotii</i>	72
MAR	Harrison	4/561	0.6	Yes	Upland	2008	<i>P. palustris</i>	82
MEL	Stone	33/636	0.6	No	Bog	2007	<i>P. elliotii</i>	54
NAN	Stone	9/628	0.28	No	Bog	2008	<i>P. elliotii</i>	92
PAN	Harrison	11/550	0.6	No	Bog	2007	<i>P. elliotii</i>	55
PIN	Jackson	10/509	0.72	Yes	Upland	2007	<i>P. palustris</i>	81
RCW	Harrison	7/567	0.8	Yes	Upland	2007	<i>P. palustris</i>	79
RCW01	Harrison	12/613	0.21	Yes	RCW	2008	<i>P. palustris</i>	71
RCW03	Harrison	34/567	0.2	Yes	RCW	2007	<i>P. palustris</i>	79
RCW15	Jackson	20/514	0.71	Yes	RCW	2008	<i>P. palustris</i>	79
RCW23	Harrison	27/542	0.96	Yes	RCW	2008	<i>P. palustris</i>	85
RCW27	Jackson	1/514	0.37	Yes	RCW	2008	<i>P. palustris</i>	79
RCW28	Harrison	15/560	0.25	Yes	RCW	2008	<i>P. palustris</i>	81
SHE	Perry	6/7, 21/7	0.6	No	Upland	2006	<i>P. palustris</i>	74
SUM	Jackson	13/522, 29/522	0.72	Yes	Upland	2007	<i>P. palustris</i>	79
TIG	Jackson	20/511	0.6	Yes	Upland	2008	<i>P. palustris</i>	80
ZAK	Harrison	26/581	0.2	No	Bog	2008	<i>P. elliotii</i>	41

encroachment, but large patches of habitat still meet my site-selection criteria. I sampled sites three times each winter, roughly once per month, except for the RCW stands, which were sampled twice, once in December and once in January. Upland sites BMX, MAR, and TIG were only sampled twice before they were burned in February 2008.

I surveyed grassland birds using a disturbance-based sampling method. I defined grassland birds as those considered grassland birds in the scientific literature. Although we surveyed all bird species, I restricted my analyses to grassland birds because these were my species of interest, and because these species are of higher conservation concern compared to most of the species detected during surveys. I sampled grassland birds using 20-m-wide fixed-width transects and a modified version of the protocols described by Carrie et al. (2002) and Texas Parks and Wildlife Project Prairie Bird (Shackleford et al. 2001). I refer to this modified method as the battue method. Protocol was as follows: Three people including myself, each with a 3-m-long, fiberglass Apache™ crappie fishing pole in each hand, lined up across a transect with our poles extended laterally from our sides. With each of us spaced apart so that the tips of our poles just overlapped, the distance between the tips of the two outermost poles was 20 m. We walked briskly along the transect using the poles to beat the grass and flush birds (a battue in bird-hunter terminology). A fourth person, carrying a 6-m mist net mounted on poles, walked behind the line of flushers. When a bird was flushed, it was identified to species if possible. If identification was uncertain, we attempted to capture the bird for identification following Bechtoldt and Stouffer (2005). Captured birds were banded with a numbered aluminum U.S. Fish & Wildlife Service band. I took standard morphological measurements, determined age when possible, and collected blood and feather samples. If a flushed bird could not be identified to species, it was classified into one of the following categories: unknown, sparrow sp., *Ammodramus* sp., or *Ammodramus* sp./Bachman's Sparrow. I walked the transect center between two other flushers for all surveys. In RCW stands, I used the same sampling method but made multiple, systematic, non-overlapping passes through the plot until it was completely sampled. I converted bird abundances to the number of birds per hectare of transect. Bird density estimates for each site (sample unit) are the averaged densities from repeated

sampling within a sampling year. At the end of each survey, we also made a tally of the number and species of all birds detected within a 120-m band centered on the transect.

Many grassland birds are inconspicuous and do not readily flush, instead staying concealed in the grass until a perceived threat comes near; without disturbing the ground vegetation, these birds can be very difficult to detect. Sampling using disturbance and fixed-area searches is more appropriate for some species of wintering grassland birds than single-person transects or point count surveys that rely on distance measures to extrapolate bird densities (Emlen 1971, Fletcher et al. 2000, Roberts and Schnell 2006). Studies conducted using only fixed point count surveys may have low-biased estimates of grassland bird abundances (Fletcher et al. 2000). A disturbance-based approach, also called flush netting, was pioneered by Chandler and Woodrey (1995) and has been used in numerous studies of wintering Henslow's Sparrows and other grassland birds. With 5–10 people conducting fixed-area searches in relatively open savannas, capture success is high.

The battue method is a modified flush-netting approach better suited to flush birds in small habitat patches with fewer people. Because the habitat patches I surveyed were smaller and had high densities of shrubs, I could not use the standard flush-netting approach with a line of five or more people. The high shrub cover and proximity to forest edge added to the difficulty of capturing birds because mist nets tended to stick to shrubs and flushed birds tended to fly into forests and shrub thickets. Moreover, the ability levels of the volunteers varied tremendously among sampling events, and this made identifying and capturing birds difficult during some surveys. My battue method had the advantage of requiring fewer volunteers, but because of this and the difficult terrain, fewer birds could be captured. Another advantage of the battue method is that it is easily repeatable and could be used in future studies.

Repeated samples like mine can allow for estimation of detection probabilities to correct occupancy and abundance estimates (MacKenzie et al. 2002, Royle and Nichols 2003). Unfortunately, I was unable to estimate detection probabilities using repeated-sampling techniques because several key assumptions underlying this method could not be met for my samples. One principal assumption to this technique is population closure: there can be no emigration or immigration during the survey period, constant or temporary, and death and birth rates must be equal. Temporary emigration or immigration (random movement) can occur when an organism's range exceeds the study area; constant emigration or immigration (nonrandom movement) can occur from high mortality, high birth rates, or true migration (Kendall 1999). When random movement occurs, estimates of occupancy or abundance within the study area will be biased high, but the estimates will be appropriate if applied to the overall local population, assuming it is closed (Kendall 1999). With nonrandom movement, however, estimates will be biased high whether applied to the study area or the general population (Kendall 1999). There are several options for dealing with nonrandom movement and lack of closure. One is to truncate the data so that surveys conducted after the nonrandom movement was detected are not included in the analysis. This method assumes that the population is closed for at least some of the surveys and that the time of closure violation can be determined. As an example, one can truncate all sample events that occurred after the last detection of the target organism (MacKenzie et al. 2002). Another option is to merge all but the first or last sampling events if emigration or immigration occurs, respectively (Kendall 1999); however, with this option, the detection probability is interpreted as the probability of detecting the organism on the first or last survey (MacKenzie et al. 2006). During the first year of sampling, I observed nonrandom movement (emigration or mortality) of Henslow's Sparrows and Sedge Wrens after the first round of surveys. This phenomenon eliminates truncation as an option for dealing with lack of

closure because the data set would be reduced to one sampling event. Pooling the last two samples not only changes the interpretation of the detection probability but also reduces the number of sampling events to two. A small sample of sites and few repeated surveys can lead to low precision of the detection probability estimate (MacKenzie et al. 2002). While the detection probabilities for my bird species are inevitably < 1 , I assume they were constant among study sites (*c.f.*, Tucker and Robinson 2003) and between years. Furthermore, the narrow widths of my transects may have increased my detection probabilities (Diefenbach et al. 2003), and repeated surveys also increase the chance of detecting rare species. Inevitably, my relative abundance estimates are biased low because of imperfect detection and because birds that could not be identified to species were excluded from analyses. I think, however, that because of the small width of my transects, and because I conducted repeated surveys, that my detection probabilities were consistent and independent of bird abundance, both of which are caveats for reliable abundance indices (Johnson 2008). My bird abundance estimates, while indices, are nonetheless representative of the abundance patterns in DSNF and thus have scientific relevance and value.

I compared abundance estimates between my battue method and the Chandler–Woodrey (1995) method of sampling. Because of detection probability issues and because my sampling method is novel, I wanted to compare bird abundance estimates determined using both survey methods for surveys conducted at the same study sites. For this, I sampled eight longleaf pine savannas in southeastern Louisiana in Tangipahoa and St. Tammany parishes for Henslow's Sparrows during the winters of 2007–2008 and 2008–2009. I sampled the second winter using both the battue and Chandler–Woodrey (1995) methods and compared the mean bird-densities of Henslow's Sparrows estimated from both methods. I used Henslow's Sparrow because this was the target species for the studies using the Chandler–Woodrey (1995) method, and it was the most numerous grassland bird at my study sites in both DSNF and Louisiana.

Vegetation Structure and Plant Species Composition Sampling

I measured habitat structure and plant species composition from mid December–early January during the winters of 2007–2008 and 2008–2009. For upland and bog stands, each 20-m-wide transect was partitioned into 20-m intervals, each interval was partitioned into four 10-by-10-m sections, and a random point was established within each 10-m section. If two random points fell within one meter of each other, a new random point was chosen. In each 10-m section, I measured canopy closure using a spherical densitometer (Lemmon 1956). In these 10-m sections, I also measured herbaceous density using a 2-m-tall, 3-cm-diameter pole marked into decimeters similar to that developed by Wiens (1974) but with a larger radius. The pole was held vertically against the ground and in each 10-cm section the number of herbaceous and woody vegetation contacts, or “hits,” was recorded. The mode herbaceous and woody heights within a 30-cm diameter of the pole were visually estimated to the nearest decimeter. For my analyses, I used only the density estimates from the first 10 cm of the pole because this range has the most influence on ground-dwelling birds. Within each 20-m transect section, at two of the four random points, I placed a 1-m² frame and estimated percent herbaceous and woody ground cover and plant species composition. Herbaceous and woody cover were treated as separate strata, thus they could total > 100%. In each 1-m² frame, I also estimated the number of woody stems at ground level using number classes (1–25, 26–50, 51–100, 101–150, 151–200, and > 200 stems).

I estimated the percent cover of each plant species with > 1% cover within the 1-m² frame. I attempted to identify all plants to species level. For abundant grass, sedge, and forb species, I collected voucher specimens; all vouchers were deposited at the Louisiana State University herbarium. As a means of variable reduction, I later grouped plant species into 15 guilds determined by the combination of their life form (graminoid, forb, or woody) and their

U.S. Fish & Wildlife Service Wetland Indicator Status (WIS). The WIS are: upland (UPL); facultative upland (FACU); facultative (FAC); facultative wetland (FACW); and obligate wetland (OBL; Appendix 1). Nomenclature follows that of Weakley (2008) and the USDA PLANTS Database (2009).

I measured tree basal area and canopy closure to determine tree densities. I used a 10-factor prism to measure tree basal area in each 20-m² vegetation plot (Avery 1967). Standing at the plot center, trees were viewed through the prism. For all trees that overlapped within the prism view, I measured diameter-at-breast height (DBH) using a Biltmore stick (Jackson 1911).

For RCW stands, I used circular vegetation-sampling plots with an 11.3-m radius, the same area as a 20-m² plot. In proportion to the size of the cluster, I established 5–10 random plots. These circular plots were divided into four wedges based on the cardinal directions and random points were located inside each wedge. The same vegetation structure and plant species composition data described above were collected at these random points.

Many of my study sites had patchy distributions of herbaceous cover and shrubs. To measure this patchiness, I used the coefficient of variation (CV) for the variables herbaceous cover, woody cover, and herbaceous density for each study site (entire transect; Wiens 1974, Rotenberry and Wiens 1980). The CV was calculated from each individual measurement within a site and thus represents heterogeneity, or patchiness, within a study site. Heterogeneity is important to measure because after averaging vegetation data over each site, this information could be lost.

Statistical Analyses

For all analyses, the individual study site, located in one of three stand types—upland, RCW, or bog—was the sample unit. Prior to analyses, bird densities were averaged over all sampling events each year for each site; vegetation measurements were averaged over each site

each year. For analyses comparing means among or between treatments, I omitted some sites and samples: site BAC, classified as stand type “other” because it was more of a degraded coastal savanna planted with slash pine, sites that had > 2 growing seasons since fire, and second-year surveys from the five bog sites sampled both years. Although this decreased my sample size, omitting these samples removed variation introduced by differences in time since fire among sites and removed the lack of independence between sites surveyed both years. Of the remaining 22 sites, 19 were one growing season since fire and three (one per stand type) were two growing seasons since fire. The total sample size was 22, consisting of six upland, six RCW, and 10 bog stands. All reported estimates of means and mean differences are least squared means unless stated otherwise. All *P*-values and confidence intervals reported for pairwise tests are Tukey–Kramer adjusted. For all tests, I used a significance level of 0.05.

Bird Analyses.—Grassland bird densities and richness were calculated according to the following rules. I included *Ammodramus* sp./Bachman’s Sparrow in grassland bird species richness estimates except for the sites where I also detected Bachman’s or Henslow’s Sparrows and risked double counting species. I also included *Ammodramus* sp. in grassland bird species richness estimates because I did not detect any other grassland birds at the site with these detections. Bird species and genera included in total grassland bird density estimates were Bachman’s and Henslow’s Sparrows, Sedge Wrens, and birds identified as *Ammodramus* sp. or *Ammodramus* sp./Bachman’s Sparrows.

The specific analysis performed on grassland bird species richness, total density, and individual species density among the three stand types and between salvaged and unsalvaged stands depended on whether the data met the assumptions for parametric analyses. All dependent variables were natural log-plus-one transformed and initially analyzed with a completely randomized, one-way ANOVA (PROC MIXED, SAS Institute Inc. 2006), and

residuals were tested for normality by Shapiro–Wilk tests (PROC UNIVARIATE). Species richness, Bachman’s Sparrow density, and Sedge Wren density data did not meet criteria for normality (Shapiro–Wilk $P > 0.05$). Therefore, non-parametric Kruskal–Wallis and Wilcoxon tests (PROC NPAR1WAY) and generalized linear model (Poisson and negative binomial; PROC GLIMMIX) alternatives were explored. Because the generalized linear models exhibited severe problems with over- and underdispersion, Kruskal–Wallis and Wilcoxon tests were performed on species richness data and Bachman’s Sparrow and Sedge Wren density data to compare stand types. I specified the EXACT statement for both the Kruskal–Wallis and Wilcoxon tests to calculate exact P -values (vs. approximate P -values based on Z or t distributions). All other analyses comparing stand types were performed with ANOVA. If an ANOVA was significant, I tested for differences between stand types using Tukey–Kramer pairwise tests. It is important to remember when testing for differences between salvaged and unsalvaged sites that only one bog stand was salvaged, all RCW stands were salvaged, and only one site was not salvaged in upland stands. Thus, any difference between treatments is likely driven more by vegetation characteristics pertaining to stand types and individual sites than effects of salvage logging *per se*. What effects salvage logging had on the vegetation cannot be determined without pre-salvage data or adequate control sites, both of which are lacking.

I used a paired t -test to compare Henslow’s Sparrow density estimates between the Chandler–Woodrey (1995) and battue methods. I tested for a difference between mid-winter Henslow’s Sparrow densities estimated using both methods at the same sites in southeastern Louisiana in winter of 2008–2009. Because the sampling dates varied between the methods, and because Henslow’s Sparrow numbers tend to decline steadily over the winter (Johnson 2006), I compared bird density estimates from the Chandler–Woodrey-method surveys conducted mid–

late January with averaged density estimates from the battue-method surveys conducted in early January and mid-February.

Because of the relationships between area and species richness and area and abundance (Preston 1948, Rosenzweig 1995), I used Spearman's correlation coefficients and bivariate plots to inspect the relationship between area sampled and grassland bird species richness, total grassland bird density, and density of Henslow's Sparrows and Sedge Wrens, the two most abundant wintering grassland birds in DSNF. The area sampled at each site roughly corresponds to the grassland patch size for that site: small patches contained short transects ranging 100–220 m, and larger patches contained 300–400 m of transect. Bachman's Sparrow was the third most abundant grassland bird in DSNF, but there were not enough detections to analyze relationships. For this analysis, I used the same reduced data set ($n = 22$ plots) that I used for comparing treatment means.

I noticed a decline in Henslow's Sparrow and Sedge Wren abundances over the first winter of sampling in DSNF and Louisiana, a trend that was not readily apparent the second year. Henslow's Sparrow numbers are known to decline gradually over winter, and this decline does not seem to vary with time since fire (Johnson 2006). I wanted to address the following questions: (1) whether the decline of Henslow's Sparrow and Sedge Wren densities over the course of winter was statistically significant; (2) whether the decline of Henslow's Sparrows varied between sampling years or between states; and (3) whether Sedge Wren decline varied between sampling years (I did not sample Sedge Wren densities in Louisiana, so I could not compare between the two states). I used separate generalized linear mixed models with a covariate (which are non-parametric analogs of conventional analysis of covariance; PROC GLIMMIX) for Henslow's Sparrow and Sedge Wren densities with study site modeled as a random variable to account for the covariance among bird densities from the same sites sampled

multiple times within a year (Paterson and Lello 2003). For all models, I used type I tests of fixed effects. During model fitting, I noticed and removed an extreme outlier (> 2.5 standard deviations; McGarigal et al. 2000) measured mid-winter the second year in Mississippi. Different potential general and generalized linear models (normal untransformed data, normal log-plus-one transformed data, Poisson, and negative binomial distributions) were fit for each species, and the appropriateness of the model was determined by examining \hat{c} (Pearson χ^2/df). Henslow's Sparrow data were best fit by a natural log transformation and normal distribution ($\hat{c} = 1.24$). Sedge Wren data were best fit by a Poisson distribution ($\hat{c} = 1.45$). For this analysis, I only included sites where Henslow's Sparrows or Sedge Wrens were detected at least once. Henslow's Sparrows were not sampled in December the first year in Louisiana. The 2009 second-year surveys from Louisiana include combined density estimates from surveys conducted with the battue method and from surveys using the Chandler–Woodrey (1995) sampling methods.

I also ran linear regressions of natural log-plus-one transformed bird densities on winter days for both species by state and year separately, designating study site as a fixed block because reduced observations precluded designating study site as a random effect. For each species–year–date combination, I fit linear, cubic, quadratic, exponential, and power models to determine if the relationships between bird densities and winter days were nonlinear. I selected the model with the best fit using AICc (Burnham and Anderson 2002) and used type I and III tests of fixed effects (type III for linear, exponential, and power models; type I for polynomials) to determine significance. The sample sizes (*i.e.*, number of sites) for the analyses of Henslow's Sparrow were $n = 9$ for 2007–2008 and $n = 10$ in 2008–2009 in Louisiana, and $n = 4$ for 2007–2008 and $n = 11$ for 2008–2009 in Mississippi. For Sedge Wren, $n = 5$ for 2007–2008 and $n = 6$ for winter 2008–2009 in Mississippi.

Vegetation Analyses.—I conducted two principal components analyses (PCA) using PROC FACTOR to reduce the number of correlated vegetation structure and plant species composition guild variables to fewer, uncorrelated principal components (PCs). I performed a PCA on the 12 structure variables and 15 plant guilds separately because of limited degrees of freedom ($n = 22$) and used a Varimax rotation to aid in the interpretation of the PCs. I retained all PCs with Eigenvalues > 1 (Guttman 1954).

To further explore plant species composition among study sites and stand types I used nonmetric multidimensional scaling (NMDS) on a Bray–Curtis dissimilarity matrix of plant species composition by percent cover. Nonmetric multidimensional scaling scores along a scale in ordination space can elucidate underlying ecological gradients. I used PROC NMS specifying the ordinal data option for nonmetric analysis (SAS Institute Inc. 2006). Nonmetric multidimensional scaling ranks the values in the dissimilarity matrix and is thus appropriate for non-normal data: it does not assume linearity nor is it influenced by zero-rich data (McCune et al. 2002). Bray–Curtis, or Sørensen, distance matrices are recommended and commonly used for community count data (Bray and Curtis 1957, Drewa et al. 2002a, McCune et al. 2002). Stress, the badness-of-fit statistic produced by NMDS, is a measure of monotonicity between the original data dimensionality and the reduced ordination dimensionality; the lower the stress value, the better the ordination dimensionality represents the dimensionality of the original data (Kruskal and Wish 1978, McCune et al. 2002). I specified two NMDS dimensions because the stress value for two dimensions was near 0.1 and because two dimensions are easier to visualize and interpret ecologically than more than two dimensions. In NMDS, the more dimensions specified, the lower the stress value, so a realistic medium must be met based on the interpretability of the dimensions and knowledge of the study system (Kruskal and Wish 1978). I tried specifying three dimensions, but additional dimensions became increasingly difficult to interpret; hence, I

decided that two dimensions were best for providing an adequate stress level and interpretability. To assist in the ecological interpretation of the dimensions, I used Spearman's correlation coefficients to examine the relationships between plant species composition values and NDMS dimension scores.

I used three MANOVAs to test for differences in vegetation structure PC scores, plant species guild PC scores, and plant species NMDS dimension scores among stand types. I used MANOVA to compare multiple means among stands while maintaining an experiment-wise error rate of 0.05. All tests were performed using the MANOVA option in PROC GLM. Pairwise differences between means were tested using Tukey–Kramer tests. Residuals were tested for normality using Shapiro–Wilk tests. All reported estimates of means and mean differences are least squared means unless stated otherwise. All *P*-values and confidence intervals reported for pairwise tests are Tukey–Kramer adjusted.

RESULTS

Grassland Bird Sampling

I detected 22 species of birds on transect over two winters, three of which were grassland species (excluding four categories of unidentified birds). An additional 26 species were detected within 50 m of the transects (Appendix 2). The three most abundant wintering grassland bird species on the transects, in order of abundance, were Henslow's Sparrow, Sedge Wren, and Bachman's Sparrow; percent site occupancy for each species was 22%, 44%, and 15%, respectively (Table 1.2). Sedge Wren densities were higher than Bachman's Sparrow densities, but only when considering all sites surveyed over both years ($n = 32$). Considering only the 22 sites used to compare mean densities, Bachman's Sparrow densities were higher than Sedge Wren densities, although Sedge Wrens occupied more sites. Out of the 27 study sites, Bachman's Sparrows occurred at five, Henslow's Sparrows occurred in eleven, and Sedge Wrens

occurred at nine. At one site (KES), I detected just two grassland birds, and each were only identified to the genus *Ammodramus*. At three RCW stands, I identified one or more birds as *Ammodramus* sp./Bachman's Sparrows. At one of these three sites, I detected both a Bachman's and Henslow's Sparrow, as well as a bird classified as *Ammodramus* sp./Bachman's Sparrow. At another of these three sites, I detected several Henslow's Sparrows along with several birds

Table 1.2. All bird species, in order of mean density, detected on transect over two winters of grassland bird surveys in DSNF, MS. Proportion of occurrence is out of 27 study sites. Grassland species are in bold.

Species	Mean density	SE	% occurrence
<i>Spizella passerina</i> (Chipping Sparrow)	3.16	1.72	0.30
<i>Turdus migratorious</i> (American Robin)	1.95	1.95	0.04
<i>Ammodramus henslowii</i> (Henslow's Sparrow)	1.76	0.55	0.44
<i>Melospiza georgiana</i> (Swamp Sparrow)	0.65	0.21	0.33
<i>Dendroica pinus</i> (Pine Warbler)	0.37	0.13	0.30
<i>Picoides borealis</i> (Red-cockaded Woodpecker)	0.37	0.37	0.04
<i>Cistothorus platensis</i> (Sedge Wren)	0.30	0.11	0.33
<i>Aimophila aestivalis</i> (Bachman's Sparrow)	0.27	0.17	0.19
<i>Sitta pusilla</i> (Brown-headed Nuthatch)	0.24	0.19	0.11
<i>Troglodytes aedon</i> (House Wren)	0.16	0.08	0.22
Unknown bird	0.16	0.05	0.30
<i>Junco hyemalis</i> (Dark-eyed Junco)	0.14	0.14	0.04
<i>Melospiza melodia</i> (Song Sparrow)	0.12	0.10	0.07
<i>Ammodramus</i> sp./<i>A. aestivalis</i>	0.08	0.05	0.11
<i>Mimus polyglottos</i> (Northern Mockingbird)	0.08	0.08	0.04
<i>Sialia sialis</i> (Eastern Bluebird)	0.06	0.04	0.07
Sparrow sp.	0.06	0.03	0.15
<i>Ammodramus</i> sp.	0.05	0.04	0.07
<i>Geothlypis trichas</i> (Common Yellowthroat)	0.05	0.04	0.07
<i>Zonotrichia albicollis</i> (White-throated Sparrow)	0.05	0.05	0.04
<i>Picoides pubescens</i> (Downy Woodpecker)	0.03	0.03	0.04
<i>Zenaida macroura</i> (Mourning Dove)	0.03	0.03	0.04
<i>Pooecetes gramineus</i> (Vesper Sparrow)	0.02	0.02	0.04
<i>Scolopax minor</i> (American Woodcock)	0.02	0.02	0.04
<i>Dendroica coronata</i> (Yellow-rumped Warbler)	0.01	0.01	0.04
<i>Sphyrapicus varius</i> (Yellow-bellied Sapsucker)	0.01	0.01	0.04

classified as *Ammodramus* sp./Bachman's Sparrows. I did not detect any other grassland birds besides *Ammodramus* sp./Bachman's Sparrow at the third of these three sites. Considering only the sites at which they occurred, density ranges of each species were: Bachman's Sparrows 0.42–2.22; Henslow's Sparrows 0.52–13.33; and Sedge Wrens 0.42–3.03. The highest densities of Henslow's Sparrows occurred in a 0.2-ha transect located in a bog < 1 ha (ZAK). I captured 25 Henslow's Sparrows and five Sedge Wrens over both years of sampling (Appendix 3).

Differences among Stand Types.—Grassland bird densities in upland, RCW, and bog stands varied among individual species. Bachman's Sparrow densities did not statistically differ among stand types ($\chi^2 = 3.84$, $df = 2$, $P = 0.1511$); however, no Bachman's Sparrows ever occurred in bog stands (Table 1.3; Fig. 1.2). Henslow's Sparrow densities were significantly different among stand types ($F = 3.94$, $df = 19$, $P = 0.0369$), with higher densities in bogs than upland stands ($t = 2.80$, $df = 19$, $P = 0.0296$); no Henslow's Sparrows occurred in upland stands (Table 1.4, Fig. 1.2). Sedge Wren densities did not statistically differ among stand types ($\chi^2 = 0.67$, $df = 2$, $P = 0.7057$). There were no statistically significant differences in grassland bird species richness ($\chi^2 = 0.50$, $df = 2$, $P = 0.7634$) or total grassland bird density ($F = 2.48$, $df = 19$, $P = 0.1106$) among stand types (Figs. 1.3–1.4).

Differences between Salvage and Unsalvaged Stands.—There were no significant differences in grassland bird species richness ($S = 88.5$, $P = 0.0641$), total grassland bird densities ($S = 115.0$, $P = 1.0$), Bachman's Sparrow densities ($S = 95.0$, $P = 0.0964$), Henslow's Sparrow densities ($S = 131.0$, $P = 0.2796$), or Sedge Wren densities ($S = 110.0$, $P = 0.6970$) between salvaged and unsalvaged stands (Table 1.5).

Method Comparisons

I found no statistically significant difference in mid-winter Henslow's Sparrow densities estimated by the Chandler–Woodrey (1995) and battue methods ($t = -0.35$, $df = 7$, $P = 0.7347$). The mean (\pm SE) difference in bird density estimates from the Chandler–Woodrey (1995) and battue methods was -0.20 ± 0.56 . Similar results from these two methods suggest that results

Table 1.3. Wilcoxon rank score sums and means for species richness, Bachman's Sparrow (BACS) densities, and Sedge Wren (SEWR) densities among stand types in DSNF, MS calculated from Kruskal–Wallis tests. There were no significant differences among stands for any of the variables.

Stand	<i>n</i>	Sums of scores	Expected under null	SD under null	Mean
<u>Species Richness</u>					
Upland	6	72.5	69.0	11.63	12.08
RCW	6	74.5	69.0	11.63	12.42
Bog	10	106.0	115.0	13.00	10.60
<u>BACS density</u>					
Upland	6	79.0	69.0	9.13	13.17
RCW	6	79.0	69.0	9.13	13.17
Bog	10	95.0	115.0	10.21	9.50
<u>SEWR density</u>					
Upland	6	74.0	69.0	10.65	12.33
RCW	6	60.5	69.0	10.65	10.08
Bog	10	118.5	115.0	11.90	11.85

from previous studies and this study are robust with respect to the methods used—important for metareplication of wildlife research (Johnson 2002).

Species–Area Relationships

There was no significant correlation between grassland bird richness and area sampled (Spearman's $R = 0.30$, $P = 0.1711$; Fig. 1.5). Grassland bird density was negatively correlated with area sampled (Spearman's $R = -0.55$, $P = 0.0074$; Fig. 1.5). Of the two most numerous

wintering grassland bird species in DSNF, Henslow's Sparrow density was negatively associated with area sampled (Spearman's $R = -0.43$, $P = 0.0465$; Fig. 1.6), but there was no significant relationship between Sedge Wren density and area sampled (Spearman's $R = 0.38$, $P = 0.0836$; Fig. 1.6).

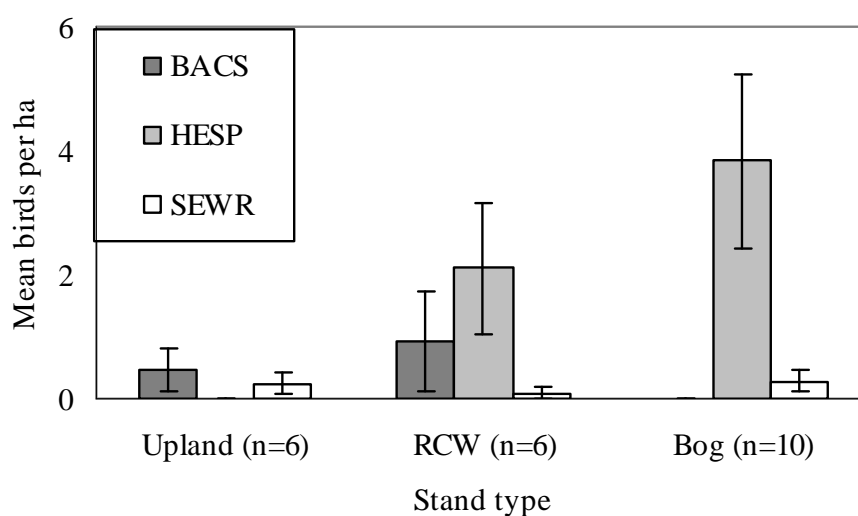


Figure 1.2. Mean (\pm SE) densities (untransformed data) for the three common species of wintering grassland birds in DSNF, MS. Henslow's Sparrows did not occur in upland stands; no Bachman's Sparrows occurred in bogs.

Bird Decline over Winter

Henslow's Sparrow and Sedge Wren densities decreased over the first winter of sampling but not over the second winter. Generalized linear modeling results showed the type I test of fixed effects for Henslow's Sparrow overall decline in density over winter was not quite statistically significant ($F = 3.73$, $df = 49$, $P = 0.0593$). General linear regression results of log-plus-one transformed Henslow's Sparrow densities on winter days by year and state varied. Bird decline in Mississippi over the first year of sampling was significant and best fit by the power

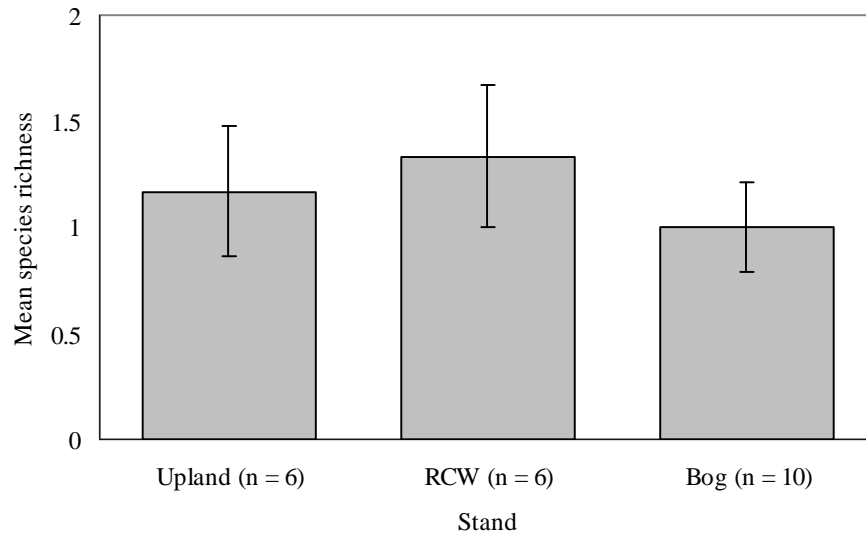


Figure 1.3. Mean (\pm SE) grassland bird species richness (untransformed data) among three stand types in DSNF, MS. There was no significant difference among stands.

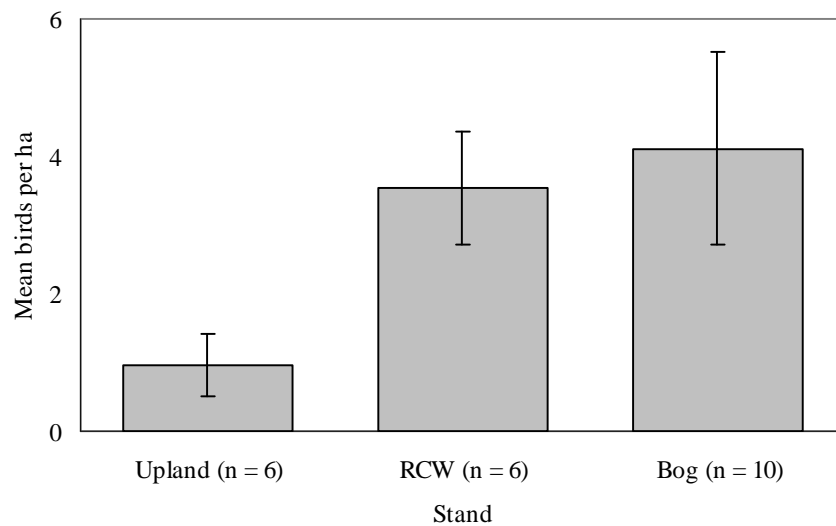


Figure 1.4. Mean (\pm SE) grassland bird densities (untransformed data) among three stand types in DSNF, MS. There was no significant difference among stands.

Table 1.4. Differences in means (log-plus-one transformed) of total grassland bird densities and Henslow's Sparrow (HESP) densities between stand types in DSNF, MS estimated from Tukey–Kramer pairwise tests. Significant differences are in bold.

Stand Comparison	Difference in means	SE	95% CI
Total density			
Bog/RCW	-0.13	0.38	-1.10–0.84
Bog/Upland	0.73	0.38	-0.24–1.70
RCW/Upland	0.86	0.43	-0.22–1.94
HESP density			
Bog/RCW	0.34	0.42	-0.72–1.39
Bog/Upland	1.16	0.42	0.11–2.22
RCW/Upland	0.83	0.47	-0.35–2.01

Table 1.5. Wilcoxon rank score sums and means from Wilcoxon two-sample tests for bird richness, total grassland bird densities, Bachman's Sparrow (BACS) densities, Henslow's Sparrow (HESP) densities, and Sedge Wren (SEWR) densities between salvaged and unsalvaged stands in DSNF, MS. There were no significant differences.

Stand	<i>n</i>	Sums of scores	Expected under null	SD under null	Mean
Species richness					
Salvaged	12	164.5	138.0	13.00	13.71
Unsalvaged	10	88.5	115.0	13.00	8.85
Total density					
Salvaged	12	138.0	138.0	15.11	11.50
Unsalvaged	10	115.0	115.0	15.11	11.50
BACS density					
Salvaged	12	158.0	138.0	10.21	13.17
Unsalvaged	10	95.0	115.0	10.21	9.50
HESP density					
Salvaged	12	122.0	138.0	14.19	10.17
Unsalvaged	10	131.0	115.0	14.19	13.10
SEWR density					
Salvaged	12	143.0	138.0	11.90	11.92
Unsalvaged	10	110.0	115.0	11.90	11.00

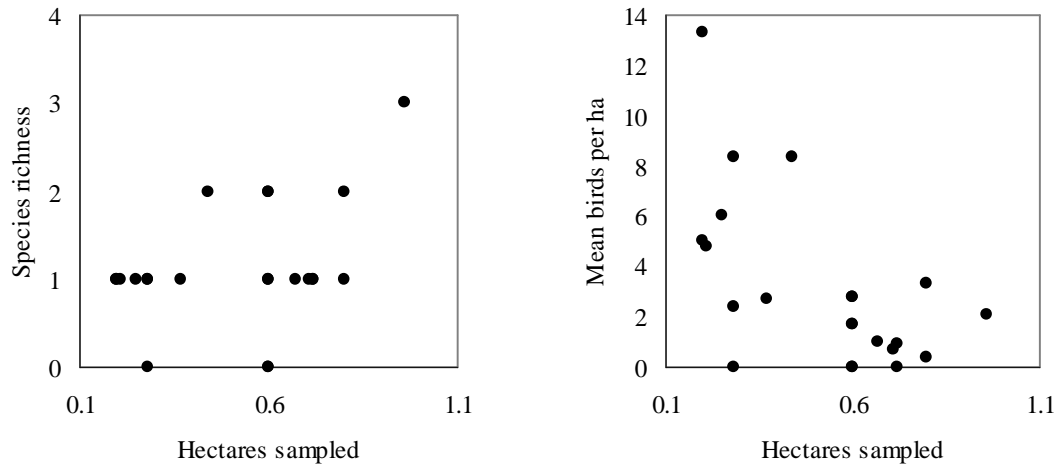


Figure 1.5. Grassland bird richness and mean total bird density plotted against area sampled in DSNF, MS. Total bird density was negatively correlated with area sampled ($R = -0.55$).

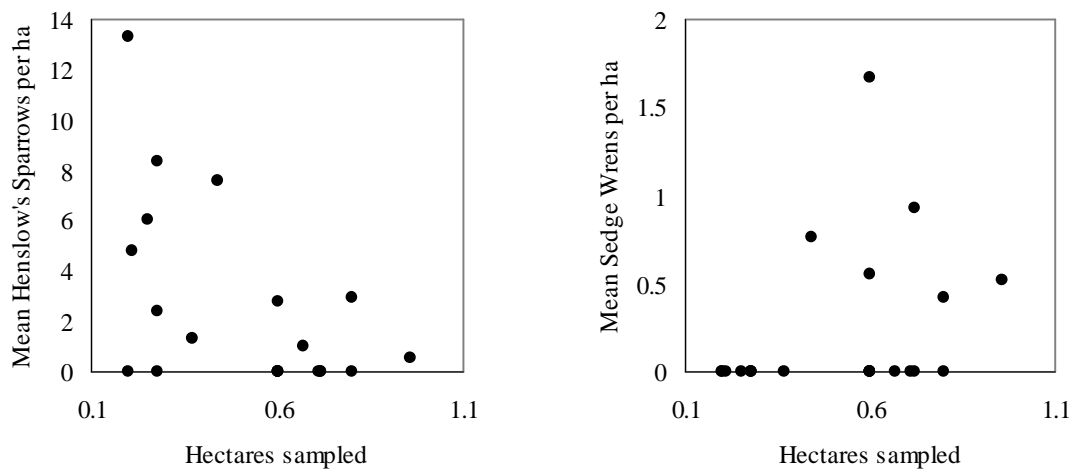


Figure 1.6. Mean Henslow's Sparrow and Sedge Wren densities plotted against area sampled in DSNF, MS. Henslow's Sparrow density was negatively correlated with area sampled ($R = -0.43$).

model ($y = \beta_0 + x^{\beta_1}$; $F = 62.94$, $df = 4$, $P = 0.0014$) showing a non-linear decrease in bird densities over winter (Fig. 1.7). The slope estimate (\pm SE) was $\beta_0 = -0.3021 \pm 0.3166$ ($t = -0.95$, $df = 4$, $P = 0.3941$, 95% CI: -1.1812–0.5771). The slope estimate was not statistically significant, which means that although the variable winter day had an effect on sparrow density, the estimate of that

effect is uncertain (MacKenzie et al. 2006). The transformed Henslow's Sparrow densities for the first sampling year in Louisiana and the second year of sampling in Mississippi were not normal, therefore making inference from these results unreliable (Figs. 1.8–1.9). None of the models of Henslow's Sparrow densities on winter days in Louisiana the second sampling year was significant (Fig. 1.9). Generalized linear modeling of Sedge Wren decline over winter days had issues with the estimated G matrix making inference from the results unreliable. None of the general linear regression models of Sedge Wren densities on winter days from the first sampling year was significant, but inspection of the bivariate plot shows a noticeable decline over the first winter of sampling in 2007–2008 (Fig. 1.8).

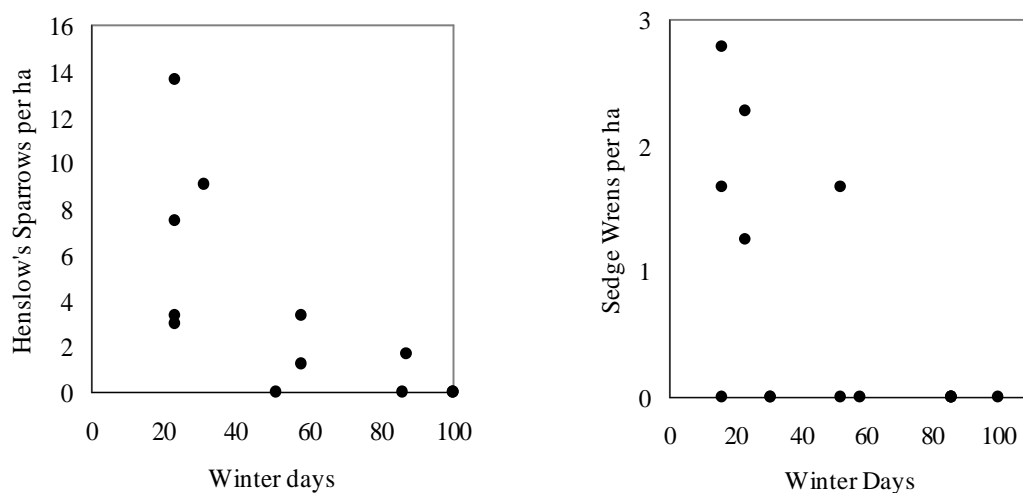


Figure 1.7. Henslow's Sparrow and Sedge Wren densities plotted on time of winter (day 1 = 16 November) during the first winter (2007–2008) in DSNF, MS.

Vegetation Structure and Plant Species Composition

Vegetation structure and plant species composition varied widely among study sites (Table 1.6). I identified 100 plant species not including canopy trees. The number of plants identified is lower than the actual number of species encountered because I grouped some

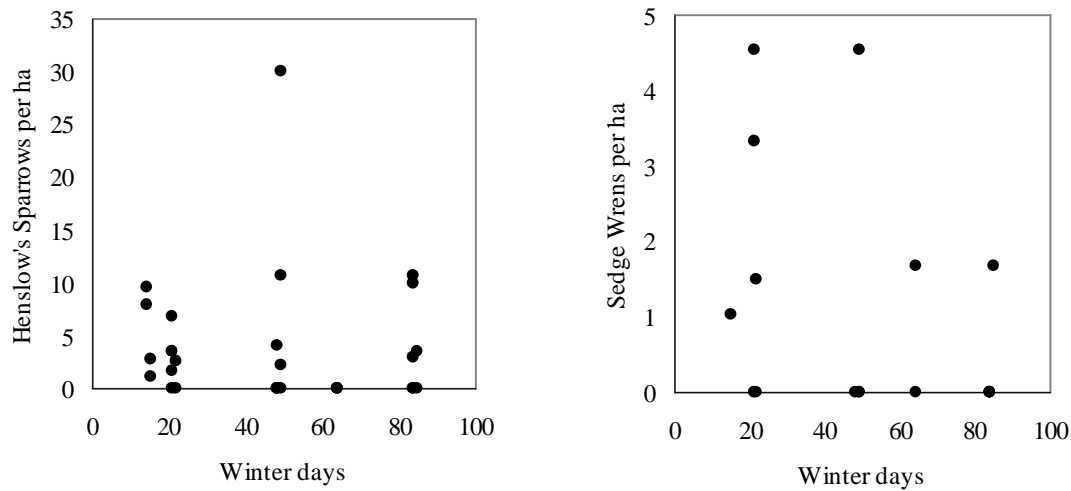


Figure 1.8. Henslow's Sparrow and Sedge Wren densities plotted on time of winter (day 1 = 16 November) during the second winter (2008–2009) in DSNF, MS.

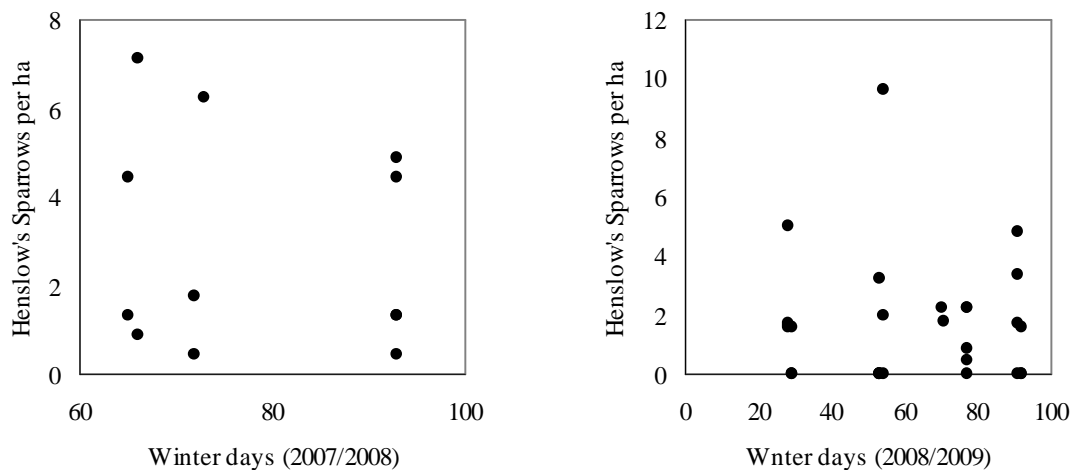


Figure 1.9. Henslow's Sparrow densities plotted on time of winter (day 1 = 16 November) during winters 2007–2008 and 2008–2009 in southeastern, LA.

species by genera or morphospecies if I could not identify them to species level (*e.g.*, *Rhynchospora* or *Dichantheium* spp.). I grouped the grasses *Aristida longespica* (slimspike threeawn) and *A. oligantha* (prairie threeawn) because I could not distinguish between these two species in their vegetative winter states. A number of *Rhynchospora* species in DSNF have

prostrate, filamentous stems and are thus structurally different from other species in the same genus. I could not identify these filamentous plants—most of which occurred in bogs—to species because fruits were rare, so I grouped them into a category called *Rhynchospora* spp. “filamentous.” Of the 100 plant species, 41 were graminoids, 26 were forbs, and 33 were woody plants. Sites contained 19–48 plant species; the mean (\pm SE) number of species for all 27 study sites was 31.0 ± 1.3 . The number of plant species detected was positively correlated with area sampled ($R = 0.73$, $P < 0.0001$). Only three plant species guilds occurred in all sites: FAC graminoids, FACW graminoids, and FACW woody plants.

Principal Components Analysis on Vegetation Structure.—Principal components analysis of the vegetation structure variables resulted in three principal components with Eigenvalues > 1 , representing 84% of the variance. The first PC represented woody understorey structure and was mostly correlated with all woody understorey structure variables, not including trees, and herbaceous height (Table 1.7). The second PC represented herbaceous structure and was mostly correlated with the remaining herbaceous structure variables and the woody cover CV. The third PC represented tree density and was mainly correlated with canopy closure and tree basal area.

Principal Components Analysis on Plant Species Composition Guilds.—Principal components analysis of plant species composition guilds resulted in five principal components with Eigenvalues > 1 , representing 72% of the variance. The first PC was mostly correlated with UPL, FACU, FAC, and FACW graminoids, OBL forbs, and moderately correlated with FACU woody plants (Table 1.8). The second PC was mostly correlated with FAC and FACW woody plants, the third PC with FAC forbs and OBL woody plants, and the forth PC with FAC forbs, OBL graminoids, and somewhat with FACU woody plants. The fifth PC was mostly correlated with UPL forbs and UPL woody plants, and had the highest, albeit low, correlation with FACU

forbs. FACU woody plants were not highly correlated with any PC and loaded almost equally on PC 1 and PC 4.

Nonmetric Multidimensional Scaling of Plant Species Composition.—Nonmetric multidimensional scaling of plant species composition resulted in the arrangement of sites along what can be interpreted as a moisture gradient for the first dimension. The final stress level was 0.1021, indicating a meaningful relationship between the ordination and the dimensionality of the data (McCune et al. 2002). Spearman's rank correlations of plant species and Dimension 1 scores showed that wetland plants were mostly correlated with the positive end of the first dimension axis and upland plants were correlated with the negative end (Table 1.9). The second dimension was harder to interpret ecologically; each axis end consisted of a mixture of grasses, forbs, and woody plants covering a range of moisture preferences. A biplot of study sites plotted on Dimensions 1 and 2 showed that sites in bog stands were distinct from sites in RCW and upland stands (Fig. 1.10). The majority of upland and RCW sites overlapped in their plant species composition in ordination space.

Differences among Stand Types.—Results from MANOVA showed that vegetation structure PC scores were statistically different among stand types (Wilk's $\lambda = 0.23$, $F_{6, 34} = 6.18$, $P = 0.0002$). The mean herbaceous structure PC scores were statistically higher in bog stands than upland ($P < 0.0001$) and RCW stands ($P = 0.0035$; Fig. 1.11), meaning that bog stands had more spatially continuous herbaceous cover and more patchy woody cover. Although there was no statistically significant differences in means for the woody structure PC, RCW stands had lower mean woody vegetation scores (Fig. 1.11). The tree density PC did not significantly differ among stand types (Table 1.10).

Table 1.6. Mean, minimum, and maximum estimates ($n = 22$) for vegetation structure measurements and plant species composition guilds in DSNF, MS.

Variable	Mean	SE	Minimum	Maximum
Tree basal area (m ² /20 m ²)	0.39	0.04	0.00	0.79
Canopy closure (%)	36.54	4.26	0.69	90.08
Herb cover (%)	58.61	3.55	20.40	94.77
Herb cover CV	45.89	4.23	5.49	93.70
Herb height (cm)	16.99	0.90	11.43	35.93
Herb density (# hits < 10 cm)	5.72	0.29	2.90	8.10
Herb density CV	57.56	3.53	30.66	103.17
Woody cover (%)	15.91	1.69	0.70	36.03
Woody cover CV	114.56	9.56	52.64	238.62
Number stems (rank median)	20.85	2.00	2.88	51.25
Woody height (cm)	26.92	2.91	1.58	52.76
Woody density (# hits < 10 cm)	0.26	0.04	0.00	0.63
Graminoid UPL	0.02	0.01	0.00	0.29
Graminoid FACU	10.57	1.83	0.00	30.33
Graminoid FAC	16.79	2.47	0.60	51.11
Graminoid FACW	18.74	3.41	0.92	74.50
Graminoid OBL	10.66	2.84	0.00	66.41
Forb UPL	0.14	0.08	0.00	2.67
Forb FACU	0.13	0.07	0.00	1.81
Forb FAC	0.85	0.27	0.00	6.50
Forb FACW	0.40	0.18	0.00	5.29
Forb OBL	5.43	1.89	0.00	42.70
Woody UPL	0.03	0.02	0.00	0.50
Woody FACU	0.41	0.12	0.00	3.13
Woody FAC	2.43	0.43	0.00	8.67
Woody FACW	13.23	1.55	0.70	33.73
Woody OBL	0.57	0.26	0.00	6.07

Results from MANOVA showed that mean plant composition PC scores were statistically different among stand types (Wilk's $\lambda = 0.09$, $F_{10, 30} = 7.01$, $P < 0.0001$). The first PC mean score was significantly lower in bog stands than upland ($P = 0.0133$) and RCW stands ($P < 0.0001$; Table 1.11). Principal component 1 was positively correlated with FACU and FAC graminoids and negatively correlated with OBL forbs, and UPL and FACW graminoids; thus,

Table 1.7. Rotated principal components pattern from a PCA on 12 vegetation structure variables measured in DSNF, MS. Values are the correlations of the raw variables with each PC. The three PCs explain a cumulative 84% of the total variance. Highest correlations are in bold.

Variable	Woody	Herbaceous	Trees
Woody cover	0.896	-0.268	0.270
Woody density	0.885	-0.279	-0.154
Number stems	0.857	-0.305	0.024
Woody height	0.738	-0.570	0.157
Herb height	0.651	-0.222	0.115
Herb density	-0.191	0.935	-0.125
Herb cover	-0.386	0.835	-0.035
Woody cover CV	-0.571	0.632	-0.068
Herb cover CV	0.613	-0.693	0.276
Herb density CV	0.550	-0.737	0.072
Canopy closure	0.136	0.091	0.948
Tree basal area	0.009	-0.378	0.869
Proportion s ² explained	61%	14%	09%

Table 1.8. Rotated principal components pattern from a PCA on 15 plant species composition guilds measured in DSNF, MS. Values are the correlations of the guilds with each PC. The five PCs explain a cumulative 72% of the total variance. Highest correlations are in bold. See Methods for guild names.

Variable	PC 1	PC 2	PC 3	PC 4	PC 5
Graminoid FACU	0.829	-0.002	-0.296	-0.179	-0.135
Graminoid FAC	0.825	0.000	-0.268	0.002	-0.074
Forb OBL	-0.616	-0.597	-0.013	0.153	-0.049
Graminoid UPL	-0.700	0.266	-0.277	-0.147	-0.356
Graminoid FACW	-0.739	-0.236	0.091	0.107	-0.305
Woody FACW	0.032	0.906	-0.111	0.160	0.035
Woody FAC	-0.106	0.705	0.230	-0.479	0.283
Woody OBL	-0.144	0.099	0.939	-0.036	-0.077
Forb FACW	-0.137	-0.128	0.861	0.114	-0.042
Forb FAC	0.129	0.364	-0.139	0.756	-0.099
Gram OBL	-0.231	-0.217	0.271	0.687	0.025
Woody FACU	0.439	0.178	-0.048	-0.472	0.201
Forb UPL	-0.064	0.071	0.029	-0.039	0.822
Woody UPL	0.179	-0.009	-0.104	-0.379	0.609
Forb FACU	0.235	0.262	-0.174	0.120	0.376
Proportion s ² explained	27%	14%	12%	11%	08%

Table 1.9. Spearman's rank correlation coefficients for correlations between plant species composition and nonmetric multidimensional scaling scores for two dimensions.

Species	Dimension 1	Species	Dimension 2
<i>Sarracenia alata</i>	0.80	<i>Panicum verrucosum</i>	0.74
<i>Eriocaulon</i> spp.	0.77	<i>Rubus</i> sp.	0.65
<i>Sarracenia psittacina</i>	0.64	<i>Ilex coriacea</i>	0.60
<i>Scleria muhlenbergii</i>	0.61	<i>Eupatorium</i> spp.	0.55
<i>Xyris</i> spp.	0.61	<i>Morella cerifera</i>	0.53
<i>Rhynchospora</i> spp. "filamentous"	0.61	<i>Carex</i> spp.	0.47
<i>Aristida palustris</i>	0.44	<i>Dichanthelium scabriusculum</i>	0.45
<i>Ctenium aromaticum</i>	0.44	<i>Iris</i> sp.	0.43
<i>Nyssa biflora</i>	0.40	<i>Andropogon glomeratus</i>	0.43
<i>Magnolia virginiana</i>	0.39	<i>Anthraenantia villosa</i>	0.38
<i>Chaptalia tomentosa</i>	0.37	<i>Smilax glauca</i>	0.38
<i>Smilax laurifolia</i>	0.36	<i>Viburnum nudum</i>	0.37
<i>Hypericum</i> spp.	0.34	<i>Osmunda</i> sp.	0.36
<i>Andropogon mohrii</i>	0.33	<i>Andropogon glaucopsis</i>	0.36
<i>Myrica heterophylla</i>	0.33	<i>Arundinaria gigantea</i>	0.36
<i>Cliftonia monophylla</i>	0.32	<i>Bidens</i> sp.	0.36
<i>Rhynchospora</i> spp.	0.31	<i>Woodwardia areolata</i>	0.36
<i>Pinus elliottii</i>	0.30	<i>Symplocos tinctoria</i>	0.34
<i>Helianthus angustifolius</i>	0.27	<i>Eleocharis tuberculosa</i>	0.34
<i>Muhlenbergia expansa</i>	0.27	<i>Erianthus giganteus</i>	0.34
<i>Dichromena latifolia</i>	0.23	<i>Aronia arbutifolia</i>	0.34
<i>Fuirena</i> sp.	0.23	<i>Cyrilla racemiflora</i>	0.33
<i>Scleria</i> spp.	0.21	<i>Carex glaucescens</i>	0.32
<i>Andropogon gyrans</i> var. <i>stenophyllus</i>	0.21	<i>Persea palustris</i>	0.31
<i>Juncus</i> sp.	0.19	<i>Lycopodiella alopecuroides</i>	0.27
<i>Dichanthelium scabriusculum</i>	0.19	<i>Eryngium integrifolium</i>	0.27
<i>Gymnopogon brevifolius</i>	0.18	<i>Zigadenus densus</i>	0.23
<i>Lycopodiella alopecuroides</i>	0.18	<i>Lyonia lucida</i>	0.23
<i>Eleocharis tuberculosa</i>	0.16	<i>Ilex vomitoria</i>	0.22
<i>Acer rubra</i> var. <i>drummondii</i>	0.16	<i>Ilex glabra</i>	0.19
<i>Anthraenantia villosa</i>	0.15	<i>Cliftonia monophylla</i>	0.19
<i>Panicum anceps</i>	0.15	<i>Gaylussacia mosieri</i>	0.19
<i>Morella cerifera</i>	0.15	<i>Cirsium</i> sp.	0.15
<i>Eryngium integrifolium</i>	0.15	<i>Diodia teres</i>	0.15
<i>Carex glaucescens</i>	0.15	<i>Mitchella repens</i>	0.15
<i>Arundinaria gigantea</i>	0.12	<i>Viola primulifolia</i>	0.13
<i>Eragrostis refracta</i>	0.11	<i>Euthamia</i> spp.	0.13
<i>Iris</i> sp.	0.10	<i>Solidago</i> spp.	0.11

Table 1.9 continued.

<i>Carex</i> spp.	0.07	<i>Panicum anceps</i>	0.11
<i>Zigadenus densus</i>	0.04	<i>Panicum virgatum</i>	0.10
<i>Lyonia lucida</i>	0.04	<i>Vaccinium stamineum</i>	0.09
<i>Aster</i> spp.	0.04	<i>Licania michauxii</i>	0.08
<i>Erianthus giganteus</i>	0.03	<i>Prunus serotina</i>	0.07
<i>Andropogon glomeratus</i>	0.01	<i>Gelsemium sempervirens</i>	0.07
<i>Viburnum nudum</i>	-0.01	<i>Pinus elliotii</i>	0.06
<i>Bigelowia nudata</i>	-0.01	<i>Magnolia virginiana</i>	0.06
<i>Bidens</i> sp.	-0.03	<i>Chasmanthium laxum</i>	0.06
<i>Woodwardia areolata</i>	-0.03	<i>Andropogon gerardii</i>	0.06
<i>Nyssa sylvatica</i>	-0.05	<i>Gymnopogon brevifolius</i>	0.05
<i>Panicum virgatum</i>	-0.05	<i>Paspalum</i> sp.	0.04
<i>Rhynchospora chapmanii</i>	-0.06	<i>Muhlenbergia expansa</i>	0.03
<i>Pityopsis graminifolia</i>	-0.07	<i>Sarracenia psittacina</i>	0.02
<i>Chasmanthium sesseliflorum</i>	-0.08	<i>Pityopsis graminifolia</i>	0.02
<i>Rubus</i> sp.	-0.08	<i>Elephantopus</i> sp.	0.01
<i>Tridens ambiguus</i>	-0.08	<i>Bigelowia nudata</i>	0.00
<i>Osmunda</i> sp.	-0.11	<i>Nyssa sylvatica</i>	-0.02
<i>Andropogon glaucopsis</i>	-0.11	<i>Vaccinium arboreum</i>	-0.02
<i>Eupatorium</i> spp.	-0.12	<i>Vaccinium elliotii</i>	-0.03
<i>Cornus florida</i>	-0.13	<i>Andropogon gyrans</i> var. <i>gyrans</i>	-0.03
<i>Persea palustris</i>	-0.13	<i>Rhynchospora chapmanii</i>	-0.03
<i>Cyrilla racemiflora</i>	-0.13	<i>Aster</i> spp.	-0.04
<i>Prunus serotina</i>	-0.15	<i>Eriocaulon</i> spp.	-0.04
<i>Andropogon ternarius</i>	-0.15	<i>Pteridium aquilinum</i>	-0.04
<i>Balduina uniflora</i>	-0.15	<i>Xyris</i> sp.	-0.06
<i>Cirsium</i> sp.	-0.16	<i>Cornus florida</i>	-0.07
<i>Mitchella repens</i>	-0.16	<i>Chasmanthium sesseliflorum</i>	-0.09
<i>Diodia teres</i>	-0.16	<i>Scleria muhlenbergii</i>	-0.09
<i>Aronia arbutifolia</i>	-0.16	<i>Andropogon mohrii</i>	-0.09
<i>Symplocos tinctoria</i>	-0.17	<i>Smilax laurifolia</i>	-0.10
<i>Viola primulifolia</i>	-0.17	<i>Sporobolus junceus</i>	-0.10
<i>Panicum verrucosum</i>	-0.18	<i>Acer rubra</i> var. <i>drummondii</i>	-0.10
<i>Sporobolus junceus</i>	-0.18	<i>Vaccinium darrowi</i>	-0.11
<i>Elephantopus</i> sp.	-0.18	<i>Myrica heterophylla</i>	-0.12
<i>Andropogon gerardii</i>	-0.18	<i>Fuirena</i> sp.	-0.14
<i>Solidago</i> spp.	-0.18	<i>Dichromena latifolia</i>	-0.14
<i>Chasmanthium laxum</i>	-0.19	<i>Chaptalia tomentosa</i>	-0.14
<i>Sorghastrum secundum</i>	-0.21	<i>Sarracenia alata</i>	-0.17
<i>Paspalum</i> sp.	-0.22	<i>Balduina uniflora</i>	-0.17

Table 1.9 continued.

<i>Vaccinium stamineum</i>	-0.22	<i>Andropogon ternarius</i>	-0.17
<i>Licania michauxii</i>	-0.22	<i>Tridens ambiguus</i>	-0.19
<i>Pteridium aquilinum</i>	-0.23	<i>Smilax pumila</i>	-0.19
<i>Smilax pumila</i>	-0.24	<i>Rhynchospora</i> spp.	-0.19
<i>Pinus palustris</i>	-0.27	<i>Eragrostis refracta</i>	-0.19
<i>Quercus nigra</i>	-0.29	<i>Quercus nigra</i>	-0.20
<i>Andropogon virginicus</i>	-0.30	<i>Nyssa biflora</i>	-0.21
<i>Euthamia</i> spp.	-0.30	<i>Ctenium aromaticum</i>	-0.22
<i>Smilax glauca</i>	-0.31	<i>Pinus palustris</i>	-0.24
<i>Gaylussacia mosieri</i>	-0.31	<i>Sorghastrum secundum</i>	-0.24
<i>Aristida longespica/oligantha</i>	-0.33	<i>Rhynchospora</i> spp. "filamentous"	-0.24
<i>Ilex coriacea</i>	-0.33	<i>Scleria</i> spp.	-0.25
<i>Vaccinium arboreum</i>	-0.35	<i>Andropogon virginicus</i>	-0.25
<i>Gelsemium sempervirens</i>	-0.35	<i>Juncus</i> sp.	-0.27
<i>Vaccinium elliotii</i>	-0.41	<i>Andropogon gyrans</i> var. <i>stenophyllus</i>	-0.27
<i>Ilex vomitoria</i>	-0.44	<i>Hypericum</i> spp.	-0.28
<i>Dichanthelium</i> spp.	-0.45	<i>Aristida palustris</i>	-0.31
<i>Andropogon gyrans</i> var. <i>gyrans</i>	-0.45	<i>Aristida longespica/oligantha</i>	-0.33
<i>Vaccinium darrowi</i>	-0.48	<i>Helianthus angustifolius</i>	-0.33
<i>Schizachyrium tenerum</i>	-0.53	<i>Schizachyrium tenerum</i>	-0.48
<i>Ilex glabra</i>	-0.54	<i>Dichanthelium</i> spp.	-0.53
<i>Schizachyrium scoparium</i>	-0.65	<i>Schizachyrium scoparium</i>	-0.55

bog stands can be expected to have less FACU and FAC graminoids and more OBL forbs and UPL and FACW graminoids than the other stand types. The UPL graminoid guild comprised only one species, *Anthaenaria villosa* (green silkscale). The mean (\pm SE) plant composition PC 1 scores for upland, RCW, and bog stands were 0.26 ± 0.26 , 1.06 ± 0.19 , and -0.79 ± 0.23 , respectively (Fig. 1.12). There were no significant differences in means of the remaining plant composition PCs among stand types. Raw vegetation structure and plants species composition guild means for each stand type are reported in Table 1.12.

MANOVA results showed that mean Dimension 1 scores from nonmetric multidimensional scaling of plant species composition are statistically different among stand

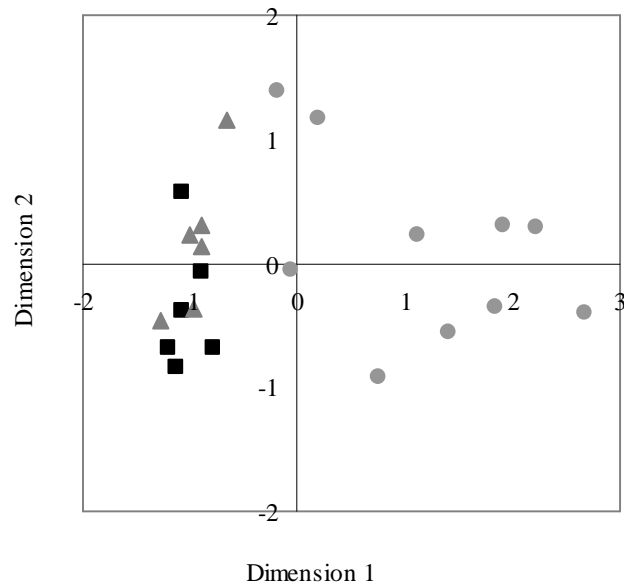


Figure 1.10. Study sites plotted on two dimensions from nonmetric multidimensional scaling of plant species composition in DSNF, MS. Triangles are upland stands, squares are RCW stands, and circles are bog stands.

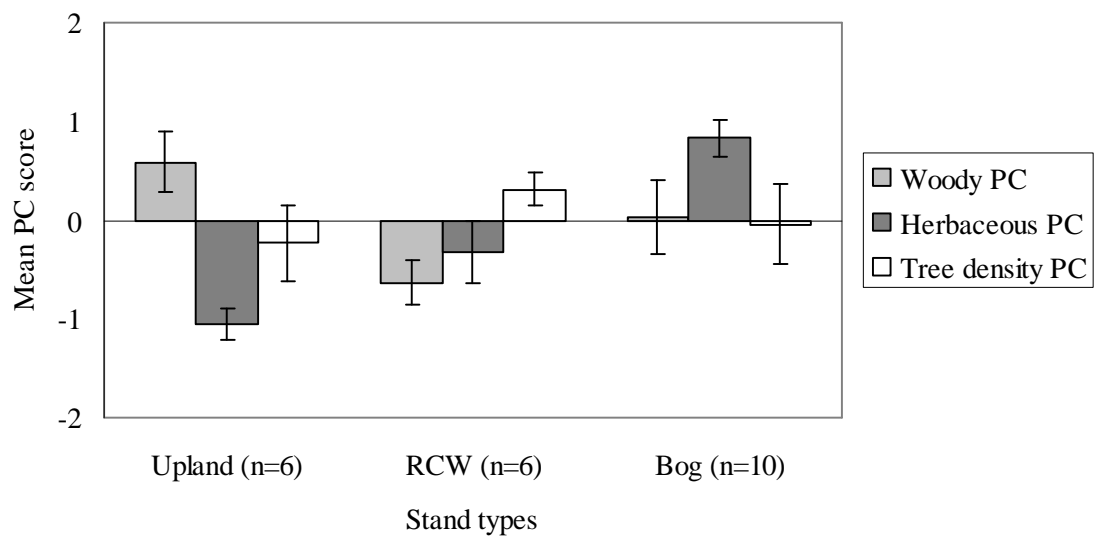


Figure 1.11. Mean (\pm SE) PC scores for vegetation structure among stand types in DSNF, MS. Bog stands had significantly higher mean herbaceous scores.

Table 1.10. Differences in mean PC scores among three stand types in DSNF, MS, estimated from Tukey–Kramer pairwise tests. The three PCs represent herbaceous structure, woody structure, and tree density. Statistically significant differences are in bold.

Comparisons	Difference in means	95% CI
Herbaceous PC		
Bog/RCW	1.17	0.38–1.95
Bog/Upland	1.88	1.10–2.67
RCW/Upland	0.72	-0.16–1.59
Woody PC		
Bog/RCW	0.67	-0.55–1.89
Bog/Upland	-0.56	-1.78–0.67
RCW/Upland	-1.23	-2.59–0.14
Tree density PC		
Bog/RCW	-0.35	-1.70–0.99
Bog/Upland	0.19	-1.16–1.54
RCW/Upland	0.54	-0.96–2.05

types (Wilk’s $\lambda = 0.21$, $F_{4, 36} = 10.52$, $P < 0.001$). Bog stands had higher mean scores than upland and RCW stands ($P < 0.0001$ for both pairwise tests; Table 1.13). Wetland plants were strongly correlated with the upper positive values of Dimension 1; thus, as would be expected, species composition of bog stands consists of more wetland plants than the other stand types. Mean (\pm SE) Dimension 1 scores for upland, RCW and bog stands were -0.95 ± 0.08 , -1.03 ± 0.06 , and 1.19 ± 0.31 , respectively (Fig. 1.12).

DISCUSSION

Bird Species Richness and Total Densities.—Grassland bird species richness and total bird density did not significantly differ by stand type or salvage treatment. Red-cockaded Woodpecker stands were the only ones that contained all three species of grassland bird, but only one of the six sites sampled was occupied concurrently by all three species. Four sites were occupied by only one of the three species, and one site had a Henslow’s Sparrow and a bird

identified to *Ammodramus* sp./Bachman's Sparrow. Although total grassland bird density did not vary significantly among stand types, this may have been due to a lack of statistical power resulting from a small sample size. The large variance in abundance means among stand types may also have been a factor. A graph of the untransformed total bird densities (Fig. 1.3) clearly shows that bog stands have a higher mean density, but the standard error is largest for this stand type. The larger mean for bog stands was driven by the high abundances of Henslow's Sparrows that occurred at some sites. A larger sample size would perhaps have shown differences in total bird densities among stands.

Table 1.11. Differences in mean PC scores for five plant species composition PCs among three stand types in DSNF, MS, estimated from Tukey–Kramer pairwise tests. The PCs represent plant species composition guilds based on plant life forms and Wetland Indicator Status. Significant differences are in bold.

Comparisons	Difference in means	95% CI
Plant PC 1		
Bog/RCW	-1.84	-2.68--1.01
Bog/Upland	-1.04	-1.88--0.21
RCW/Upland	0.80	-0.13–1.74
Plant PC 2		
Bog/RCW	0.12	-1.15–1.39
Bog/Upland	-0.80	-2.07–0.47
RCW/Upland	-0.92	-2.34–0.50
Plant PC 3		
Bog/RCW	0.50	-0.85–1.84
Bog/Upland	0.02	-1.33–1.36
RCW/Upland	-0.48	-1.99–1.02
Plant PC 4		
Bog/RCW	0.61	-0.58–1.79
Bog/Upland	1.20	0.02–2.39
RCW/Upland	0.60	-0.73–1.92
Plant PC 5		
Bog/RCW	0.09	-1.17–1.34
Bog/Upland	-0.87	-2.13–0.38
RCW/Upland	-0.96	-2.37–0.44

Vegetation Structure and Plant Species Composition.—Herbaceous vegetation structure in bog stands was statistically different from that in upland and RCW stands. Bogs tended to have higher and less heterogeneous herbaceous cover and density. These results are consistent with other studies. For example, Kirkman et al. (2001) found that wet–mesic sites in longleaf pine–wiregrass savannas in Georgia had higher ground cover biomass than xeric sites. Note that ground-layer biomass has been reported to be positively correlated with herbaceous cover in pine

Table 1.12. Means of vegetation structure measurements and plant species composition guild values among three stand types in DSNF, MS.

Variable	Upland	SE	RCW	SE	Bog	SE
Tree basal area (m ² /20 m ²)	0.45	0.06	0.51	0.06	0.29	0.07
Canopy closure (%)	33.65	5.14	31.20	2.40	37.73	7.96
Herb cover (%)	42.87	4.54	63.93	3.78	69.52	4.54
Herb cover CV	60.46	4.40	37.51	5.38	36.35	6.53
Herb height (cm)	19.14	2.15	14.68	0.52	16.20	1.14
Herb density (# hits < 10 cm)	5.08	0.55	4.65	0.49	6.77	0.26
Herb density CV	73.72	5.99	49.91	2.99	46.81	3.16
Woody cover (%)	20.95	1.54	10.59	1.28	13.35	2.79
Woody cover CV	75.84	4.25	96.34	7.72	151.27	14.99
Number stems (# ranks)	25.52	2.28	18.91	2.85	17.60	3.56
Woody height (cm)	43.62	2.63	15.93	2.60	18.87	3.58
Woody density (# hits < 10 cm)	0.41	0.06	0.13	0.06	0.19	0.05
Graminoid UPL	0.00	0.00	0.00	0.00	0.04	0.02
Graminoid FACU	17.19	2.83	20.15	3.81	2.97	1.05
Graminoid FAC	19.80	3.60	36.82	3.98	7.78	1.44
Graminoid FACW	5.09	1.80	6.03	2.83	33.08	4.91
Graminoid OBL	1.62	0.28	1.90	1.41	20.90	4.86
Forb UPL	0.33	0.26	0.03	0.03	0.08	0.04
Forb FACU	0.25	0.18	0.25	0.16	0.01	0.01
Forb FAC	0.15	0.05	0.77	0.66	1.40	0.50
Forb FACW	0.07	0.05	0.07	0.07	0.77	0.37
Forb OBL	0.01	0.00	0.00	0.00	11.56	3.43
Woody UPL	0.05	0.03	0.08	0.08	0.00	0.00
Woody FACU	1.03	0.28	0.41	0.21	0.02	0.01
Woody FAC	4.79	0.84	1.05	0.40	1.41	0.41
Woody FACW	16.19	1.61	8.96	1.53	11.59	2.59
Woody OBL	0.49	0.47	0.00	0.00	0.89	0.47

savannas (Fuller 2004). Walker and Peet (1983), working in pine–wiregrass savannas in North Carolina, also found higher aboveground biomass in mesic savannas compared to dryer sites.

One explanation for this difference in DSNF is that in drier upland sites, belowground competition for moisture may be more intense, with woody plants limiting the establishment of

Table 1.13. Differences in mean dimension scores from NMDS of plant species composition among three stand types in DSNF, MS, estimated from Tukey–Kramer pairwise tests. Statistically significant differences are in bold.

Comparisons	Difference in means	95% CI
Dimension 1		
Bog/RCW	2.22	1.30–3.13
Bog/Upland	2.14	1.23–3.05
RCW/Upland	-0.08	-1.10–0.95
Dimension 2		
Bog/RCW	0.46	-0.39–1.30
Bog/Upland	-0.05	-0.90–0.79
RCW/Upland	-0.51	-1.46–0.43

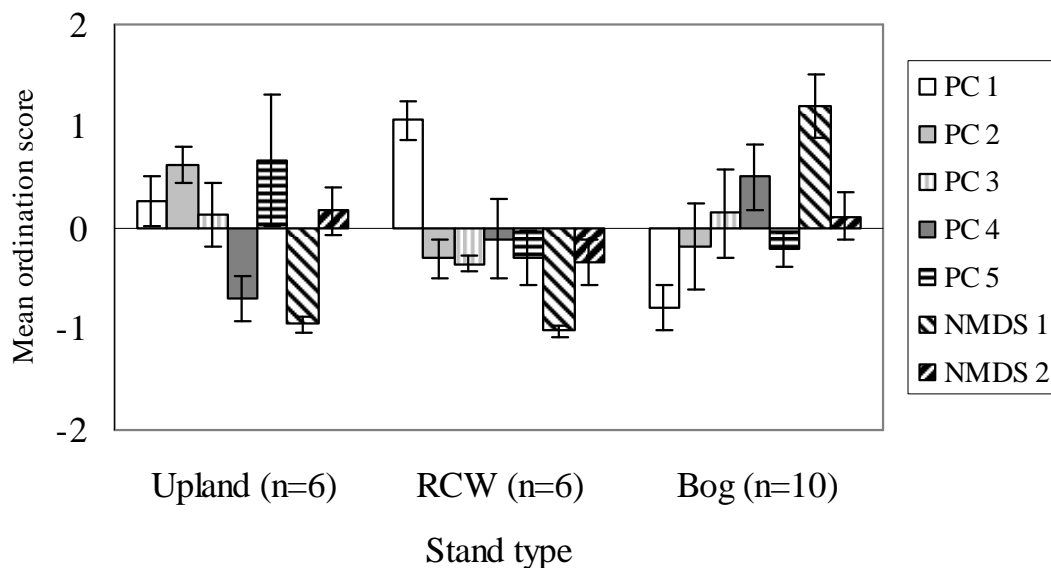


Figure 1.12. Mean (\pm SE) PC and NMDS scores for ordinations on plant species composition guilds and plant species composition, respectively, among stand types in DSNF, MS. Bog stands had significantly higher scores for the first PC and NMDS Dimension 1, both of which represent wetland plants.

herbaceous species and thus creating patches of bare ground and shrub thickets (Ludwig et al. 2004a, Ludwig et al. 2004b). Competition in bogs may be reduced because of plentiful moisture, enabling the coexistence of multiple species of woody and herbaceous plants (Kirkman et al. 2001). Although not significantly different, RCW stands had more herbaceous cover and structure than upland sites. This is consistent with Masters et al. (1996), whose growing season sampling in Arkansas showed that forb, legume, grass, and sedge standing crop biomass were all higher in woodpecker-managed stands than in non-managed stands.

Woody understorey vegetation structure did not differ significantly among stand types, except that woody cover tended to be more heterogeneous in bog stands, as reflected by the high loading of the woody cover coefficient of variation on the herbaceous structure PC. Although the difference was not significant, the mean woody understorey vegetation PC score was lower in RCW stands than in upland and bog stands. Red-cockaded Woodpecker stands are managed by the reduction of woody midstorey and understorey vegetation via mowing and prescribed fire; thus, one would expect this stand type to have less woody understorey vegetation as has been shown in numerous studies. Visual inspection of the managed woodpecker stands in DSNF lead one to conclude that these stands are less woody than many upland stands not managed for woodpeckers, although my sampling did not show this expected difference. The lack of difference in woody vegetation, besides heterogeneity, between upland and bog stands is interesting because, historically, woody species such as *Ilex* spp. occurred at low densities in bogs (Folkerts 1982, Bridges and Orzell 1989, Olson and Platt 1995, Brewer 1998). However, Brewer (2002), working in DSNF, found that densities of *Ilex coriacea*, *I. glabra*, and *I. vomitoria* were higher in bogs that had been used for timber production. Not only can soil disturbance caused by timber management increase the seedling emergence of woody plants, but increased tree densities from tree planting and intrusion can increase the seed deposition and

abundance of woody understorey plants by providing perches for birds that disperse the seeds (Brewer 1998, 2002, Hinman et al. 2008). Past fire suppression and more-recent dormant season prescribed fires have also increased woody understorey plant density in bogs (Hodgkins 1958, Boyer 1992, Drewa et al. 2002a, b). Even so, while many bogs in DSNF have dense woody understorey vegetation comparable to upland stands, the herbaceous ground cover is not sparse and heterogeneous as in upland stands, and woody cover in bogs tends to be patchy, perhaps occurring in drier areas. It must be noted that an exception to this pattern occurs in bogs where high densities of crawfish chimneys and mounds increase herbaceous patchiness (see Chapter 2 Discussion).

Plant species composition in bog stands differed statistically from that in upland and RCW stands. These differences largely conformed to expected habitat associations of herbaceous wetland and upland species. In general, bogs had more FACW graminoids and OBL forbs, but less FAC and FACU graminoids than both other stand types. Bogs had more OBL graminoids and FAC forbs but less FACU woody plants than upland stands. Multivariate analysis of variance of NMDS and PC scores of plant species composition clearly showed that bog stands had different species composition than the other two stand types. This pattern is not surprising because the transition from uplands to bogs represents a moisture gradient. Plant species composition changes according to the moisture requirements of particular plants, and plant species richness tends to peak in bogs (Walker and Peet 1983, Kirkman et al. 2001). Bogs have high pH levels and are nutrient poor, even compared to adjacent upland pine savannas (Folkerts 1982, Kirkman et al. 2001). This, along with moisture preferences, refines the process of habitat filtering, and only plants that are adapted to these extreme conditions will occur in bogs (Kirkman et al. 2001, Webb et al. 2002), hence the large number of endemics.

Only one woody plant guild differed statistically among stand types. This reflects that the woody plant community in DSNF and most pine savannas is composed of habitat generalists. Drewa et al. (2002a) observed distinct herbaceous plant communities in bogs, but noted that woody plant composition was similar between upland and bog sites. My results also suggest that in DSNF, management for Red-cockaded Woodpeckers does not significantly alter plant species composition compared to non-managed upland stands. However, if I had sampled throughout the growing season, I would have inevitably detected more plant species, perhaps revealing differences between the two stand types for part of the annual cycle.

Bachman's Sparrow.—Bachman's Sparrows occurred in upland and RCW stands but not in bog stands. These results support those of Allen et al. (2006) who found that Bachman's Sparrows were more common in upland habitats compared to wetter pocosins in North Carolina longleaf pine savannas. One reason Bachman's Sparrows did not occur in bogs is simply sensitivity to moisture. Bogs are often filled with standing water (Folkerts 1982), and some species of ground-dwelling birds may prefer dryer habitats. For example, Chipping Sparrows and Dark-eyed Juncos often forage on the ground, but neither species was observed in bog habitats. I do not know of any studies documenting Bachman's Sparrow use of pitcher plant bogs.

Although I detected birds at some of my sites that could not be identified to species, only one of these ever occurred in a bog stand. I think that some of the unidentified birds and many of the birds identified to *Ammodramus* sp./Bachman's Sparrows were, in fact, Bachman's Sparrows because of their actions when we attempted to catch them. In my experience, these sparrows are much harder to catch than *Ammodramus* sparrows. *Ammodramus* are secretive, but after they flush from a spot and fly to a new location, they tend to remain in that general location, making chasing and catching easier. Bachman's Sparrows, however, in the winter tend to flush

once and run after landing (Dunning 1993). I frequently observed flushed Bachman's Sparrows flying to near-by upturned root balls where they would presumably enter tunnels under the root balls to take protection. Bachman's Sparrows have been documented using underground burrows (Dean and Vickery 2003), and one of our technicians witnessed a flushed Bachman's Sparrow going into a hole underneath a root ball. Upturned root balls are a common feature in mature pine savannas in DSNF due to the damage caused by Hurricane Katrina. I was never able to relocate birds once they flushed towards a root ball. Although the actions of many unidentified birds were reminiscent of Bachman's Sparrows, I did not feel confident making an identification call based on this alone. Density estimates for Bachman's Sparrows may be biased low leading to the lack of significant differences among stand types. Nonetheless, I am confident that Bachman's Sparrows did not occur in bogs during the winter, although there may have been differences in bird densities between upland and RCW stands that I did not detect because of unidentified Bachman's Sparrows.

Based on our observations and data from the breeding-season component of this project, Bachman's Sparrows are common in savaged-logged sites that have been recently burned. Of the sites I surveyed, four of the five that contained Bachman's Sparrows were salvaged logged. In contrast, Dunning and Watts (1991), who surveyed Bachman's Sparrows in post-salvage-logged Francis Marion National Forest in South Carolina after Hurricane Hugo, found that post-hurricane sparrow occupancy in clear-cut stands increased after the hurricane. They hypothesized that birds were leaving mature, salvage-logged stands because the logging had destroyed much of the habitat. The authors, however, said nothing about the fire history of their study sites and how this may have affected stand occupancy before and after the hurricane (Dunning and Watts 1991). Bachman's Sparrow abundances are higher in recently-burned stands because fire helps maintain adequate herbaceous vegetation (Tucker et al. 2004, Cox and

Jones 2009). In DSNF, salvage logging does not appear to have destroyed the habitat in the manner described by Dunning and Watts (1991) because there are few signs of damage to the herbaceous layer (*e.g.*, skidder tracks). There is also evidence that Bachman's Sparrows prefer habitats with low tree density (Haggerty 2000), which would be the case in salvaged stands.

Bachman's Sparrows are typically associated with mature longleaf pine savannas with dense herbaceous ground cover and low shrub cover (Dunning and Watts 1990, Haggerty 1998, Plentovich et al. 1998a, Tucker et al. 2004). This association, however, does not appear to hold true in DSNF in winter because birds seem to prefer upland stands, most of which have high shrub cover and patchy herbaceous cover. Cox and Jones (2009) found evidence that Bachman's Sparrows use the same territories in winter and in the breeding season and will maintain year-round home ranges, which implies that it may be appropriate to apply habitat preference results from breeding season studies to winter ecology and vice versa. Some studies offer support for the trends I observed in DSNF. Haggerty (1998) suggested that Bachman's Sparrows may prefer patchy herbaceous ground cover because they are associated with cespitose grasses, which may facilitate the capture of prey during the breeding season by increasing the ease of movement of foraging birds. Haggerty (2000), conducting a region-wide study across five states, also found that Bachman's Sparrow preferences of forb cover, vegetation height, and tree density varied widely across regions. Cox and Jones (2009) found that Bachman's Sparrow winter abundances at sites in Georgia were positively correlated with bare ground and were negatively correlated with increased grass structure and shrubs < 1 m in height. Indeed, I detected Bachman's Sparrows in the spring in several stands that had been burned the previous month and lacked any herbaceous layer, the ground being mostly bare with burned shrubs and thousands of small *Ilex* spp. shoots. Variability in habitat preferences is also reflected by their use of both mature and clearcut timber stands (Haggerty 1988, Tucker et al. 1998).

I observed numerous times that Bachman's Sparrows would use tree crowns for singing perches during the breeding season and root balls for predator escape refugia. Perch structure availability was also suggested by Dunning and Watts (1990) as an important component for determining Bachman's Sparrow abundance in clearcuts. The authors observed that sites with high bird abundances were those that had tall shrubs and standing dead timber, from which Bachman's Sparrows would often sing. Sites with low bird abundances had been logged with different methods that did not have standing dead timber or tall shrubs. Dean and Vickery (2003) documented Bachman's Sparrow use of burrows in palmetto clumps as predator escape refugia, where birds would actually hide from predators underground. They hypothesized that burrow use is probably more common in open, treeless habitats than areas with trees where birds can take refuge. I never saw a Bachman's Sparrow flush into a tree, but frequently observed them flushing toward root balls, even in treed stands.

Lack of perch site availability and upturned root balls could explain why Bachman's Sparrows avoided bogs. Most bogs in DSNF either have few trees or have a thick canopy of slash pines. In either case, Hurricane Katrina did little damage to trees in bogs; hence, there are few downed trees and less coarse woody debris in bog habitats. Upland and RCW stands were heavily damaged and salvaged, and these stands are still covered with pine tree crowns. In addition, during the breeding season, the abundance of downed tree crowns and upturned root balls in DSNF increased the probability of stand occupancy by Bachman's Sparrows (unpublished data). If Bachman's Sparrows prefer downed pine crown perches and root balls, this would mean a preference for hurricane-damaged sites, leading one to conclude that natural disturbance, including fire, has played a principal role in creating Bachman's Sparrow habitat in DSNF. Bogs with open canopies may not contain adequate perch sites and escape refugia, while bogs with closed canopies may not be preferable because of high tree densities.

As was found in other studies, Bachman's Sparrows used RCW clusters. Habitat management for the endangered woodpecker creates habitat suitable for Bachman's Sparrows (Dunning and Watts 1990, Wilson et al. 1995, Plentovich et al. 1998a, Wood et al. 2004). I found Bachman's Sparrows in two of the six clusters sampled. Similarly, Plentovich et al. (1998a) found that not all RCW clusters surveyed in Eglin Air Force base in Florida were suitable for Bachman's Sparrows. The mean breeding season home range size of Bachman's Sparrows ranges approximately 1.5–4.8 ha and varies with time since fire, timber age, and vegetation structure (Haggerty 1998, Stomber and Krementz 2006, Cox and Jones 2007). Many of the woodpecker clusters in DSNF are perhaps too small (< 0.5 ha) to be of value to Bachman's Sparrows, particularly if the surrounding habitat is inadequate. Clusters occupied by Bachman's Sparrows could compose only a portion of the bird's total home range; thus, usage may not necessarily imply RCW cluster preference *per se*.

Henslow's Sparrow.—Henslow's Sparrows occurred only in bog and RCW stands. In the Gulf Coast, Henslow's Sparrows seem to prefer some grassland habitats over others. Working in pitcher plant bogs and managed upland pine stands in Alabama, Plentovich et al. (1999) found Henslow's Sparrows only in pitcher plant bogs and transition zones between bog and upland pine habitats. Other studies have found high densities of Henslow's Sparrows in upland longleaf pine habitats (Carrie et al. 2002, Johnson 2006, Palasz 2008). In my study sites in southeastern Louisiana, seven are longleaf pine flatwoods bogs, and three are longleaf pine upland habitats. All ten sites are in excellent condition and are burned annually or biannually; Henslow's Sparrows are abundant in both habitat types. Previously, Plentovich et al. (1999) was the only study to look at site occupancy in a landscape that featured a connected mosaic of upland and bog habitats. While Henslow's Sparrows will use both upland longleaf pine

savannas and bogs, there may be a preference for bogs when both habitat types are found in close proximity—information that would be valuable to forest managers.

Henslow's Sparrows may exhibit less area sensitivity in bogs compared to upland habitats, perhaps because of improved resources. For example, Tucker and Robinson (2003), working in Alabama and Florida pitcher plant bogs, found Henslow's Sparrows occupying bog patches ranging from 0.06–1.17 ha. This is not surprising considering the mean winter home range for Henslow's Sparrows in bogs is 0.3–0.6 ha (Thatcher 2003, Bechtoldt and Stouffer 2005). Henslow's Sparrow densities in DSNF bogs are particularly high compared to other locations that have not only better-maintained bogs, but larger patches (*e.g.*, flatwoods bogs in Louisiana or Mississippi Sandhill Crane National Wildlife Refuge). For example, during a single January survey of the smallest bog I sampled in DSNF, I recorded what is, to the best of my knowledge, the highest Henslow's Sparrow density ever recorded on their wintering grounds (6 birds in 0.2 ha = 30 birds/ha!). A preference for bogs over upland habitats could be a result of drastic differences in habitat quality and land-use history. In places like DSNF, where most upland habitats are degraded and most high-quality habitat exists in bogs, a preference for bogs and a lack of area sensitivity is evident. The differences in the quality of upland longleaf pine habitats studied by various researches are hard to discern, but they may explain why results of Henslow's Sparrow habitat-type preferences differ. More research comparing bird trends in pristine upland and bog habitats would be valuable for determining if Henslow's Sparrows are more abundant or exhibit less area sensitivity in specific habitat types.

Henslow's Sparrow use of RCW stands was ephemeral, indicating that these stands may not be as important as bogs for providing wintering habitat. The mean Henslow's Sparrow density in RCW stands declined from 3.54 in late November to 0.6 in early January. Henslow's Sparrows are site faithful during the core months of winter (December–February; Plentovich et

al. 1998b, Thatcher et al. 2006, Johnson et al. 2009); my results suggest temporary use of RCW stands by transient birds that had not yet settled on winter territories. In the southern portion of their wintering range, Henslow's Sparrows exhibit post-migration movement through November, but by late November most birds have settled and remain site faithful until spring departure in early March (Plentovich et al. 1998b, Thatcher 2003, Johnson et al. 2009). This typical migration trend corresponds loosely with what I observed in DSNF. Perhaps in areas with low-quality habitat patches, birds are forced to spend more time moving around looking for suitable habitat. Red-cockaded Woodpecker stands had lower herbaceous structure than bog stands, indicating that the habitat was of lower quality than bogs where bird densities did not significantly decline during the same year of sampling. Most studies that have assessed site fidelity of Henslow's Sparrows on the wintering grounds have focused on larger, more pristine and homogeneous habitat patches where site fidelity may be easier if resources are not limited in late winter. Patch size, too, could be another reason that birds did not remain in woodpecker stands. Henslow's Sparrows occupy small habitat patches in high-quality bogs, but, as mentioned above, could exhibit more area sensitivity in lower-quality habitats. Predation, which may be greater for grassland birds in areas with low herbaceous cover, could have also played a role in bird density decline (Shriver 1996, Perkins and Vickery 2001). Despite the ephemeral use of RCW stands by Henslow's Sparrow, there was still at least one bird using a stand in January and, along with the occurrence of other grassland birds, this corresponds with previous studies suggesting that RCW habitat management benefits grassland birds.

Perhaps the primary reason why Henslow's Sparrows avoided or abandoned upland habitats in DSNF is the lack of a dense, spatially homogenous herbaceous layer, even in stands regularly managed with fire. Henslow's Sparrow abundance is highest within the first several years following fire because fire helps maintain the dense herbaceous cover required by birds

(Chandler and Woodrey 1995, Carrie et al. 2002, Tucker and Robinson 2003, Bechtoldt and Stouffer 2005, Johnson et al. 2009). Most upland stands I sampled had experienced only one growing season since fire and were in their best condition possible for Henslow's Sparrows, yet the herbaceous layers were still too sparse and patchy to provide suitable habitat.

Henslow's Sparrows occurred in both salvaged and unsalvaged stands. The majority of salvaged-stand occurrences were in RCW stands. The one salvaged bog stand sampled had Henslow's Sparrows both years of the study. Because birds occupied a salvaged bog and salvaged RCW stands, I conclude that the apparent absence of Henslow's Sparrows in upland salvaged stands is due not to salvage logging, but attributable to the inadequate vegetation structure that has resulted from historic and recent land use (*i.e.*, fire suppression and dormant season fires). In recent years, the Forest Service has been increasing the yearly proportion of growing season prescribed fires in DSNF, and this may increase the effectiveness of habitat restoration with fire. Logging operations can have negative effects on grassland birds (Dunning and Watts 1990, 1991), but the strict monitoring of salvage operations in DSNF by Forest Service personnel may have reduced many of the discernable negative impacts on the landscape. Bogs are inevitably more impacted by timber operations because moist soil is more sensitive to disturbance (Dunning and Watts 1991, Brewer 2002). Consequently, prohibition of timber harvesting, include salvaging, in bogs is crucial for protecting rare plants and animals.

Based on the trends I observed, I conclude that DSNF does not compose a major portion of Henslow's Sparrow wintering grounds. This is because most of the grassland habitat in DSNF consists of upland stands, and because even with recent fire these stands do not attract Henslow's Sparrows. Compared to places like the Mississippi Sandhill Crane Refuge and others managed specifically for the restoration of longleaf pine herbaceous vegetation, DSNF has low abundances of wintering Henslow's Sparrows relative to its total size. I do not want to imply

that management for Henslow's Sparrows in DSNF is not important. On the contrary, these birds are increasingly dependent on small, isolated patches of habitat across the southeastern U.S. in the winter, particularly during migration when birds are more vulnerable, and DSNF provides these habitats. As the grassland habitat in DSNF is improved, it will become more suitable for all three common wintering grassland bird species occurring in the forest. The crucial decision is on which stand type to focus restoration efforts. Upland stands are already suitable for two of the three grassland bird species, and if restored, they may become suitable for Henslow's Sparrow. Bogs can be maintained with slash pine removal and continual prescribed fire, but upland stands require a reduction in shrub cover, which will not happen from prescribed fire alone—especially dormant season fires—but requires a combination of fire, mechanical removal, and herbicide application (Boyer 1992, Olson and Platt 1995, Drewa et al. 2002b). For Henslow's Sparrows, small-scale bog restoration will probably be the most effective management strategy.

Sedge Wren.—Sedge Wrens occurred in all stand types and salvage treatments; densities did not differ among stands or between salvage treatments. Even so, only one Sedge Wren was ever detected in RCW stands. Sedge Wrens were not detected in either of two studies looking at winter bird communities in RCW clusters (Conner et al. 2002, Provencher et al. 2002). Little is known about Sedge Wren winter ecology, but modeling of stand occupancy based on vegetation variables suggests Sedge Wrens in DSNF may prefer woody understorey vegetation (see Chapter 2 Results). Woody understorey vegetation is abundant in all stand types and salvage treatments in DSNF but is lowest, albeit not statistically, in RCW clusters. Although my results do not show statistically fewer Sedge Wrens and lower woody structure in woodpecker clusters, I suspect that there are differences and that further sampling may reveal significant differences.

Bird Abundance Decline over Winter and Annual Variation.—Henslow's Sparrow and Sedge Wren abundances declined over the first winter of sampling in DSNF, but not over the second winter. It appears that there is annual variation in the winter grassland bird trends in DSNF. Previous work with Henslow's Sparrows has shown that steady, but slight, declines in bird abundances occur over winter after mid-December (Johnson et al. 2009). There are several explanations for this decline: birds left the bog stands in which they first occurred, there was unusually high mortality, or both. Henslow's Sparrows exhibit high site fidelity during midwinter (Plentovich et al. 1998b, Thatcher et al. 2006, Johnson et al. 2009), so mid-winter movement seems unlikely, particularly in high-quality bog habitats unless there is a resource shortage. In the Lower East Gulf Coastal Plain ecoregion, DSNF is one of the most southerly wintering grounds for Henslow's Sparrow, and limited habitat patches could become saturated with birds, creating a shortage of resources because of intraspecific competition. If resources were limited in small, isolated bogs, then birds may have been forced to search for better food resources. The Mississippi Sandhill Crane National Wildlife Refuge is about 20 km southeast from the southern end of DSNF. This refuge has more high-quality pine savannas and pitcher plant bogs and has a large population of wintering Henslow's Sparrows (K. Hackman, pers. comm). It is possible that birds left DSNF for better habitat at the crane refuge. Although such winter movement may not be typical in many Henslow's Sparrow habitats, it is possible: in Louisiana, a Henslow's Sparrow that was captured at the same location once in November and once in December was captured a third time in January approximately 20 km from the original place of capture (Johnson 2006).

High predation and stress-induced mortality during the winter of 2007–2008 because of low precipitation the previous growing season is perhaps the best explanation for the observed over-winter decline. Thatcher et al. (2006), working in the Mississippi Sandhill Crane NWR,

observed that Henslow's Sparrow survival over two winters was best explained by time since fire and sampling year, with lower survival rates the first year of sampling. They hypothesized that drought during the previous growing season could have affected habitat quality and food resources. They observed that Henslow's Sparrow home ranges were larger during the first sampling year, which may have lead to increased movement and predator exposure. Similar patterns have been observed in grasslands in southwestern North America (Macías-Duarte et al. 2009). For example, Pulliam and Parker (1979) showed that seed resources can be limited for some wintering sparrow species after growing season droughts, and winter sparrow densities in Arizona were shown to be negatively correlated with rainfall from the previous growing season (Dunning and Brown 1992). Indeed, in 2007, precipitation at the Saucier Experimental Forest in DSNF was 4–159 mm (mean = 74 mm) below average every month from March–November except for above-average precipitation in October (85 mm above average), while the 2008 growing season received a mixture of above- and below-average precipitation (National Climatic Data Center 2009). This may have limited vegetation growth and seed production and lead to greater mortality because of predation and limited resources.

Caution must be taken when interpreting the trends observed during this study because they are based on only two seasons of observations, and as with most studies, long-term research is needed to elucidate real patterns in annual variation. I think the current conditions in DSNF influence the observed habitat-type preferences of the birds I studied. Trends such as over-winter decline in bird abundances and the ephemeral use of RCW stands by Henslow's Sparrows could be isolated, stochastic events, or they could be common. Because these habitats change so rapidly between years and after fire, the quality of habitat types will always be in flux, and specific habitats may appeal to birds differently between years. More research is needed to address lingering questions such as Sedge Wren winter site fidelity, Sedge Wren use of RCW

stands, and the long-term role of isolated bog patches in providing Henslow's Sparrow winter habitat, particularly compared to upland habitats or larger bogs. Research looking at over-winter survival of grassland birds in DSNF, especially compared to survival in high-quality habitats, is needed. If over-winter decline in bird abundance is common, then the forest could be serving more as a wintering-ground sink, with less-than-average over-winter survival, although this may be a natural part of the system (Pulliam 1988).

Comments on Identification Issues.—Birds were detected in upland and RCW stands that could not be identified to species, and this may have altered density estimates in these stands. Some Henslow's Sparrows could have occurred in upland sites, and Bachman's Sparrow and Sedge Wren numbers were probably underestimated because they could not be identified; still, the general trend of Henslow's Sparrow bog and RCW stand preferences over regular upland stands—salvaged or unsalvaged—is apparent. Identification issues, I think, were more problematic than detection probability issues. I am confident that most birds on transect were detected, but not all detections were identified. Reliance on volunteers that often lacked experience with bird identification was a problem in both upland and RCW sites where any grassland or shrub-scrub bird could have occurred. Often, a bird would go unidentified because a volunteer would see the bird and announce it to the team, but the bird would disappear into the forest edge before anyone else could see it. Many birds that were partially identified were chased, which allowed for better looks, but the high density of shrubs made escape easy, and the operation of a mist net in these shrubs hindered capture. Even for a team of skilled birders, grassland birds can be extremely hard to identify in the winter, and of the many locations where I have surveyed grassland birds, DSNF was the most difficult. Thus, the issue of unidentified birds is a caveat of this study that could not be avoided. The lack of detection probabilities is another caveat, but I have already rationalized why these could not be reliably estimated.

Conclusions

It is difficult to determine any direct effects that post-Katrina salvage logging may have had on grassland birds, and I argue that inferences about salvage-logging effects based solely on the observed bird–stand trends would be misleading. I think that land-use history and prescribed fire have had a much greater effect on grassland bird habitat in DSNF. Indirect effects may take years to become apparent. For example, the opening of the canopy from the hurricane may benefit the herbaceous layer in the long term and thus may benefit grassland birds. The majority of the grassland habitats in DSNF consist of salvage-logged, upland pine savannas. These areas are suitable for Bachman’s Sparrows and Sedge Wrens, but they currently do not support substantial numbers of wintering Henslow’s Sparrows. Upland stands are improved by management for RCWs and are used by grassland birds, but RCW stands make up only a small fraction of the total grassland area in DSNF. I recommend that forest managers continue to focus on the restoration of bogs by removing invasive slash pines and with prescribed fire to increase the amount of suitable habitat. Increasing the number and size of RCW clusters will also benefit the wintering grassland bird community. Restoration of shrub-encroached upland pine habitat is more of a challenge, but frequent prescribed fire appears to be adequate for providing Bachman’s Sparrow and Sedge Wren winter habitat.

CHAPTER 2. HABITAT PREFERENCES OF HENSLOW'S SPARROWS AND SEDGE WRENS WINTERING IN DE SOTO NATIONAL FOREST, MS, USA

INTRODUCTION

It has long been proposed that habitat structure is a primary factor determining site occupancy and community composition of birds. Work by MacArthur and colleagues in the 1960s demonstrated that bird species diversity increased in relation to the number of vertical and horizontal vegetation layers, or complexity, in forest ecosystems (MacArthur and MacArthur 1961, MacArthur 1965, MacArthur et al. 1966). In the 1970's and 1980's, Wiens and Rotenberry, working with grassland and shrubsteppe birds, extended the idea of habitat structure to include measurements of local heterogeneity, or patchiness, and the application of different spatial scales of study (Wiens 1974, 1976, Rotenberry and Wiens 1980, Wiens and Rotenberry 1981, Wiens 1989, Kotliar and Wiens 1990). These pioneering studies led the way in promulgating the importance of small-scale habitat structure and patch sensitivity for many grassland birds (Herkert 1994a, b, Johnson and Igl 2001). Plant species composition may be important for grassland birds by directly shaping habitat structure and by providing preferred food resources (Grzybowski 1982, Moorcroft et al. 2002). Specific plant species or guilds can be useful indicators for predicting the occurrence or abundance of certain species, particularly when birds prefer specific species for food or nesting substrates (Plentovich et al. 1999, Rider et al. 2006, Mitchell and Wilson 2007).

Various terms and definitions have been used to describe habitat structure. McCoy and Bell (1991) described habitat structure as three distinct components: heterogeneity, complexity and scale. Heterogeneity refers to the variation in the distribution or abundance of different structural aspects, for example, a continuous or patchy tree canopy. Complexity refers to the total amount of different structural components, such as the number of vertical layers in a forest.

Scale is the area considered when measuring heterogeneity and complexity and must be appropriate for the organisms studied.

Henslow's Sparrow (*Ammodramus henslowii*), a bird that responds to specific vegetation structure, is a grassland species of concern occurring in midwestern and eastern U.S. and southeastern Canada (Fig. 2.1; Herkert 1994a, Herkert et al. 2002). Henslow's Sparrows breed from extreme northern Arkansas to Kansas and Minnesota and east throughout the Midwest to Pennsylvania and southern Ontario (Herkert et al. 2002). It is endangered or threatened in 16 U.S. states and endangered in Canada (Burhans 2002). The wintering range spans from eastern Texas through Florida and southeastern North Carolina (Herkert et al. 2002, Holimon et al. 2004). Between 1966 and 1984, Henslow's Sparrows declined at an average annual rate of 8.6%—primarily from habitat loss—making it the fastest declining songbird in the U.S. during that time (Sauer et al. 2008). Currently, the overall population is stable; populations are increasing in the midwestern breeding range and declining in the eastern range (Sauer et al. 2008). Because of their secretive nature on the wintering grounds, Henslow's Sparrows are difficult to monitor, and until the last 15 years virtually nothing was known about their winter ecology (Chandler and Woodrey 1995, Pruitt 1996).

A number of regional studies conducted in the last 15 years have helped elucidate the winter ecology and habitat preferences of Henslow's Sparrows. The majority of Henslow's Sparrows winter in habitats maintained by periodic fire (Pruitt 1996). They prefer the dense, herbaceous ground layer and low litter typical of frequently-burned grasslands; thus, as time since fire increases, Henslow's Sparrow densities decrease (Chandler and Woodrey 1995, Plentovich et al. 1999, Carrie et al. 2002, Tucker and Robinson 2003, Bechtoldt and Stouffer 2005, Johnson 2006, Holimon et al. 2008). In longleaf pine (*Pinus palustris*) and slash pine (*Pinus elliottii*) savannas, shrub encroachment following fire suppression leads to afforestation



Figure 2.1. Henslow's Sparrow range map (from Herkert et al. 2002).

and the disappearance of the herbaceous layer (Brockway et al. 2005). Several studies have shown that Henslow's Sparrows occupy sites with low shrub cover; however, they can tolerate shrubs as long as a dense herbaceous layer still exists (Carrie et al. 2002, Holimon et al. 2008, Palasz 2008). In Louisiana, Henslow's Sparrows may be negatively associated with tree density (Carrie et al. 2002). This may result from the effects of a closed canopy on the herbaceous layer in longleaf pine savannas because herbaceous plant diversity tends to be higher in open areas (Brewer 1998, Harrington and Edwards 1999, Platt et al. 2006).

Because Henslow's Sparrows winter in a variety of grassland habitats, the importance of specific plant species—be it for structure or food supply—in predicting sparrow occupancy or abundance varies among regions and habitat types. In southern Alabama, Henslow's Sparrows occupancy was best predicted by the presence of *Sarracenia* spp. (pitcher plant) and the disturbance-loving grass *Panicum verrucosum* (warty panicgrass; Plentovich et al. 1999), while no relation between sparrow occupancy and pitcher plants was found in a similar study in Alabama and Florida (Tucker and Robinson 2003). In soil barrens in southern Arkansas,

Henslow's Sparrow flush points had higher cover of *Aristida* spp. (threeawn) and *Rhynchospora globularis* (globe beaksedge) than random points (Holimon et al. 2008). Johnson (2006) found that in southern Louisiana longleaf pine flatwoods, Henslow's Sparrow abundance increased with cover of *Andropogon* spp. (broom sedge), *Schizachyrium scoparium* (little bluestem), *S. tenerum* (slender bluestem), *Muhlenbergia expansa* (cutover muhly) and *Ctenium aromaticum* (toothache grass), and a variety of Asteraceae species. Many of these species are most abundant the first growing season after fire and correspond to Henslow's Sparrow's preference for recently-burned grasslands. While some species like *M. expansa*, *C. aromaticum*, and *Rhynchospora* spp. may provide preferred vegetation structure and contribute seeds to the food supply, seeds of *Andropogon* spp. and *Schizachyrium* spp. are not preferred foods of Henslow's Sparrows. Presumably, these species are important because of their contribution to vegetation structure (Fuller 2004, Johnson 2006, DiMiceli et al. 2007). There are contradictory results on the importance of seed resources in determining Henslow's Sparrow abundance, but evidence suggests that Henslow's Sparrows are seed generalists and that this allows them to occupy such an array of grassland types (Tucker and Robinson 2003, Fuller 2004, Bechtoldt and Stouffer 2005, Johnson 2006, DiMiceli et al. 2007).

Many of the discrepancies in the results from the various studies of Henslow's Sparrow winter habitat preferences can be attributed to the variety of small-scale habitats and locations covered by these studies, including variation in management practices. Because of the isolation and patchiness of winter Henslow's Sparrow habitats, a small-scale focus is appropriate because as the spatial scale of a study increases, it becomes increasingly difficult to discern specific patterns within a single habitat type (Wiens 1989). It is also important to repeat studies across a larger area—that is, to metareplicate—to determine which trends are region specific and which hold true among regions and habitat types (Wiens 1981, Johnson 2002). Indeed, caution must be

taken when applying results from one region to management strategies in other regions (Johnson 2002).

Another grassland bird, the Sedge Wren (*Cistothorus platensis*), occurs from Canada to Argentina. The North American subspecies (*C. p. stellaris*) winters in the southeastern U.S. from coastal North Carolina to eastern Texas and south through Florida and Mexico (Fig. 2.2; Herkert et al. 2001). Between 1966 and 2007, the North American population increased by 1.53% annually, with the population remaining stable over the last 20 years (Sauer et al. 2008) due largely to the success of the Conservation Reserve Program (Johnson and Igl 1995). Because of the Sedge Wren's nomadic tendencies on the breeding grounds, however, breeding bird survey results should be viewed with caution (Bedell 1996). Where Sedge Wrens have exhibited regional decline (*e.g.*, the Northeast and eastern Great Lakes regions), it is mainly due to habitat loss (Peterjohn and Sauer 1999). On the breeding grounds in the U.S. Midwest and Canada, Sedge Wrens occupy a wide variety of habitats. Found primarily in dry to mesic grasslands with tall, dense herbaceous vegetation and moderate forb cover, they also occupy short-grass prairies, fields planted with crops such as rice and hay, and lightly-grazed pastures (Dechant et al. 1999 and references therein). Structurally, nesting Sedge Wrens prefer a tall, dense herbaceous layer with some woody cover; their tolerance of residual vegetation or litter varies among studies (Niemi and Hanowski 1984, Sample 1989, Delisle and Savidge 1997). In North Dakota, Sedge Wren occupancy increased with proximity to wetlands and decreased with tree cover (Cunningham and Johnson 2006).

Studies on Sedge Wren winter ecology and winter habitat preferences in North America are few, and most descriptions of habitat use are anecdotal (Herkert et al. 2001). In Florida, Sedge Wrens were described using fresh and brackish sedge marshes, wet palmetto prairies, old



Figure 2.2. Sedge Wren range map (from Herkert et al. 2001).

fields with dense, matted herbaceous layers, and longleaf pine savannas dominated by *Rhynchospora chapmanii* (Chapman's beaksedge; Sprunt 1954, McNair 1998, Herkert et al. 2001). Imhof (1976) described Sedge Wrens in Alabama using grassy marsh edges, bogs, damp patches of *Andropogon virginicus* (broomsedge bluestem), and occasionally drier grasslands with herbaceous cover 60–90 cm high. Lowery (1974) mentions Sedge Wrens in Louisiana preferring grassy marshes in coastal areas and dry *Andropogon* fields inland. In a study in mesquite grasslands in Texas, Sedge Wrens were found in “tall and rank vegetation” similar to that described by Johnson and Igl (1995) on their breeding grounds in North Dakota (Reynolds and Krausman 1998). In the only study to date looking at Sedge Wren winter habitat preferences, Baldwin et al. (2007), working in Texas coastal prairies, concluded that Sedge Wrens were more abundant in prairies with two to three growing seasons since fire than in prairies with one growing season since fire. The best predictor of Sedge Wren abundance was a dense herbaceous layer; shrub density had no significant effect. Sedge Wrens were more common in stands with *Baccharis halimifolia* (eastern baccharis) and *Triadica sebifera* (Chinese

tallow) and in prairies dominated by *Schizachyrium scoparium* or *Spartina patens* (saltmeadow cordgrass) rather than mixed-species stands (Baldwin et al. 2007). In Louisiana and Mississippi, Sedge Wrens and Henslow's Sparrows are frequently found in the same winter habitats, although Sedge Wrens appear to occupy a broader range of habitats (pers. obs.).

I sampled the wintering grassland bird communities, habitat structure, and plant species composition over two winters in De Soto National Forest (DSNF), De Soto Ranger District, in southern Mississippi. The objective of my study was to determine what vegetation structural features and plant species best predict stand occupancy by Sedge Wrens and Henslow's Sparrows in grassland patches. Because predictive variables vary among ecoregions and habitat types, it is important to assess the influence of specific vegetation variables on these birds on a small-scale, regional basis. The results from my study are intended to help forest managers in DSNF identify habitat patches most suitable for Henslow's Sparrows and Sedge Wrens and set goals for current and future habitat-restoration projects.

METHODS

Study Site

The study sites are described in Chapter 1.

Grassland Bird Sampling

The grassland bird sampling is described in Chapter 1.

Vegetation Structure and Plant Species Composition Sampling

The vegetation sampling is described in Chapter 1.

The identification of *Scleria muhlenbergii* was verified with the assistance of Diane M. Ferguson, the collections manager at the Louisiana State University herbarium. We used several plant keys, along with herbarium specimens, to identify this uncommon morphotype of *S. muhlenbergii*.

Statistical Analyses

For the following analyses, the individual study site was the sample unit ($n = 27$). To maintain independence among sites, I did not use the second-year samples from the five bog sites sampled both years. Bird species were analyzed by presence–absence at a site; vegetation-sampling measurements were averaged over each study site. I conducted two principal components analyses (PCAs) using PROC FACTOR in SAS 9.2 (SAS Institute Inc. 2006) to reduce the number of correlated vegetation structure and plant species composition guild variables to fewer, uncorrelated principal components (PCs). I performed a PCA for the 12 habitat variables and 15 plant guilds separately because of limited degrees of freedom. Because I used a different subset of study sites in these analyses, I could not use the PCAs from Chapter 1 ($n = 22$). I used a Varimax rotation to aid in the interpretation of the PCs and retained all PCs with Eigenvalues > 1 (Guttman 1954).

I used logistic regression to model the probability of Henslow's Sparrow and Sedge Wren site occupancy (PROC GLIMMIX, SAS Institute Inc. 2006). I used the vegetation structure and plant species composition PCs as the independent predictor variables in an information–theoretic model selection approach (Burnham and Anderson 2002). The third plant species PC was highly correlated with the woody structure PC, so I did not use the third plant PC when constructing candidate models. I did not include area sampled in candidate models, because although Henslow's Sparrow density decreased with area sampled, I do not believe that this was a biological phenomenon. I estimated an overdispersion factor ($\hat{c} = \text{Pearson } \chi^2/\text{df}$) by specifying residual as a random variable for global models to determine the goodness of fit and the appropriate probability distribution (Burnham and Anderson 2002). Henslow's Sparrow and Sedge Wren data were best modeled using the binomial distribution ($\hat{c} = 1.40$ and 1.13 , respectively). Models were ranked using Akaike's Information second-order Criterion (AIC_c)

for small sample size (Burnham and Anderson 2002). I also calculated an r^2_{GIC} (r^2 General Information Criterion) for each model (Wright 2001). This is a pseudo r^2 calculated from any one of the common information criteria used for model selection and is useful for reporting model results because it represents the relative proportional variance explained by a model. The formula is as follows:

$$r^2_{\text{GIC}} \text{ for model } x = 1 - \exp[(\text{GIC}_{\text{model } x} - \text{GIC}_{\text{null model}})/n],$$

where, for my data and modeling approach, the GIC is the AIC_c value.

The global models for both species were as follows: logit (bird occupancy) = herbaceous structure PC + woody structure PC + tree density PC + plant PC 1 + plant PC 3 + plant PC 4 + plant PC 5 (see results for interpretation of the PCs). I choose the variables in my candidate models based on parsimony, ease of interpretation, and my own biological knowledge of grassland birds and the longleaf pine ecosystem. For both Henslow's Sparrow and Sedge Wren, I created 21–29 candidate models per species (Appendices 6 and 7). I found the best models by first fitting combinations of the vegetation and plant species composition PCs. After determining the best models using these components, I constructed models with the raw variables that constituted the PCs to determine if any of these variables provided a better fit (Johnson 1998). If a plant guild variable provided a better fit, I inspected the plant species that constituted the guild to determine if any of those plant species made biological sense to include in the model, instead of the guild, based on my own observations in the field and those of previous studies. I modeled all raw variables from PCs separately because of potential problems with multicollinearity. This approach provided an effective way to test models with many, correlated variables. If raw variables provided an equal or better fit than the PCs, I retained the model with the raw variables because these are more parsimonious: a simple, measurable vegetation variable is more useful

than ordination scores, which are harder to interpret and require more statistical knowledge. I included all models with $\Delta \text{AICc} < 2$ in the final confidence set (Burnham and Anderson 2002).

RESULTS

Principal Components Analyses

Vegetation Structure.—See Chapter 1 for a description of the raw vegetation and bird data used in the following analyses. Principal components analysis of the vegetation structure variables resulted in three PCs with Eigenvalues > 1 , representing 82% of the variance (Table 2.1). The first PC represented woody understorey structure and was positively correlated with all woody structure variables (except trees), herbaceous height, and the herbaceous cover density CV, and was negatively correlated with the woody cover CV. The second PC represented herbaceous structure and was positively correlated with herbaceous cover and structure and negatively correlated with the herbaceous density CV. The third PC represented tree density and was most strongly correlated with canopy closure and tree basal area.

Plant Species Composition Guilds.—Principal components analysis of the plant species composition guilds resulted in five PCs with Eigenvalues > 1 , representing 70% of the variance (Table 2.2). The first PC was positively correlated with FACU and FAC graminoids and FACU woody plants and negatively correlated with UPL and FACW graminoids. The second PC was positively correlated with OBL woody plants and FACW forbs. The third PC was positively correlated with FAC and FACW woody plants and negatively correlated with OBL forbs. The fourth PC was positively correlated with UPL and FACU forbs and UPL woody plants. The fifth PC was positively correlated with FAC forbs and OBL graminoids.

Modeling Henslow's Sparrow and Sedge Wren Occupancy

Henslow's Sparrow Models.—Henslow's Sparrow site occupancy was best predicted by the herbaceous density CV and cover of the sedge *Scleria muhlenbergii* Steud. (Muhlenberg's

nut-rush, pitted nut-rush, SCMU8). Increasing herbaceous density CV reduced the probability of Henslow's Sparrow occupancy (Fig. 2.3). Increasing cover of *S. muhlenbergii* increased the probability of occupancy (Fig. 2.4). The null model, in comparison with the best model, had a $\Delta AICc$ of 13.50 (Appendix 5). The best model was the only model with $\Delta AICc < 2$, whereas the next best model had a $\Delta AICc$ of 5.99. The r^2_{GIC} for the best model was 0.39. The parameter estimates for both variables had confidence intervals that contained zero, meaning that although there is an effect, the degree of the effect cannot be quantified (MacKenzie et al. 2006). The parameter estimate and corresponding odds ratio for *S. muhlenbergii* cover is very large (Table 2.3). The size of the parameter estimate is probably due to parameter estimate inflation that sometimes occurs when using logistic regression on a small sample (Nemes et al. 2009).

Table 2.1. Rotated PC patterns from a PCA on 12 vegetation structure variables measured in DSNF, MS. Values are the correlations of the raw variables with each PC. The three PCs explain a cumulative 82% of the total variance. Highest correlations are in bold.

Variable	Wood PC	Herbaceous PC	Trees PC
Woody density	0.902	-0.166	0.037
Woody cover	0.884	-0.155	0.383
Number of stems	0.819	-0.313	0.055
Woody height	0.815	-0.319	0.256
Herb cover CV	0.693	-0.553	0.379
Herb height	0.580	-0.236	-0.025
Woody cover CV	-0.693	0.468	-0.158
Herb density	-0.163	0.943	-0.096
Herb cover	-0.480	0.773	-0.166
Herb density CV	0.529	-0.755	0.151
Canopy closure	0.122	-0.031	0.922
Tree basal area	0.118	-0.230	0.888
Proportion s ² explained	60%	13%	09%

Table 2.2. Rotated PC patterns from a PCA analysis on 15 plant species composition guilds measured in DSNF, MS. Guilds were created by combinations of each species' life form and Wetland Indicator Status. Values are the correlations of the guilds with each PC. The five PCs explain a cumulative 70% of the total variance. Highest correlations are in bold.

Variable	PC 1	PC 2	PC 3	PC 4	PC 5
Gram FAC	0.813	-0.267	-0.048	-0.049	0.028
Gram FACU	0.804	-0.323	0.095	-0.152	-0.114
Woody FACU	0.549	-0.090	0.152	0.204	-0.347
Gram UPL	-0.746	-0.266	0.174	-0.200	-0.117
Gram FACW	-0.751	0.133	-0.311	-0.222	0.000
Woody OBL	-0.129	0.927	0.068	-0.042	-0.053
Forb FACW	-0.155	0.871	-0.117	-0.033	0.022
Woody FACW	-0.061	-0.139	0.862	-0.058	0.240
Woody FAC	0.073	0.139	0.744	0.186	-0.353
Forb OBL	-0.582	0.052	-0.640	-0.034	-0.021
Forb UPL	-0.062	0.084	0.055	0.808	-0.013
Woody UPL	0.169	-0.097	-0.041	0.684	-0.293
Forb FACU	0.175	-0.149	0.140	0.491	0.353
Forb FAC	0.024	-0.071	0.069	-0.050	0.850
Gram OBL	-0.235	0.352	-0.371	-0.082	0.512
Proportion s ² explained	28%	12%	11%	10%	09%

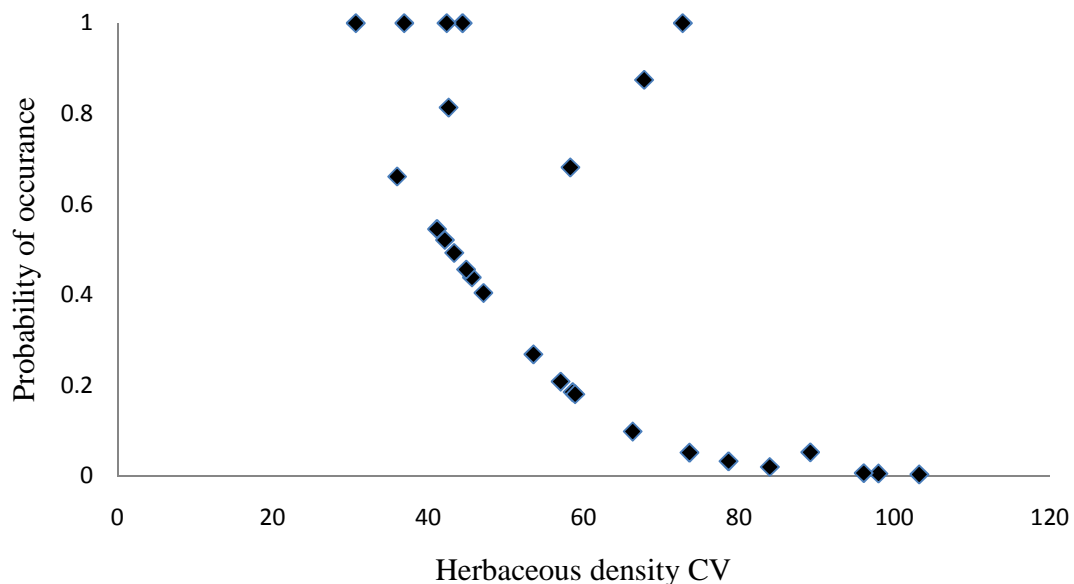


Figure 2.3. Scatter plot of herbaceous density CV (representing increasing heterogeneous herbaceous density) plotted on predicted probability of occupancy from the best model logistic model of Henslow's Sparrow site occupancy.

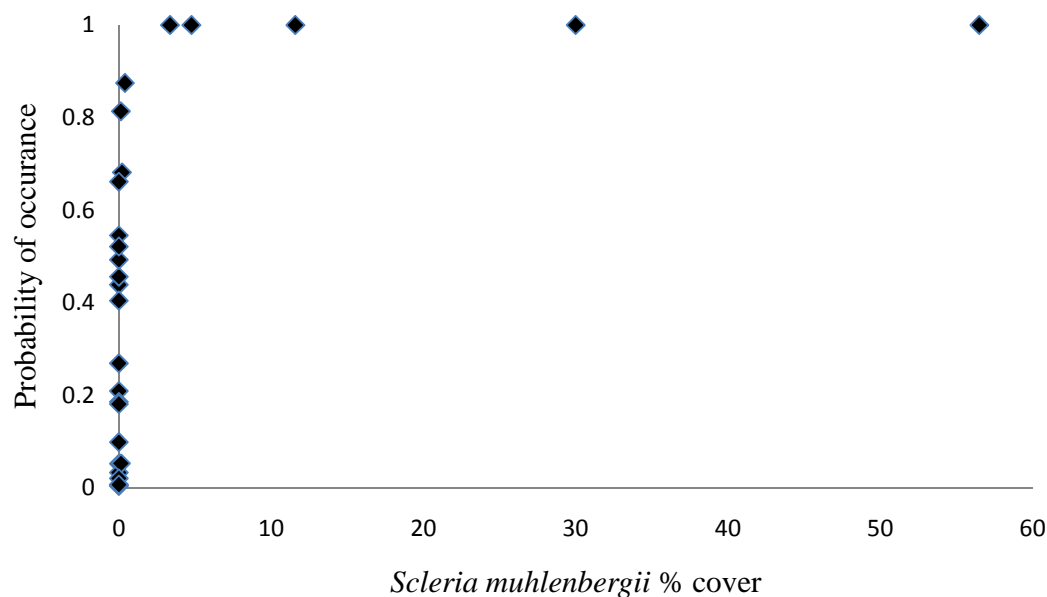


Figure 2.4. Scatter plot of *Scleria muhlenbergii* percent cover plotted on predicted probability of occupancy from the best logistic model of Henslow's Sparrow site occupancy.

Table 2.3. Parameter estimates for the best model predicting Henslow's Sparrow occupancy from vegetation structure and plant species composition in DSNF, MS. Parameter estimates are the log of the odds ratios. The r^2_{GIC} was 0.39. SCMU8 is cover of the sedge *Scleria muhlenbergii*.

Effect	Estimate	SE	95% CI		Odds ratio	<i>t</i>	df	<i>P</i>
Intercept	4.10	2.69	-1.46	9.65		1.52	24	0.1411
Herb density CV	-0.10	0.05	-0.21	0.02	0.91	-1.78	24	0.0884
SCMU8	10.75	6.67	-3.00	24.51	46,788.57	1.61	24	0.1198

Sedge Wren Models.—Sedge Wren occupancy in DSNF was best predicted by decreasing tree basal area and increasing woody understorey vegetation structure. The null model compared with the best model had a ΔAICc of 4.99 (Appendix 6). The best model contained the variable tree basal area and the woody structure PC. No other models had a $\Delta \text{AICc} < 2$, although four other models had a ΔAICc between 2.0 and 3.0. The r^2_{GIC} for the best model was 0.17; hence, the best model represented only a small improvement over the null model. Basal area had a slope with a confidence interval that did not overlap zero, but the

confidence interval for the woody structure PC did overlap zero, meaning that although there is a positive effect, this effect is not directly quantifiable (Table 2.5; MacKenzie et al. 2006).

Table 2.4. Parameter estimates for the best model predicting Sedge Wren occupancy from vegetation structure and plant species composition in DSNF, MS. Parameter estimates are the log of the odds ratios. The r^2_{GIC} was 0.17.

Effect	Estimate	SE	95% CI		Odds ratio	<i>t</i>	df	<i>P</i>
Intercept	1.07	1.04	-1.08	3.23		1.03	24	0.3134
Woody PC	1.22	0.63	-0.08	2.51	3.38	1.94	24	0.0641
Basal area	-6.99	3.25	-13.71	-0.28	0.00	-2.15	24	0.0420

DISCUSSION

Henslow's Sparrow.—Many studies have shown the importance of dense herbaceous structure for Henslow's Sparrows. Because herbaceous structure is maintained by fire, Henslow's Sparrows densities are typically higher in recently-burned pine savannas and decrease with time since fire; thus, birds are more likely to occupy places that have been burned the previous year. In DSNF, because of the many degraded habitats, a history of frequent fire does not mean a stand will be suitable for Henslow's Sparrows—many stands that were recently burned were not occupied by birds because of the sparse herbaceous cover. In degraded savannas suffering from shrub encroachment, certain shrub species, particularly *Ilex* spp., resprout quickly following fire and begin competing with herbaceous species, creating a sparse, heterogeneous herbaceous layer. In many upland stands in DSNF, herbaceous “layer” is misleading because the vegetation never forms a layer, occurring in isolated patches separated by bare ground and shrub clumps (Fig. 2.5). While small, isolated herbaceous patches may consist of dense clumps of grasses and forbs, at the stand level, the herbaceous vegetation is too patchy

and cannot support Henslow's Sparrows. Thus, the CV for herbaceous density is a better predictor of sparrow occupancy than herbaceous density.

Patchy herbaceous structure can have negative effects on Henslow's Sparrows in several ways. First, with reduced overall herbaceous cover, one can infer that there may be less seed resources to support wintering birds. Plant diversity is lower in upland habitats, which may lead to fewer seed choices and less food availability. Exposure to predation is another reason birds might avoid patchy vegetation. Henslow's Sparrows spend the majority of their time on the ground, and they often move through corridors in the vegetation (Thatcher et al. 2006). Thatcher et al. (2006) showed that Henslow's Sparrow mortality was higher in savannas that had not been recently burned. They speculated that as ground litter increased with time since fire, it filled in the herbaceous corridors, causing birds to walk on top of the vegetation where they were more exposed to avian predators, a main source of mortality. Rotenberry and Wiens (1980) found that abundances of tallgrass prairie birds sampled in Kansas and Oklahoma, included Henslow's Sparrow and Grasshopper Sparrow (*Ammodramus savannarum*), were negatively correlated with ground cover heterogeneity, but they did not address reasons for species-specific responses.

Low herbaceous cover caused by burrowing crawfish may be a reason three bog stands were not occupied by Henslow's Sparrows during at least one season of sampling. Two of these were the largest, most pristine bogs in the forest, with a history of frequent growing- and dormant season fires. One of these two bogs had experienced two growing seasons since fire and had formed a thick herbaceous layer that may not have appealed to Henslow's Sparrows. The other of these two bogs did not have Henslow's Sparrows the first season of sampling but had one bird the next season. This bog and the third had low herbaceous cover and considerable exposed bare ground because of high densities of crawfish chimneys and mounds created by burrowing crawfish in the genus *Fallicambarus*. Crawfish are common in DSNF bogs where

they alter plant species composition and structure by burrowing, clipping roots, burying seeds, and mixing soil (Brewer 1999a, b, Welch et al. 2008). I did not collect data on the number of crawfish chimneys and mounds in my study sites, but it appears that these structures are negatively correlated with herbaceous cover. Bogs with high densities of crawfish structures tend to lack the dense, continuous herbaceous cover required by Henslow's Sparrows, and birds in these places would be more exposed to predators because of the exposed bare ground (Fig. 2.6). Most of the *Fallicambarus* crawfish that depend on bogs are sensitive species with limited ranges. One species, *F. gordonii* (Camp Shelby burrowing crayfish) is endemic to Perry County in DSNF and is a candidate for listing under the Endangered Species Act (Johnston and Figiel 1997). Management for Henslow's Sparrows by controlling crawfish is not an option.

Cover of the sedge *S. muhlenbergii* increases the probability of Henslow's Sparrow occurrence. The USDA lists *S. muhlenbergii* as a facultative wetland plant, but in DSNF it occurs primarily in bogs, where it can be quite abundant (*S. muhlenbergii* is called *S. reticularis* var. *pubescens* Britton or *S. setacea* Poir. by some authors). *Scleria muhlenbergii* never occurred in RCW stands and occurred in only one upland stand. This annual sedge, or "nut-rush," has many long (20–90 cm), weak, reclining stems, 1–5 mm wide, that form loose, tangled mats in the vegetation (Fig. 2.7) and produces small, 1–3-mm achenes (Godfrey and Wooten 1979, Flora of North America Committee 2003). This plant occurs throughout the southeastern U.S. and in some parts of the Northeast and Midwest. The morphospecies occurring in DSNF is unusual in that the achene surface is smooth and glabrous compared to the more-common morphospecies with pitted, pubescent achenes (Flora of North America Committee 2003). All bogs that had high densities of Henslow's Sparrows also had high *S. muhlenbergii* cover. I suspect that the tangled mat of stems sprawling prostrate through the vegetation creates an herbaceous layer that is structurally ideal for Henslow's Sparrows foraging and cover. The growth structure of this

plant, combined with its color in winter, may also create a ground pattern suitable for Henslow's Sparrow's camouflage plumage.

Evidence suggests that *Scleria* spp. may be a preferred food of Henslow's Sparrows. Fuller (2004), working in coastal flatwoods bogs at the Mississippi Sandhill Crane NWR found that *Scleria* spp. achenes were the second most abundant item in Henslow's Sparrow crop contents and ranked second in crop composition by mass, with *Rhynchospora* spp. achenes ranking first in both mass and abundance. She mentioned that *Scleria* spp. was common on her sample plots on the refuge, but that seed heads often lacked achenes, which is what I also observed in DSNF. *Scleria* is an annual and senesces in autumn (W.J. Platt, LSU Life Sciences Department, pers. comm.), so more of these seeds are available to ground-foraging birds early in winter compared to seeds of species that remain on the stalk later into winter. Further evidence of *Scleria* preference by Henslow's Sparrows was found by DiMiceli (2006) who, working in southeastern Louisiana flatwoods bogs, found that *Scleria* spp. were the most frequent seeds in Henslow's Sparrow fecal samples, occurring in 84% of the samples, followed by *Rhynchospora* spp. Although *Scleria* spp. frequency varied among their sampling months, the frequency of occurrence never dropped below 70%, even in March and early April, suggesting that although these seeds senesce in early winter, abundance does not necessarily decline significantly before the end of winter. If Henslow's Sparrows use habitat cues to choose their wintering grounds when they arrive in the fall, then the presence or abundance of *S. muhlenbergii* could be an important cue in identifying high-quality habitat. Thus, if forest managers in DSNF wish to determine areas potentially important for wintering Henslow's Sparrows for restoration or preservation efforts, they should target bogs with high *S. muhlenbergii* cover. *Scleria muhlenbergii* is most abundant the first growing season after fire and decreases substantially without recent (< 1 growing season) fire (W.J. Platt, pers. comm.). Plant species composition

and vegetation structure, in general, change with time since fire; hence, a bog that does not meet the proper criteria for Henslow's Sparrows one year may meet them later following fire.

Sedge Wren.—Sedge Wren site occupancy was best predicted by decreasing tree basal area (the raw variable) and increasing woody understorey vegetation (woody vegetation structure PC). The parameter estimates for the woody understorey vegetation PC and basal area had wide confidence intervals that overlapped zero, so I cannot quantify how much effect they had on the probability of Sedge Wren occurrence. The model explained very little of the variation in Sedge Wren occurrence, so there are undoubtedly other variables affecting occupancy. The many habitat types used by Sedge Wrens across the southeast show that they are habitat generalist on the wintering grounds (Sprunt 1954, Lowery 1974, Imhof 1976, McNair 1998, Hamel 2003, Baldwin et al. 2007), so it is not surprising that the best model explained so little variation. Still, the best model was an improvement over the null model, suggesting that the selected variables might be useful in predicting Sedge Wren occurrence. Preferences for woody vegetation and low tree densities have also been observed on the breeding grounds (Niemi and Hanowski 1984, Sample 1989, Cunningham and Johnson 2006).

My finding that Sedge Wrens prefer woody understorey vegetation contrasts with that of Baldwin et al. (2007), the only other study of Sedge Wren winter habitat preferences. They found that Sedge Wrens were more common in sites with 2–3 growing seasons since fire. Site occupancy and abundance were not associated with shrub densities, but with dense herbaceous vegetation like the kind occurring two to three years after fire. Their study, however, was conducted in Texas coastal prairies, an ecosystem much different from pine savannas. Reynolds and Krausman (1998) also observed that Sedge Wrens frequented tall, dense herbaceous vegetation on the wintering grounds. A bird with such broad habitat preferences may respond differently in different habitats, which is why multiple studies across different regions are



Figure 2.5. Photograph of the patchy herbaceous ground cover typical of upland longleaf-slash pine stands in DSNF, MS. This picture was taken in April 2009 after one full growing season since fire.



Figure 2.6. Photograph of a bog with high crawfish disturbance. This picture was taken in October 2009 at the site DAD after one growing season since fire. This site never had any Henslow's Sparrows.



Figure 2.7. Photograph of herbaceous ground cover dominated by *Scleria muhlenbergii* (the wiry brown graminoid) typical in some bogs in DSNF after one growing season since fire. This picture was taken in April 2009 at site ZAK, which had the highest density of Henslow's Sparrows.

important. Also, it is difficult to compare woody structure between my study and that of Baldwin et al (2007). They used a point-centered quarter method, while I not only used different measurements, but I combined them using PCA. It is also possible that high woody density in the coastal prairies could correspond to low woody density in DSNF. Another explanation would be that Sedge Wrens in DSNF are not responding directly to woody understorey vegetation, but to the dense herbaceous vegetation that accumulates along with woody understorey vegetation as time since fire increases. Most of the sites that were occupied, however, had experienced only one growing season since fire, and upland stands rarely accumulate thick herbaceous vegetation.

If Sedge Wrens prefer dense herbaceous vegetation, then high tree basal area could have an indirect negative effects on Sedge Wrens. Numerous studies have shown that high tree

density in pine savannas alters the herbaceous vegetation because of shading and belowground competition. Understorey biomass tends to be greater in gaps (McGuire et al. 2001), and species richness, along with density and abundance of herbaceous species, is also higher in gaps and away from trees (Brewer 1998, Harrington and Edwards 1999, Platt et al. 2006). Light availability leads to higher species richness and herbaceous density in canopy gaps, but belowground competition for resources may also be important (Brewer 1998, Harrington et al. 2003). In my study, herbaceous cover was negatively correlated with basal area, but herbaceous density was not significantly correlated with basal area. Sedge Wren preference for woody understorey vegetation and low tree basal area are contradictory because high tree density increases the establishment of woody understorey vegetation (Brewer 1998, Hinman et al. 2008), so most areas with high tree densities also have high shrub densities. This presents the possibility that retention of woody understorey vegetation in the best model could be a spurious result. Although Sedge Wrens may be tolerant of woody understorey vegetation as is reflected by their preference for habitats that have > 1 growing season since fire, they may not necessarily prefer habitats with woodier understorey vegetation. At some point, woody vegetation will inhibit Sedge Wrens as it competes with and reduces herbaceous plants. More research on Sedge Wren winter habitat preferences, particularly in longleaf pine savannas, is needed.

Conclusions

My result that herbaceous density influences Henslow's Sparrow occupancy is consistent with previous studies conducted across multiple ecoregions and habitat types. This study, however, is the first to stress the importance of continuous, homogeneous herbaceous density. Many areas, particularly longleaf pine savannas, that might appear suitable for Henslow's Sparrows are in fact unsuitable because of the patchy distribution of herbaceous vegetation. Restoring the herbaceous component of longleaf pine savannas to a continuous herbaceous layer

should be one of the principal goals for forest managers interested in managing for grassland birds. My results also expand on other studies suggesting a dietary relationship between Henslow's Sparrows and *Scleria*. These results reflect the importance of metareplication because as plant species composition changes across regions, the importance of specific species will change. In DSNF, where *S. muhlenbergii* is common in bogs, it appears this sedge may be important in predicting Henslow's Sparrow occupancy. More research looking at how it affects bird abundance would be valuable.

Many of the management practices currently used in DSNF benefit grassland birds by improving their habitat. Prescribed fire, bog restoration, and the removal of woody understorey vegetation for management of RCWs all benefit grassland birds. Furthermore, the reduction of trees that occurred from Hurricane Katrina may indirectly improve the habitat by promoting the development of a healthy herbaceous layer. Further thinning of dense pine stands would also assist in the restoration of the natural herbaceous layer. The majority of prescribed fires in DSNF are dormant season fires. Although these fires initially reduce woody understorey vegetation and improve the herbaceous layer, they may have the long-term effect of increasing woody understorey vegetation density. Nonetheless, prescribed fire during any season is preferable to no fire because without it afforestation would be inevitable. Frequent dormant season fires help maintain a continually shifting mosaic of temporary habitats suitable for grassland birds, depending on the forest type and bird species. The amount of growing season fires used yearly in DSNF has increased over the last decade, but logistical constraints such as an available work force, weather, and private property make the sole use of growing season fires unrealistic (K. Coursey, pers. comm.). Focusing on the restoration of the many bogs occurring in the forest is perhaps the best strategy for maintaining pockets of high-quality habitat suitable for Henslow's Sparrows and Sedge Wrens, while increasing the size and number of RCW clusters

will also increase the area of potential habitat for all grassland bird species, including Bachman's Sparrows. Awareness of the importance of these rare bog habitats for grassland birds is crucial for the holistic management of the forest. As forest-management challenges become more complex with increasing urbanization and the inevitability of future hurricanes, protecting these sensitive areas from the impacts of development and timber salvaging will benefit local wintering grassland bird populations.

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**APPENDIX 1. PLANT SPECIES GUILD ASSIGNMENTS BASED ON LIFE FORM AND
WETLAND INDICATOR STATUS.**

Graminoids	WIS	Woody	WIS
<i>Andropogon gerardii</i>	FAC	<i>Ilex vomitoria</i>	FAC
<i>Andropogon virginicus</i>	FAC	<i>Morella cerifera</i>	FAC
<i>Chasmanthium sesseliflorum</i>	FAC	<i>Nyssa sylvatica</i>	FAC
<i>Panicum anceps</i>	FAC	<i>Quercus nigra</i>	FAC
<i>Panicum virgatum</i>	FAC	<i>Smilax glauca</i>	FAC
<i>Schizachyrium tenerum</i>	FAC	<i>Symplocos tinctoria</i>	FAC
<i>Dichanthelium</i> spp.	FAC	<i>Vaccinium elliotii</i>	FAC
<i>Paspalum</i> sp.	FAC	<i>Gelsemium sempervirens</i>	FAC
<i>Sporobolus junceus</i>	FACU	<i>Rubus</i> sp.	FAC
<i>Andropogon ternarius</i>	FACU	<i>Cornus florida</i>	FACU
<i>Aristida longespica/oligantha</i>	FACU	<i>Pinus palustris</i>	FACU
<i>Gymnopogon brevifolius</i>	FACU	<i>Prunus serotina</i>	FACU
<i>Schizachyrium scoparium</i>	FACU	<i>Vaccinium arboreum</i>	FACU
<i>Sorghastrum secundum</i>	FACU	<i>Vaccinium stamineum</i>	FACU
<i>Andropogon glaucopsis</i>	FACW	<i>Vaccinium darrowi</i>	FACU
<i>Andropogon glomeratus</i>	FACW	<i>Aronia arbutifolia</i>	FACW
<i>Arundinaria gigantea</i>	FACW	<i>Cyrilla racemiflora</i>	FACW
<i>Chasmanthium laxum</i>	FACW	<i>Ilex coriacea</i>	FACW
<i>Ctenium aromaticum</i>	FACW	<i>Ilex glabra</i>	FACW
<i>Dichromena latifolia</i>	FACW	<i>Lyonia lucida</i>	FACW
<i>Eleocharis tuberculosa</i>	FACW	<i>Magnolia virginiana</i>	FACW
<i>Eragrostis refracta</i>	FACW	<i>Myrica heterophylla</i>	FACW
<i>Erianthus giganteus</i>	FACW	<i>Pinus elliotii</i>	FACW
<i>Juncus</i> sp.	FACW	<i>Smilax laurifolia</i>	FACW
<i>Muhlenbergia expansa</i>	FACW	<i>Viburnum nudum</i>	FACW
<i>Panicum verrucosum</i>	FACW	<i>Hypericum</i> spp.	FACW
<i>Rhynchospora</i> spp.	FACW	<i>Gaylussacia mosieri</i>	FACW
<i>Scleria</i> spp.	FACW	<i>Cliftonia monophylla</i>	OBL
<i>Scleria baldwinii</i>	FACW	<i>Nyssa biflora</i>	OBL
<i>Tridens ambiguus</i>	FACW	<i>Persea palustris</i>	OBL
<i>Carex</i> spp.	FACW	<i>Acer rubra</i> var. <i>drummondii</i>	OBL
<i>Aristida palustris</i>	OBL	<i>Licania michauxii</i>	UPL
<i>Andropogon gyrans</i> var. <i>gyrans</i>	OBL	<i>Smilax pumila</i>	UPL
<i>Andropogon gyrans</i> var. <i>stenophyllus</i>	OBL		
<i>Andropogon mohrii</i>	OBL		
<i>Carex glaucescens</i>	OBL		
<i>Dichanthelium scabriusculum</i>	OBL		
<i>Fuirena</i> sp.	OBL		
<i>Rhynchospora chapmanii</i>	OBL		
<i>Rhynchospora</i> spp. "filamentous"	OBL		
<i>Anthraenantia villosa</i>	UPL		

Appendix 1 continued.

<i>Forbs</i>	
<i>Helianthus angustifolius</i>	FAC
<i>Aster</i> spp.	FAC
<i>Cirsium</i> sp.	FAC
<i>Elephantopus</i> sp.	FAC
<i>Euthamia</i> spp.	FAC
<i>Solidago</i> spp.	FAC
<i>Diodia teres</i>	FACU
<i>Mitchella repens</i>	FACU
<i>Pteridium aquilinum</i>	FACU
<i>Balduina uniflora</i>	FACW
<i>Bigelovia nudata</i>	FACW
<i>Chaptalia tomentosa</i>	FACW
<i>Eryngium integrifolium</i>	FACW
<i>Eupatorium</i> spp.	FACW
<i>Iris</i> sp.	FACW
<i>Zigadenus densus</i>	FACW
<i>Viola primulifolia</i>	FACW
<i>Bidens</i> sp.	FACW
<i>Osmunda</i> sp.	FACW
<i>Eriocaulon</i> spp.	OBL
<i>Lycopodiella alopecuroides</i>	OBL
<i>Sarracenia alata</i>	OBL
<i>Sarracenia psittacina</i>	OBL
<i>Woodwardia areolata</i>	OBL
<i>Xyris</i> spp.	OBL
<i>Pityopsis graminifolia</i>	UPL

APPENDIX 2. ALL BIRD SPECIES, IN ORDER OF PERCENT OCCURRENCE IN 27 STUDY SITES, DETECTED < 50 M FROM TRANSECT AND ON TRANSECT FOR ALL SURVEYS OVER TWO WINTERS.

Bird Species	% occurrence	Bird Species	% occurrence
Pine Warbler	0.85	Common Yellowthroat	0.15
Swamp Sparrow	0.81	Red-cockaded Woodpecker	0.15
Chipping Sparrow	0.74	Red-shouldered Hawk	0.15
House Wren	0.74	<i>Ammodramus</i> sp.	0.11
Brown-headed Nuthatch	0.67	Blue-headed Vireo	0.11
American Robin	0.63	Mourning Dove	0.11
Sedge Wren	0.52	Ruby-crowned Kinglet	0.11
Carolina Wren	0.48	White-throated Sparrow	0.11
Henslow's Sparrow	0.48	American Kestrel	0.07
American Crow	0.44	Cedar Waxwing	0.07
Eastern Bluebird	0.41	Dark-eyed Junco	0.07
Bachman's Sparrow	0.37	Downy Woodpecker	0.07
Song Sparrow	0.37	Golden-crowned Kinglet	0.07
American Goldfinch	0.30	Northern Cardinal	0.07
Eastern Phoebe	0.30	Northern Mockingbird	0.07
Red-bellied Woodpecker	0.30	Red-winged Blackbird	0.07
Unknown bird	0.30	Turkey Vulture	0.07
Yellow-bellied Sapsucker	0.30	American Woodcock	0.04
Hairy Woodpecker	0.22	Black Vulture	0.04
Yellow-rumped Warbler	0.22	Carolina Chickadee	0.04
Eastern Towhee	0.19	Le Conte's Sparrow	0.04
Northern Flicker	0.19	Purple Martin	0.04
Pileated Woodpecker	0.19	Red-headed Woodpecker	0.04
Unknown sparrow	0.19	Red-tailed Hawk	0.04
<i>Ammodramus</i> sp./	0.15	Tufted Titmouse	0.04
Bachman's Sparrow			
Blue Jay	0.15	Vesper Sparrow	0.04

**APPENDIX 3. BIRD SPECIES BANDED DURING GRASSLAND BIRD SURVEYS
OVER TWO WINTERS IN MS.**

Band Number	Species	Age	Sex	Banding Date	Location
1990-20027	HESP	HY	U	12/08/2007	CAR
1990-20028	HESP	HY	U	12/08/2007	CAR
1990-20029	HESP	U	U	12/08/2007	CAR
1990-20030	HESP	U	U	12/08/2007	CAR
1990-20032	HESP	U	U	12/08/2007	CAR
1990-20033	HESP	U	U	12/08/2007	CAR
1990-20034	HESP	U	U	12/08/2007	BOG
1990-20035	HESP	HY	U	12/09/2007	BOO
1990-20036	HESP	U	U	12/16/2007	CAR
1990-20037	HESP	SY	U	01/05/2008	BOG
1990-20038	HESP	AHY	U	01/12/2008	BOO
1990-20039	HESP	AHY	U	01/12/2008	PAN
1990-20076	HESP	HY	U	11/29/2008	RCW28
1990-20077	HESP	HY	U	12/06/2008	CAR
1990-20078	HESP	HY	U	12/06/2008	CAR
1990-20079	HESP	HY	U	12/06/2008	NAN
1990-20080	HESP	HY	U	12/06/2008	GOB
1990-20081	HESP	U	U	12/07/2008	BOO
1990-20082	HESP	HY	U	12/07/2008	BOO
1990-20083	HESP	SY	U	01/03/2009	ZAK
1990-20084	HESP	SY	U	01/03/2009	GOB
1990-20085	HESP	AHY	U	01/03/2009	ZAK
1990-20086	HESP	AHY	U	01/03/2009	GOB
1990-20211	HESP	AHY	U	02/07/2009	CAR
1990-20212	HESP	SY	U	02/07/2009	BOG
1990-20026	SEWR	U	U	12/01/2008	SUM
1990-20031	SEWR	HY	U	12/08/2007	CAR
2530-08501	SEWR	HY	U	12/06/2008	DAN
2530-08502	SEWR	U	U	12/06/2008	DAN
2530-08505	SEWR	ASY	U	02/08/2009	DAN

APPENDIX 4. ALL PLANT SPECIES, IN ORDER OF PERCENT OCCURRENCE OUT OF 27 STUDY SITES, ENCOUNTERED DURING VEGETATION SURVEYS.

Species	% occurrence	Species	% occurrence
<i>Ilex glabra</i>	1.00	<i>Aster</i> spp.	0.15
<i>Dichanthelium</i> spp.	0.96	<i>Chaptalia tomentosa</i>	0.15
<i>Muhlenbergia expansa</i>	0.93	<i>Viburnum nudum</i>	0.15
<i>Schizachyrium scoparium</i>	0.89	<i>Andropogon gyrans</i> var. <i>stenophyllus</i>	0.11
<i>Ctenium aromaticum</i>	0.85	<i>Eleocharis tuberculosa</i>	0.11
<i>Panicum anceps</i>	0.81	<i>Sarracenia psittacina</i>	0.11
<i>Ilex coriacea</i>	0.81	<i>Scleria</i> spp.	0.11
<i>Aristida palustris</i>	0.70	<i>Solidago</i> spp.	0.11
<i>Schizachyrium tenerum</i>	0.70	<i>Sorghastrum secundum</i>	0.11
<i>Ilex vomitoria</i>	0.70	<i>Tridens ambiguus</i>	0.11
<i>Gaylussacia mosieri</i>	0.67	<i>Aronia arbutifolia</i>	0.11
<i>Andropogon virginicus</i>	0.63	<i>Nyssa biflora</i>	0.11
<i>Rhynchospora</i> spp.	0.59	<i>Persea palustris</i>	0.11
<i>Andropogon gyrans</i> var. <i>gyrans</i>	0.56	<i>Smilax pumila</i>	0.11
<i>Rhynchospora</i> spp. "filamentous"	0.52	<i>Symplocos tinctoria</i>	0.11
<i>Panicum verrucosum</i>	0.48	<i>Andropogon gerardii</i>	0.07
<i>Smilax laurifolia</i>	0.48	<i>Andropogon ternarius</i>	0.07
<i>Vaccinium darrowi</i>	0.48	<i>Carex glaucescens</i>	0.07
<i>Helianthus angustifolius</i>	0.44	<i>Chasmanthium laxum</i>	0.07
<i>Gelsemium sempervirens</i>	0.44	<i>Elephantopus</i> sp.	0.07
<i>Magnolia virginiana</i>	0.44	<i>Viola primulifolia</i>	0.07
<i>Vaccinium elliotii</i>	0.44	<i>Cliftonia monophylla</i>	0.07
<i>Sarracenia alata</i>	0.41	<i>Cyrilla racemiflora</i>	0.07
<i>Xyris</i> spp.	0.41	<i>Licania michauxii</i>	0.07
<i>Morella cerifera</i>	0.41	<i>Vaccinium stamineum</i>	0.07
<i>Rubus</i> sp.	0.41	<i>Andropogon glaucopsis</i>	0.04
<i>Aristida longespica/oligantha</i>	0.37	<i>Balduina uniflora</i>	0.04
<i>Eragrostis refracta</i>	0.37	<i>Bidens</i> sp.	0.04
<i>Eriocaulon</i> spp.	0.37	<i>Bigelowia nudata</i>	0.04
<i>Andropogon glomeratus</i>	0.33	<i>Chasmanthium sesseliflorum</i>	0.04
<i>Dichanthelium scabriusculum</i>	0.33	<i>Cirsium</i> sp.	0.04
<i>Paspalum</i> sp.	0.33	<i>Dichromena latifolia</i>	0.04
<i>Pityopsis graminifolia</i>	0.33	<i>Diodia teres</i>	0.04
<i>Scleria muhlenbergii</i>	0.33	<i>Erianthus giganteus</i>	0.04
<i>Quercus nigra</i>	0.33	<i>Eryngium integrifolium</i>	0.04
<i>Vaccinium arboreum</i>	0.33	<i>Fuirena</i> sp.	0.04

Appendix 4 continued.

<i>Andropogon mohrii</i>	0.30	<i>Gymnopogon brevifolius</i>	0.04
<i>Arundinaria gigantea</i>	0.30	<i>Iris</i> sp.	0.04
<i>Panicum virgatum</i>	0.30	<i>Juncus</i> sp.	0.04
<i>Pinus palustris</i>	0.30	<i>Mitchella repens</i>	0.04
<i>Smilax glauca</i>	0.30	<i>Osmunda</i> sp.	0.04
<i>Eupatorium</i> spp.	0.26	<i>Sporobolus junceus</i>	0.04
<i>Euthamia</i> spp.	0.22	<i>Woodwardia areolata</i>	0.04
<i>Rhynchospora chapmanii</i>	0.22	<i>Zigadenus densus</i>	0.04
<i>Hypericum</i> spp.	0.22	<i>Acer rubra</i> var. <i>drummondii</i>	0.04
<i>Anthaenanthia villosa</i>	0.19	<i>Cornus florida</i>	0.04
<i>Carex</i> spp.	0.19	<i>Lyonia lucida</i>	0.04
<i>Lycopodiella alopecuroides</i>	0.19	<i>Nyssa sylvatica</i>	0.04
<i>Pteridium aquilinum</i>	0.19	<i>Pinus elliotii</i>	0.04
<i>Myrica heterophylla</i>	0.19	<i>Prunus serotina</i>	0.04

APPENDIX 5. SET OF CANDIDATE MODELS USED TO MODEL HENSLOW'S SPARROW OCCUPANCY. PLANT1–PLANT5 ARE PCS OF THE PLANT SPECIES GUILDS. GRAM IS GRAMINOID, HERB IS HERBACEOUS STRUCTURE, WOOD IS WOODY UNDERSTOREY STRUCTURE, TREE IS TREE DENSITY, AND CV IS COEFFICIENT OF VARIATION.

Model	AIC _c	ΔAIC _c	w _i	r ² _{GIC}
Herb density CV + SCMU8	25.76	0.00	0.06	0.39
Herb density CV + Plant1 + Plant5	31.74	5.99	0.04	0.24
Herb density CV + Plant1 + Herb density CV*Plant1	31.85	6.09	0.04	0.24
Herb density CV + Plant1	32.44	6.68	0.04	0.22
Herb density CV + Plant5	32.60	6.84	0.04	0.22
Herb density CV	33.54	7.78	0.04	0.19
Herb PC + Plant4 + Herb PC*Plant4	33.81	8.05	0.04	0.18
Herb cover	34.49	8.73	0.04	0.16
Herb density CV + Plant1 + Plant4	35.17	9.41	0.04	0.14
Herb density CV + Gram OBL	35.61	9.85	0.04	0.13
Herb density CV + Gram FACW	35.79	10.03	0.04	0.12
Herb density CV + Plant4	36.00	10.24	0.04	0.11
Herb PC + Plant1 + Plant4 + Herb PC*Plant1 + Herb PC*Plant4	36.87	11.11	0.03	0.08
Herb PC	37.48	11.72	0.03	0.06
Wood PC + Herb PC	37.76	12.00	0.03	0.05
Herb PC + Plant1	37.89	12.13	0.03	0.05
Herb density	38.87	13.12	0.03	0.01
Null	39.26	13.50	0.03	0.00
Wood PC	39.73	13.97	0.03	-0.02
Herb PC + Plant4	39.89	14.14	0.03	-0.02
Herb PC + Tree PC	39.92	14.16	0.03	-0.02
Herb PC + Plant1 + Herb PC*Plant1	40.20	14.44	0.03	-0.04
Herb PC + Plant1 + Plant4	40.43	14.67	0.03	-0.04
Wood PC + Herb PC + Tree PC	40.46	14.70	0.03	-0.05
Wood PC + Plant1 + Plant2	40.61	14.85	0.03	-0.05
Tree PC	41.51	15.75	0.03	-0.09
Wood PC + Plant2	41.89	16.14	0.03	-0.10
Wood PC + Tree PC	42.19	16.43	0.03	-0.11
Global	46.52	20.76	0.02	-0.31

**APPENDIX 6. SET OF CANDIDATE MODELS USED TO MODEL SEDGE WREN
OCCUPANCY.**

Model	AIC _c	ΔAIC _c	w _i	r ² _{GIC}
Wood PC + Basal area	28.08	0.00	0.07	0.17
Basal area	30.49	2.41	0.06	0.09
Wood PC + Canopy closure	30.59	2.52	0.06	0.09
Herb PC + Tree PC	30.93	2.86	0.06	0.08
Canopy closure	31.00	2.93	0.06	0.07
Wood PC + Herb PC + Tree PC	31.24	3.17	0.06	0.07
Wood PC	32.73	4.65	0.05	0.01
Null	33.06	4.99	0.05	0.00
Plant5	33.56	5.48	0.05	-0.02
Herb PC + Plant4	34.59	6.51	0.05	-0.06
Plant2	35.10	7.03	0.05	-0.08
Wood PC + Herb PC	35.16	7.09	0.05	-0.08
Herb PC	35.30	7.22	0.05	-0.09
Plant1	35.39	7.31	0.05	-0.09
Wood PC + Plant4 + Wood PC*Plant4	35.95	7.87	0.04	-0.11
Plant1 + Plant5	36.09	8.01	0.04	-0.12
Plant2 + Plant4	36.77	8.69	0.04	-0.15
Plant1 + Plant4	37.04	8.97	0.04	-0.16
Plant1 + Plant2	37.63	9.55	0.04	-0.18
Plant1 + Plant2 + Plant4 + Plant5	40.69	12.61	0.04	-0.33
Global	46.86	18.78	0.03	-0.67

**APPENDIX 7. VOLUNTEERS, IN NO PARTICULAR ORDER, WHO HELPED
CONDUCT BIRD SURVEYS OVER TWO WINTERS IN MS AND LA.**

<u>Mississippi</u>	<u>Louisiana</u>
Peter Markos	Phred Benham
Cecilia Leumas	Ken Hackman & Madison High
Nick Winstead	Leslie Frank
Lou Proudfoot	Adam Walz
Jerry Litton & Molly	Jeremy White
Jeremy Nicholson	Beth Wiggins
Dave Fox	Gigi Savona
Megan	Gigi's friend & son
Hugo Gee	Jessie Deichmann
Laura Palasz	Jessie Brauch
Toni Taylor–Salisbury	Ariele Baker
Jen Anderson	Erik Johnson
Mike Byrne	Stefan Woltmann
Josh Schafer	Jacob Saucier
Jonathan Carpenter	Erin Herbez
Falyn Owens	Randy & Sue Steil
Carla	Prospective grad student
Jessica Mostacedo	Hanna Bieberly
Joe McGee	Luke Powel
Phil Stouffer	Karl Mokross
Becca Babbins	Aviane Aguiard
Isaac Knowles	Cathrine Norma
	Craig Luckner
	Eric Hoff
	Laura Palasz
	Toni Taylor–Salisbury
	Jonathan Carpenter
	Falyn Owens
	Dave Fox
	Jen Anderson

**APPENDIX 8. ALL AREAS EXPLORED, REFERENCED BY NEAREST ROAD OR
LANDMARK, FOR POTENTIAL STUDY SITES.**

322	423/Little Biloxi WMA
349	426 and side roads
354	434B
405	440/Big foot Rd.
420	Airey Tower Rd. 440-McHenry
425	Bachman's Sparrow sites
428	Beaver Pond Rd. and side roads
434	Blackwell farm Rd.
440	Blackwell Farm Rd.
441	Brooks' study sites and surrounding areas
447	Carnes Rd.
304D	Carrbridge Rd.
309 west to 353, south along 309K	CC Road
309A	Deep Creek Rd.
313B	East wire Rd.
317 n to 306 to 358C	Hwy 15
333/Red Creek Rd.	Larue Rd.
334A	Leaf River Head Quarters
35000 around Fort Shelby.	Martha Redmond Rd.
375/New Zion Rd.	Marvin Williams Rd.
406D	McHenry Rd.
406E	Old Biloxi Rd.
406G	Ramsey Rd.
409/Black Rd.	RCW clusters and surrounding areas
419B off Blackwell Farm Rd.	Scarborough Rd.
420C	Walker Rd. east
421 and side roads	West McHenry Rd. near Little Biloxi WMA

VITA

Matt Brooks grew up in High Point, North Carolina, and discovered nature at an early age as a Boy Scout. He received his Bachelor of Science degree in environmental science with a concentration in ecology and field biology from the University of North Carolina, Asheville, in 2002. While in school in Asheville, Matt volunteered at a botanical garden and worked at a plant nursery, as a camp counselor, a landscaper, a musician, an environmental educator, a lifeguard, and a pizza maker. After living in Asheville for five years, Matt became a wandering biological field technician, focusing on birds, botany, and occasionally environmental education. His work and wanderlust took him to Hawaii, Peru, Utah, for three seasons, Texas, Colorado, for two seasons, Ecuador, North Carolina, Louisiana, and many places in between. When not naturalizing or working on science, Matt can be found playing his guitar, banjo, or fiddle, and laughing with friends.