

HABITAT CHARACTERISTICS INFLUENCING RESIDENT AND OVER-WINTERING
GRASSLAND BIRDS ON THE DRY PRAIRIE OF SOUTH-CENTRAL FLORIDA

by

ADAM BROOKS BUTLER

(Under the Direction of John P. Carroll)

ABSTRACT

Grassland birds have declined for nearly 4 decades, likely because of continual fragmentation and degradation of grassland ecosystems. Relative to other systems, little attention has been given to these issues on the dry prairie of Florida. This ecosystem is home to several species of concern and also serves as over-wintering grounds for a number of the continent's short-distance migrants. I investigated the relationship between habitat and patch size on 2 breeding birds of the dry prairie: Bachman's sparrows (*Aimophila aestivalis*) and Eastern meadowlarks (*Sturnella magna*). I also evaluated burning regime and habitat effects on 2 migrants: grasshopper sparrows (*Ammodramus savannarum p.*) and sedge wrens (*Cistothorus platensis*). Response to patch size differed between the breeding species, yet each preferred habitat conditions that were a result of long-term frequent fire. The 2 wintering species responded most significantly to the short-term effects of fire, although their preference of burn intervals differed.

INDEX WORDS: Dry prairie, grassland birds, Bachman's sparrow, eastern meadowlark, grasshopper sparrow, sedge wren, habitat characteristics, patch size, winter migrants, south Florida, habitat use, prescribed fire

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I would like to dedicate this work to wood ducks whistling through ancient oaks at sunrise, the brilliance of dogwoods in April, cool creeks on hot summer days, the color of red maples in October, calls of cardinals on clear spring mornings, the hypnotizing sound of wind rustling through the pines, and so much more, in hopes that my meager efforts will help ensure that simple miracles such as those will always be enjoyed.

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-- ABB --

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CHAPTER 1

INTRODUCTION, LITERATURE REVIEW, AND STUDY OVERVIEW

INTRODUCTION

Among all the guilds of bird species in North America, none have collectively witnessed population declines that rival those exhibited by the suit of species that are dependent upon savannah, prairie, or grassland ecosystems (Knopf 1994, Samson and Knopf 1994, Brawn et al. 2001, Vickery and Herkert 2001, Brennan and Kuvlesky 2005, Sauer et al. 2005). Knopf (1994) collectively described the group of avian species associated with grassland or savannah habitats as exhibiting “steeper, more consistent, and more geographically widespread declines than any other behavioral or ecological guild.” Brawn et al. (2001) addressed the breadth of the decline in grassland species by comparing population trends of different groups of species based on habitat preference. They found that the grassland and open-savannah habitat groups had the first and third highest proportions of species in decline, 56 % and 33 %, respectively.

Grass-dominated ecosystems were historically the most abundant ecotype in North America, accounting for nearly 50 % of the continent’s land surface, an area encompassing nearly 400 million ha (Sims et al. 1978). When the degree of habitat loss from this ecotype is considered, its not surprising that avian species associated with these ecosystems are in decline. Since European settlement, most North American prairie ecosystems have lost greater than 80 % of their former range (Samson and Knopf 1994, Noss et al. 1995). In regions where the soil fertility is high and the land is desirable for agriculture, these losses have exceeded 99 % (Samson and Knopf 1994). However, until very recently, conservation of grassland, savannah,

and prairie ecosystems, as well as their inhabitants, was largely ignored in North America. Protection efforts have tended to focus on forested, wetland, or tropical ecosystems because of their high biodiversity, yet rarely are these ecosystems as imperiled (Samson and Knopf 1994, Noss et al. 1995, Thompson 2007).

This unfolding conservation crisis has been recognized by the scientific community, and a number of recent studies have begun to address the countless factors contributing to population declines of grassland-obligate birds (Herkert and Knopf 1998, Brennan and Kuvlesky 2005). The taxonomic and geographic ranges of the species dependent upon grasslands make it difficult to unravel the specific causes of their declines. Most research has focused on individual species within particular systems, and the subtle differences between most grassland-obligate species make it difficult to diagnose an all-encompassing mechanism driving population declines. Nevertheless, it does appear apparent that several ubiquitous problems seem to be working independently and concurrently. Most of these issues fit broadly into either of 2 categories: habitat fragmentation and habitat degradation (Herkert et al. 1996, Vickery et al. 1999).

SOURCES OF GRASSLAND BIRD DECLINE

Habitat fragmentation and habitat degradation are separate issues, although they can be intertwined and have similar effects (Caley et al. 2001, Briggs et al. 2005). The most obvious result of habitat fragmentation is a direct decrease in available habitat, but more subtle problems are isolation of remnants, reduction in patch size, increases in the amount of edge within remnant patches, and changes to the character of the landscapes that remain (Johnson 2001). Whereas habitat fragmentation deals with regional decreases in the quantity of habitat and its effects on metapopulation dynamics, habitat degradation effects populations on a much more local level.

Habitat Degradation

Numerous factors can degrade the suitability of grassland habitats for avian species. When in combination, many of these effects can completely change the vegetative integrity of a grassland or prairie. Published literature from across the globe abounds with examples of grassland degradation, most commonly through encroachment of woody-stemmed plant species (Archer et al. 1995). Examples of this phenomenon have been shown in most North American grassland and prairie ecosystems, including tallgrass prairies of the Midwest and Great Plains (Bragg and Hulbert 1976, Knight et al. 1994, Briggs et al. 2002, Hoch et al. 2002), south Texas savannas (Archer et al. 1988), southwestern desert grasslands (Schlesinger et al. 1990), and in the upper Great Basin of Oregon (Miller and Rose 1995). Issues surrounding degradation of North American grassland ecosystems show some variation from region to region, but the underlying problem in all cases seems to be alterations to the natural disturbance regimes.

Historically, the primary ecological force that shaped most North American savannas was fire (Joern and Keeler 1995, Brawn et al. 2001). Both natural and anthropogenic fires were crucial in maintaining the rich communities of graminoids and forbs for which prairies are known. Fire was the most important component in maintaining grassland ecosystems, because in the absence of fire, most grasslands will begin to undergo vegetative succession into forests (Sauer 1950, McPherson 1997), and even slight successional changes can affect suitability for grassland-obligate birds (Grant et al. 2004). Additionally, fire regimes play an important role in maintaining floral diversity and species richness (Hiers et al. 2000, Bowles et al. 2003, Fynn et al. 2004, Feldman and Lewis 2005), and these features may have significant consequences on habitat selection by birds (Rotenberry 1985).

Many studies have addressed the effects that vegetative succession and the subsequent increase in woody-stemmed species have on the distribution, abundance, and demographics of grassland birds. Some of the earliest work on vegetative succession showed that the transition from grassland habitat to shrub-scrub habitat can happen rapidly in the absence of fire or other disturbance. Grassland birds were found to be particularly sensitive to this change, and could become absent within only a few years (Johnston and Odum 1956, Wiens 1969).

More recent studies have also addressed the effects of fire suppression and woody encroachment on grassland birds in the northern Great Plains. Johnson (1997) evaluated the effects of fire on the suite of passerine species that resided on U.S. Fish and Wildlife Service land in central North Dakota. Although responses to fire varied, he was able to assemble all species into distinct groups based on their relationship with fire. All grassland-obligate species reached their highest density on treatment plots in which woody vegetation was reduced through prescribed burns. These findings have been supported through subsequent studies in different regions (Madden et al. 2000, Coppedge et al. 2004), with some results suggesting that only 25 % coverage by woody-stemmed shrubs may be the upper range of tolerance for many grassland passerines (Chapman et al. 2004).

In addition to affecting abundance and distribution, encroachment by woody vegetation has also been implicated in decreasing nest success. Johnson and Temple (1990) found that nest success and brood parasitism by brown-headed cowbirds (*Molothrus ater*) decreased and increased, respectively, for 5 grassland species as distance to woody vegetation decreased and time since burning increased. Burger et al. (1994) also found that proximity to woody vegetation decreased persistence of artificial nests in prairie fragments in southwestern Missouri. With

(1994) found dramatic negative effects of woody shrubs on nesting success of the McCown's longspur (*Calcarius mccownii*), a grassland-obligate passerine in north-central Colorado.

Habitat Fragmentation

In addition to direct declines in habitat quality through encroachment by woody vegetation, grassland birds across North America must face more subtle problems associated with habitat fragmentation. Early studies of the effects of habitat loss and subsequent landscape fragmentation on birds mostly focused on forested ecosystems; however it is now apparent grassland birds are susceptible to the same processes. (Brennan and Kuvlesky 2005). North American prairie ecosystems typically occupy landscapes that are important for agriculture and livestock production; consequently, many grassland ecosystems have been converted to other land uses, thus breaking apart historic prairie landscapes. These new land uses often provide little habitat for grassland species in the areas that remain (Brennan and Kuvlesky 2005).

The most obvious result of habitat fragmentation is the direct loss of available habitat, but a more subtle problem is the nature of the habitat that remains. Species are considered sensitive to fragmentation if they are found in lower densities with decreased individual fitness within patches that remain as the landscape is broken into increasingly smaller blocks (Walters 1998). Not only does fragmentation decrease the amount of available habitat, it can also change the overall context of the landscape. This 'context' encompasses a number of other factors such as patch shape, juxtaposition, availability of corridors, as well as the surrounding matrix in which patches are embedded. These changes can be detrimental to a number of different population parameters such as dispersal and fecundity (Forman and Gordon 1986, Vos et al. 2002).

Over the last decade numerous researchers have tried to quantify the effect of habitat fragmentation and patch size on grassland birds. Vickery et al. (1994) reported that the

abundance of 6 of the 10 grassland species they studied in grassland-barren sites of Maine was positively correlated with patch size. Johnson and Igl (2001) found strong sensitivity to patch size in several species of grassland-obligate passerines located throughout the northern Great Plains, though the degree of sensitivity varied by region and species. Recently, Davis (2004) showed that patch size influenced the frequency of detection for grassland species in southern Saskatchewan.

This trend of area sensitivity has been attributed to several factors, each affecting the demographics of grassland birds in different ways. Foremost among these theories is that reductions in patch size can negatively affect nest success and fecundity (Herkert et al. 2003). Typically, reductions in overall patch size are accompanied by increases in edge-to-interior ratios within the patch. Increases in proportions of edge within patches are believed to increase use by mesomammalian predators (Paton 1994). For example, Winter et al. (2000) demonstrated the effects of increased edge on prairie fragments in Missouri by finding that nesting success was lower and mammalian predator abundance higher within 50 m of forest edges. Nesting success and fecundity can additionally be lowered because increases in nest parasitism by species such as brown-headed cowbirds, which are typically found in higher densities near edges (Johnson and Temple 1990).

Other theories to explain avian sensitivity to patch size have been proposed. Ambuel and Temple (1983) described avian habitat and community interactions in patches of varying sizes. They found no difference in the vegetative composition of habitat in relation to patch size, but observed that the bird community changed dramatically as size decreased. They concluded that abundances of aggressive edge species increased and eventually excluded many of the migratory species. These results suggest that smaller fragment size may cause increases in

interspecific competition within the bird community and creates a situation in which certain species are unable to exist in the presence of edge-related species.

There is also evidence that resource availability may decrease with patch size. In Ontario, the biomass of invertebrates at sample sites in large forest fragments was nearly double that found in sample sites within smaller fragments (Burke and Nol 1998). Additionally, fragment size and degree of isolation may affect insect richness and abundance, particularly for the order Coleoptera (Burke and Goulet 1998). This disparity in resource availability seemed to have measurable consequences on insectivorous ovenbirds (*Seiurus aurocapillus*), and may be contributing to their avoidance of smaller patches. Males in smaller fragments were observed to have lower pairing success, possibly because female ovenbirds found small fragments to be inadequate breeding sites (Burke and Nol 1998).

Fragmentation and the ensuing alteration of landcover types also leads to changes in the character and composition of the overall landscape. These differences in 'landscape context' can have profound effects on the distribution and abundance of certain species. Using a combination of Breeding Bird Survey (BBS) and U.S. Geological Survey landcover data, Flather and Sauer (1996) found that landscape changes negatively affected many species, particularly Neotropical migrants. Abundance of this suite of species was consistently greater in landscapes that were characterized by more natural habitats that were free of human alterations.

Woody encroachment within the overall landscape seems to have the same effect on grassland birds as it does at smaller scales. Grant et al. (2004) studied the response of grassland birds to woody encroachment at the landscape level. They found that occurrence of grassland species decreased dramatically as cover by woodland increased to only 5 – 20 % within a 500 m radius of their survey plots. Similarly, Veech (2006) found that forest landcover was more

common in landscapes adjacent to decreasing populations of grassland birds than in those occupied by stable or increasing populations.

THE DRY PRAIRIE OF SOUTH-CENTRAL FLORIDA

Although the breadth of research surrounding declines in grassland avian species has increased dramatically in recent years, most of the focus has been on prairie ecosystems of the Midwest and Great Plains. Comparatively little attention has been given to savannah and prairie systems within the Southeastern United States; in particular the dry prairie of south-central Florida. This ecosystem provides the most extensive area of grassland habitat remaining in the Southeastern United States (DeSelm and Murdock 1993), and is vital habitat to a number of avian species, some of which are endemic to the region. The dry prairie also serves as critical wintering habitat to a number of migratory grassland passerines that breed in the more northerly latitudes of North America.

Range and Extent

Dry prairie, found only in south-central Florida, is a treeless, pyrogenic landscape dominated by wiregrass (*Aristida beyrichiana*), scattered saw palmetto (*Serenoa repens*), and low-growing runner oak (*Quercus minima*; United States Fish and Wildlife Service 1999). Dry prairie typically occupies upland sites pocketed with scattered wetlands, interspersed ephemeral wet prairies, and “islands” of hardwood hammock and cabbage palm (*Sabal palmetto*). Over the last century, anthropogenic land-use changes in the region have reduced the quantity and quality of this ecosystem; it is estimated that less than 2 % of the dry prairie remains intact in an unaltered state (Noss et al. 1995), prompting its listing as a G2, or globally imperiled, community type (FNAL and DNR 1990) by the Florida Natural Areas Inventory.

Early accounts estimate that in excess of 2 million ha of dry prairie once existed within peninsular Florida (Davis 1967). Most of the historic extent of the dry prairie was along the Kissimmee River in Osceola, Highlands, and Okeechobee Counties, and the region west of Lake Okeechobee throughout Glades, Hendry, Charlotte, and DeSoto Counties, with isolated patches scattered into Sarasota and Manatee Counties (United States Fish and Wildlife Service 1999). Due to differences in mapping classification, there are substantial discrepancies in estimates of the current extant of dry prairie. Kautz et al. (1993) estimated that in 1989, there were approximately 560,000 ha of dry prairie within southern Florida. Cox et al. (1997) gave a considerably greater estimate of the area occupied by dry prairie. They reported 1,385,176 ha of existing dry prairie as of 1989. More recently, in 2003 the Florida Freshwater Fish and Wildlife Commission (2004) completed a Landsat survey of landcover types across the state and listed approximately 497,000 ha of dry prairie. However, it should be noted that their inclusion of cutover flatwoods as dry prairie suggests that the actual quantity of true prairie may be somewhat less than this summation.

Regardless of the authority used, it is obvious that the future of this ecosystem and its wildlife community are in jeopardy. Over the last half-century, the dry prairie has witnessed many of the same ecological transformations to which other North American grasslands have been subjected (Mealor and Prunty 1976, United States Fish and Wildlife Service 1999). The consequences of these changes on the avian community are comparable to other regions. For instance, within peninsular Florida, populations of eastern meadowlark (*Sturnella magna*), northern bobwhite (*Colinus virginianus*), and Bachman's sparrow (*Aimophila aestivalis*) have declined at annual rates of 4.8, 3.8, and 5.0 %, respectively, since the inception of the BBS in 1966 (Sauer et al. 2005). Yet, with the exception of the endangered Florida grasshopper sparrow

(*Ammodramus savannarum floridanus*), scant research exists investigating the underlying mechanisms of the declines for most species within this region.

Threats and Conservation

Arguably the most immediate threat that the avian community faces is the direct loss of prairie habitat and subsequent isolation of remnant patches. Over the last half-century, much of the dry prairie has been converted to other land uses such as exotic grass pastures, citrus groves, and residential development (United States Fish and Wildlife Service 1999). Recent rises in the demand of land for urban development and agriculture have likely accelerated the rate of conversion (Kautz 1998), heightening the need to understand the avian response to landscape level processes.

The effects of habitat loss and fragmentation have been documented for the Florida grasshopper sparrow. Shriver and Vickery (1999) assessed potential grasshopper sparrow habitat and concluded that there was likely an inadequate amount of contiguous dry prairie habitat to insure the persistence of the species. Delany (2007) conducted a follow-up study to that of Shriver and Vickery (1999) and reached a similar conclusion. In reference to the grasshopper sparrow's future, they suggested that, "the paucity and fragmented distribution of suitable dry prairie will be limiting factors for recovery." It is also of note that Delany (2007) found less than 66 % of the amount of "high quality" dry prairie than did Shriver and Vickery (1999), suggesting an alarming amount of habitat loss in the few years that separated the 2 studies.

Perkins et al. (2003) and Perkins and Vickery (2007) indirectly assessed the effects of edge on several members of the dry prairie bird community, including the Florida grasshopper sparrows. They compared nest success and territory densities of each species in relation to distance from patch edges. Density of Florida grasshopper sparrows was found to be greater as

distance from patch edges increased, and all species seemed to exhibit abnormally low nest success, which the authors attributed in part to the fragmented nature of the dry prairie. These results suggest that at least some species that utilize dry prairie may show similar sensitivity to fragmentation and edge effects as do grassland birds in other regions; however, the work of Perkins et al. (2003) and Perkins and Vickery (2007) has not been replicated on dry prairie fragments outside the Kissimmee River valley.

Most of the environmental disturbance regimes that shaped the dry prairie's structure have been altered, further degrading the suitability of this ecosystem for many avian species. Historically, the dry prairie was set within a landscape that was extremely conducive to fire. The climate of southern Florida produces extremely dry winter and spring conditions followed by the onset of a distinct rainy season in early June. The frequent thunderstorms that preceded the rainy season occurred when conditions for ignition of the prairie's fine fuel load were ideal. Lightning strike fires were common, with the highest occurrence in the period from April through mid-June (United States Fish and Wildlife Service 1999). The flat, uninterrupted expanses of prairie offered no obstruction to the wind driven fires, creating a situation in which the only hindrance to burns was lack of accumulated fuel. These conditions kept fire return intervals short, although there is debate as to what the exact fire frequency would have been. It does appear clear, however, that the natural frequency was at least every 1 - 3 years (FNAI and DNR 1990, United States Fish and Wildlife Service 1999), with high probability that most prairies burned practically every year (Harper 1921;1927).

Human objectives for land management on the dry prairie have dramatically changed the frequency and timing of fire. Since the early 1900s, much of the dry prairie has been used for livestock production, and cattlemen frequently set fires during the fall and late winter to promote

new forage for their livestock during times of low natural growth (Mealor and Prunty 1976, Watts 2002). This change from the natural late-spring / early-summer burns to anthropogenic winter burns has shifted the composition of the prairie's plant community, to the detriment of many forbs, toward more woody-stemmed species (Platt et al. 1988, Brewer and Platt 1994, Olson and Platt 1995, United States Fish and Wildlife Service 1999). In recent years, the rise in residential development across the southern portions of the state has led to fire suppression, further decreasing the abundance of many forbs and other flowering species (United States Fish and Wildlife Service 1999, Watts 2002).

This shift toward dominance by woody species on Florida's dry prairie is similar to the phenomenon that has been observed in other grassland ecosystems, and likely has had similar effects on the avian community. Of particular concern is the increased dominance of saw palmetto, which resprouts rapidly after dormant season fires (Tanner et al. 1986, United States Fish and Wildlife Service 1999). Changes in fire regimes have allowed saw palmetto to become much more abundant than it was historically (Yarlett 1965), and this increase has likely been particularly detrimental to grassland birds.

There is limited research available documenting the negative consequences that altered fire regimes and increases in saw palmetto have had on the bird community. Fitzgerald and Tanner (1992) investigated the effects of prairie management and found that grassland-associated bird species were absent from control plots where saw palmetto and shrubs such as gallberry (*Ilex glabra*) were allowed to dominate. Other studies have shown that the Florida grasshopper sparrow to be extremely sensitive to both the timing and frequency of burns. Bachman's sparrows were found to be less sensitive to those factors, but still required frequent fire to provide proper habitat structure (Shriver et al. 1999, Shriver and Vickery 2001, Tucker Jr.

et al. 2006). These studies all concluded that dry prairie should be burned every 1 - 3 years, but in a highly pyrogenic environment such as the dry prairie, the difference between 1-year and 3-year fire return intervals can lead to profound changes in the structure of the vegetative community and the percent coverage by species such as saw palmetto.

The dry prairie of southern Florida also serves as a major over-wintering area for short-distance migrant grassland species that breed on northern prairies. Generally, the wintering ecology of most migrant grassland birds is poorly understood, and this aspect of the life-history cycle in those species has been cited as a major requirement needed to advance grassland bird conservation (Herkert and Knopf 1998, Vickery and Herkert 2001, Brennan and Kuvlesky 2005). Some evidence suggests that declines in fitness and survival of migrants during winter may have serious consequences on continental demographics (Marra et al. 1998, Basili and Temple 1999). The role in which alterations of areas such as Florida dry prairie play in the population trends of grassland-obligate migrants is largely unknown, though it has been suggested that within many Southeastern grasslands, including Florida dry prairie, optimal habitat conditions for wintering grassland birds may no longer exist (Hunter 1990, Lymn and Temple 1991).

Although the dry prairie seems to be at a critical ecological turning point, there are reasons to be optimistic about its future. Recently, non-profit organizations have teamed with state and federal agencies to target dry prairie fragments on private lands for restoration. Similarly, the same partnership is also working toward better management of upland habitats, including dry prairie, on public lands across the state of Florida. These ventures offer much promise to improve conditions for the many species that depend on this ecosystem for habitat. However, to be most effective, gaps in research need to be filled. With the exception of the endangered Florida grasshopper sparrow, there has been little effort to study how habitat and

landscape characteristics influence the remaining members of the dry prairie avian community, and almost no research has been conducted on winter migrants. Considering that several members of these groups are experiencing steep population declines, this information seems critical. The paucity of knowledge that exists regarding the dry prairie and its avian inhabitants suggests that if conservation efforts within the region are to be successful, a better understanding of the factors that drive avian abundance and demographics needs to be brought forth. The intent of this study was to address some of these deficiencies.

STUDY OVERVIEW

The objective of my research was to investigate the relationship between habitat characteristics and variation in the abundance and occurrence of select songbird species that breed and over-winter on south Florida dry prairie. I was also interested in evaluating area sensitivity and the effects of patch size on certain members of the breeding community. An in-depth investigation of these factors on all member of the resident and migrant bird community would have been logistically difficult, therefore I focused my efforts on only 4 species. Two of which, the Bachman's sparrow (*Aimophila aestivalis*) and the eastern meadowlark (*Sturnella magna*), are year-round residents that breed on the dry prairie. The other 2, the grasshopper sparrow (*Ammodramus savannarum pratensis*) and the sedge wren (*Cistothorus platensis*), are grassland-obligate migrants that breed on northern prairies and utilize south Florida savannahs as wintering grounds. All 4 species have been exhibiting steady population declines for at least the last half-century (Sauer et al. 2005), resulting in each being considered species of concern by the conservation community, yet all are still common enough on Florida dry prairie to obtain the samples sizes needed for satisfactory inference about their response to the variables of interest.

This thesis is divided into 3 main chapters, and each chapter is written so that its particular objectives and conclusions are able to stand alone in the context of the overall thesis. This chapter has given an overview of the status of grassland birds, reasons why these species are declining, and how these patterns relate to the dry prairie of southern Florida. In Chapter 2, I evaluate the effects that characteristics such as coverage by saw palmetto, density of the herbaceous layer, and size of prairie fragments have on Bachman's sparrows and eastern meadowlarks. Chapter 3 focuses on the 2 migrant species, grasshopper sparrows and sedge wrens, and how they respond to vegetation variables and prescribed burning. Finally, Chapter 4 offers a summary of my results, what these findings mean for bird conservation within the region, and knowledge gaps that need to be addressed by future research. Ultimately, I believe the outcomes of this project should provide biologists, landowners, and managers with the tools needed to make decisions regarding where and how to focus conservation efforts within the south-central Florida.

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CHAPTER 2
EFFECTS OF PATCH SIZE AND HABITAT CHARACTERISTICS ON BREEDING
BIRDS IN SOUTH FLORIDA'S DRY PRAIRIE

Butler, A.B., J.A. Martin, W.E. Palmer, and J.P. Carroll. To be submitted to The Journal of
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CHAPTER 2

**EFFECTS OF PATCH SIZE AND HABITAT CHARACTERISTICS ON BREEDING
BIRDS IN SOUTH FLORIDA'S DRY PRAIRIE**

INTRODUCTION

Avian species associated with grassland, prairie, and savannah ecosystems across North America have been exhibiting declining populations for the last half-century (Samson and Knopf 1994, Brennan and Kuvlesky 2005, Askins et al. 2007). No other guild of birds has shown steeper, more consistent, and more widespread declines than grassland-obligate birds (Knopf 1994). Problems surrounding the decline of grassland birds have been ignored by the conservation community until recently, however, over the past decade research has begun to focus on the mechanisms underlying this decline. Two central themes acting at separate spatial scales seem to drive population declines. The first is habitat degradation resulting from the advancement of plant succession in the absence of the natural disturbances that maintain prairie conditions. The second is habitat fragmentation resulting from both the loss of grassland habitat and the subsequent reduction in size and availability of the fragments that remain (Herkert et al. 1996, Vickery et al. 1999, Thompson 2007).

The dry prairie of south-central Florida constitutes one of the largest remaining expanses of native grassland in the Southeastern United States. Over the last century, this ecosystem has experienced severe losses in both the extent of its area and the habitat quality of fragments that remain (United States Fish and Wildlife Service 1999). These changes have been the most likely factors fueling regional population declines of bird communities that depend upon grassland and

savannah habitat (Sauer et al. 2005). However, when compared with other North American grassland and savannah systems, few studies have been undertaken to support that assumption.

Research on the endangered Florida grasshopper sparrow (*Ammodramus savannarum floridanus*), an endemic resident of the dry prairie, suggests that that species is area sensitive and requires large patches of habitat (Perkins et al. 2003). Similar to other grassland species, Florida grasshopper sparrows also appear to respond negatively to encroachment by woody-stemmed plant species and are particularly susceptible to increases in saw palmetto (*Serenoa repens*), a woody-stemmed shrub that is common to the region (Delany et al. 1985, Tucker Jr. and Bowman 2006). In the absence of frequent burns, saw palmetto can dominate dry prairie sites, often to the detriment of many grassland birds (Fitzgerald and Tanner 1992).

Though research has given biologists and land managers a decent understanding of the habitat needs of the Florida grasshopper sparrow at both site- and patch-level scales, little to no research has focused on quantifying the effects of patch size and vegetation on other members of the dry prairie community, and consequently objective-based management of dry prairie is largely based on untested assumptions. Therefore, the objectives of this study were to observe and quantify the effects that patch size and habitat characteristics have on members of the dry prairie's avian community. I limited the scope of this study to 2 common, yet regionally declining species: Bachman's sparrow (*Aimophila aestivalis*) and the eastern meadowlark (*Sturnella magna*). Neither of these species is restricted to Florida's dry prairie; however, dry prairie provides critical habitat for each within the south-central portion of the state. Few other avian species have shown higher rates of decline within south-central Florida (Sauer et al. 2005), and I believe that the magnitude of the downward trends suggests that they may have a higher sensitivity to changes in the quantity and quality of dry prairie than do many other species. My

decision to focus on Bachman's sparrows and eastern meadowlarks was based on a belief that their population trends are indicators of ecosystem health, and thus they would serve as adequate "umbrella species" (Lambeck 1997). Therefore, inference gained from their response to habitat conditions could be extrapolated to other members of the dry prairie bird community, thus helping to direct land management toward an approach which would benefit the greatest number of grassland and savannah species that are dependent upon dry prairie.

METHODS

Study Area

This project was conducted within the historic range of the dry prairie on sites within DeSoto, Hardee, Highlands, Manatee, Polk, and Sarasota counties in south-central Florida (Figure 2.1). Five of the study sites were privately-owned ranches that were managed to meet the objectives of cattle production and owner recreation. Each of the remaining 3 sites were publicly owned and managed to preserve the biodiversity of the region. The climate of the region consists of mild, dry winters followed by the onset of a distinct rainy season in early June, causing the summer to be extremely hot and humid. Average annual rainfall is between 1425 and 1440 mm, over 60 % of which falls in the period between June and September. The combination of low elevation, shallow relief, and an impeding soil horizon causes periodic inundation of most sites during the summer rainy season.

Delineation of Dry Prairie Patches

All privately-owned study sites were selected based on landowner willingness to grant access to their property; therefore, selection of dry prairie patches for survey was not random. Instead, when feasible, I surveyed all dry prairie patches on a property, or randomly selected a

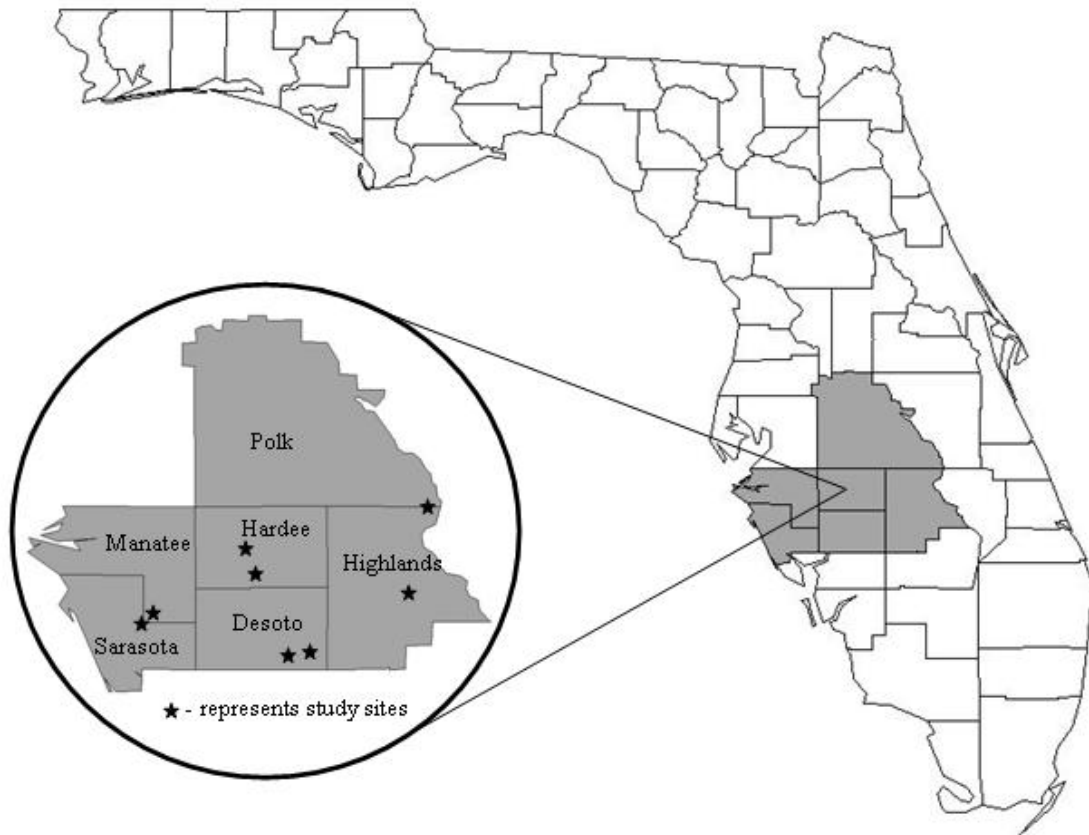


Figure 2.1. Study site locations within Desoto, Hardee, Highlands, Manatee, Polk, and Sarasota counties, Florida.

subset of patches from those available. Dry prairie patches were identified by on-the-ground inspections or through use of satellite imagery (FFFWC 2004). I then visually delineated contiguous patches of dry prairie using ArcView[®] v.3.2 (ESRI 1999). Patch edges were defined as abrupt changes in landcover such as wetland, hardwood hammock, or exotic-grass pasture. Major roads that were paved or covered with gravel, and sand covered roads >10 m wide were also considered patch edges. Interior ephemeral wetlands were considered as part of the patch. Interior fences and unmaintained secondary roads <10 m wide were also considered patch features and were not used to delineate boundaries.

Bird Surveys

I conducted surveys for birds on 4 May – 29 June, 2005 and 27 April – 22 June, 2006 using fixed-distance point counts. The sampling radius for points was chosen to be 100 m; this distance is slightly larger than radii typically used in forested systems, but the openness of the prairie ecosystem was conducive to a larger effective area of detection. Point count locations were randomly distributed within dry prairie patches but were not placed within 100 m of patch edges, and were spaced ≥ 250 m from one another to minimize the probability of double-counting individual birds. Multiple points were established within patches, and the number of points each patch received was approximately proportional to its size. The number of points within patches ranged from 1 – 15; however one large patch (594.2 ha) was surveyed at 31 separate points. Bird surveys began at sunrise and continued for 4 hours and were not conducted in unfavorable weather conditions (Robbins 1981). Each point was visited twice within each year.

Counting techniques that rely on empirical models of detectability are preferred over surveys that merely provide an index of populations or do not account for individuals present but not detected (Rosenstock et al. 2002). Therefore, I utilized distance sampling methods (Buckland

et al. 1993) to account for the failure to observe all birds within survey plots. In the field, birds were assigned into 1 of 5 distance bands (0 -19, 20 – 39, 40 – 59, 60 – 79, and 80 – 100 m) based on their distance from the observer. Prior to data collection, all observers received intensive training sessions to aid in their ability to accurately estimate distances to both visually and aurally detected birds.

Vegetation Surveys

Vegetation measurements were taken using a variation of the methods outlined in Herrick et al. (2005) and visual obstruction methodology proposed by Robel et al. (1970). The percent of coverage by major vegetative functional groups was assessed along 2, 25 m lines that were extended in randomly selected directions from the point center of the bird count. At half-meter intervals along the line, a metal pin-flag was dropped. All functional classes of vegetation touching the pin-flag were recorded, and it was also noted whether the pin-flag struck leaf litter or bare soil. A measurement of vegetation height was taken every 2.5 m. At the 20 and 40 m interval, the density of the vegetation was measured using visual obstruction methodology suggested by Robel et al. (1970). For data analysis, I used the mean vegetation measurements from each point.

Data Analysis

Detection Probability - I believed that detection probability for each species could potentially be influenced by certain habitat characteristics that were of interest. These conditions, such as coverage by saw palmetto, varied between points and could potentially bias inference if not addressed. Therefore, prior to analyzing species response to habitat and patch variables, I investigated detection probability for each species within each year by fitting combinations of half-normal, uniform, and hazard-rate base functions and various series

expansions using both conventional and multiple covariate distance sampling (MCDS) methods in program DISTANCE (Thomas et al. 2005). I constructed base models to predict global detection rates, and then added habitat variables via the MCDS method in an attempt to account for further variation in detection probability (Buckland et al. 2004). All multiple covariate models were built using either the half-normal or hazard-rate base functions with series expansions, and up to 4 adjustment terms. The relative likelihood of each of the candidate models from the MCDS and CDS methods was then assessed using Akaike's Information Criterion (AIC). If models failed to converge or did not meet goodness-of-fit tests, they were not included within the candidate set. For inference about detection rates of each species within each year, I only report the output from the confidence set of models, which I define as those models with ΔAIC values within 4 units of the best-fitting, most parsimonious models (Table 2.1).

Through analysis of the distance data I was able to build global detection models to explain variation in the detection rates of each species within each year. For the eastern meadowlark models in each year and the Bachman's sparrow models in 2005, habitat covariates did not provide better explanations of detection probability than base models, and the uniform detection models with the cosine adjustment term best fit the data (Table 2.1). According to the uniform models, the entire area of the point counts fell within the effective detection radius, and detection probability for eastern meadowlarks in both years and Bachman's sparrows in 2005 was near 1.0 (Figure 2.2). During 2006, the model that best fit the Bachman's sparrow data contained the half-normal detection function with the cosine adjustment term and included the covariate for saw palmetto coverage. The effective detection radius was only 50 m, substantially lower than the previous year (Table 2.1).

Table 2.1. Summary of AIC model selection statistics, effective detection radius (EDR), X^2 goodness-of-fit, and global detection probability with 95% confidence intervals from confidence set of models used explain detection rates of Bachman's sparrow (*Aimophila aestivalis*) and eastern meadowlark (*Sturnella magna*) on Florida dry prairie, 2005 – 2006.

Model	AIC	Δ AIC	EDR	X^2 GOF p-value	Detection Probability	95% Confidence Interval	
						Upper	Lower
Bachman's sparrow							
<i>Aimophila aestivalis</i>							
2005							
Uniform + cosine	500.4	0.00	100.00	0.876	1.00	1.00	1.00
Half-normal + hermite-polynomial	502.3	1.91	97.95	0.775	0.96	0.74	1.00
Half-normal + simple-polynomial	502.3	1.91	97.95	0.775	0.96	0.74	1.00
Half-normal + cosine	502.3	1.91	97.95	0.775	0.96	0.74	1.00
Half-normal + cosine + PALM	504.3	3.88	97.72	0.575	0.95	0.91	1.00
2006							
Half-normal + 1 cosine adjustment + PALM	305.1	0.00	49.54	0.069	0.25	0.20	0.29
Eastern meadowlark							
<i>Sturnella magna</i>							
2005							
Uniform + cosine	457.1	0.00	100.00	0.086	1.00	1.00	1.00
2006							
Uniform + cosine	340.9	0.00	100.00	0.425	1.00	1.00	1.00
Half-normal + cosine	342.9	2.00	99.99	0.277	1.00	0.72	1.00
Half-normal + 1 cosine adjustment + ROBEL	344.2	3.29	83.13	0.271	0.69	0.62	0.78

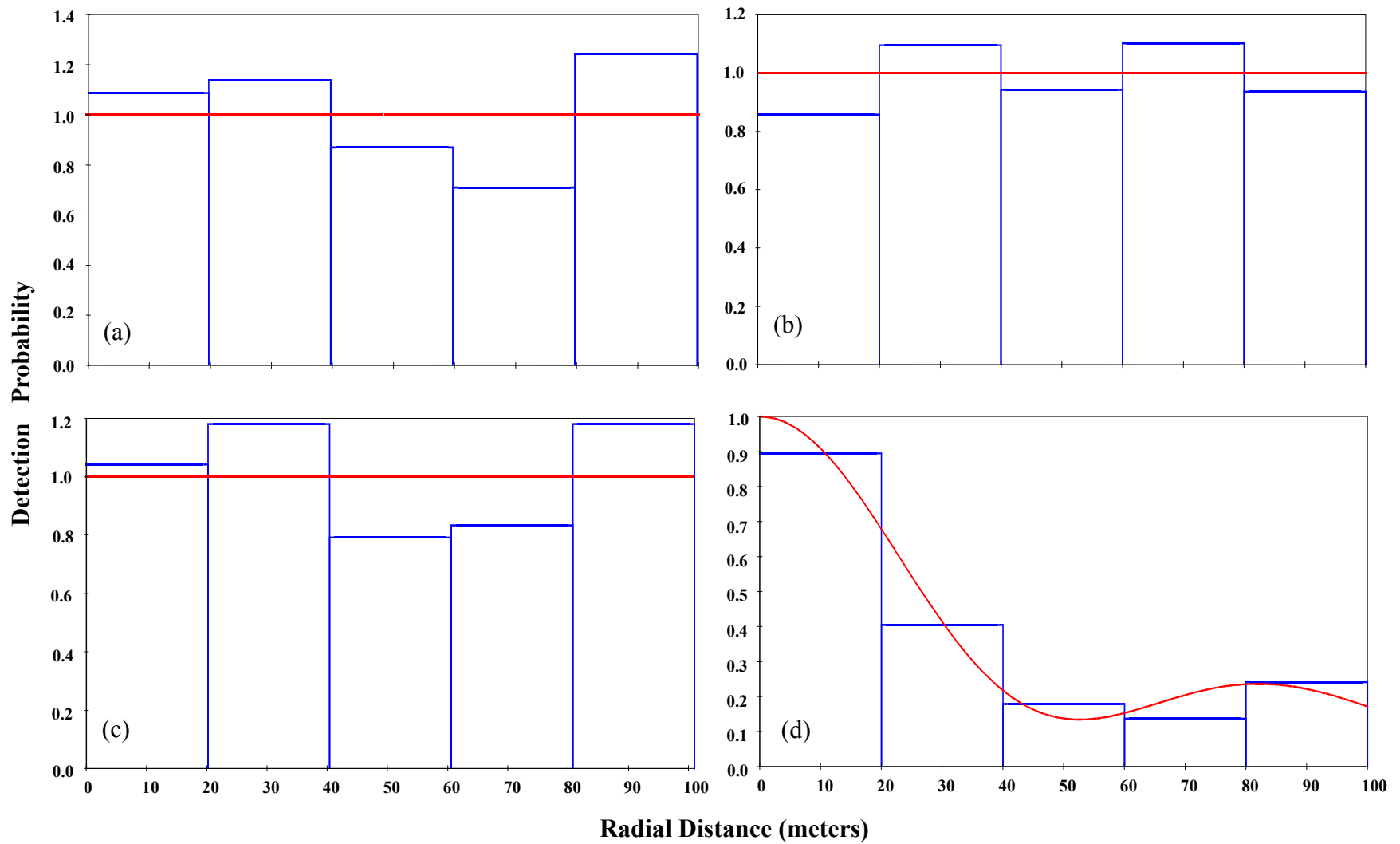


Figure 2.2. Best-fitting global detection function (red line) and detection probability of 5 distance bands for e. meadowlark (*Sturnella magna*) during (a) 2005, (c) 2006, and Bachman's sparrow (*Aimophila aestivalis*) during (b) 2005, (d) 2006 on Florida dry prairie.

Because the DISTANCE models for eastern meadowlark during both years, and Bachman's sparrow in 2005 suggested that detection rates were near 1.0 (Figure 2.2), I assumed detection was approximately 100 %. Under this circumstance, use of the unadjusted count data would not introduce additional bias into the predictor models using those data sets. Use of unadjusted count data for Bachman's sparrows during 2006 was problematic because the detection rate was substantially less than 1.0, and seemed to be correlated with one of the habitat variables (saw palmetto coverage) that would be included within the predictor models (Table 2.1). However, after careful consideration, I decided to proceed with the analysis using the count data. I believed that this was justifiable because, though saw palmetto coverage was in the best-fitting model, detection appeared to remain relatively constant across the various levels of palmetto coverage (Figure 2.3). If detection remained stable in relation to increases in palmetto, then approximately the same proportion of birds would go undetected, regardless of the level of palmetto coverage. Therefore, the inferences made about palmetto coverage would still be valid and not biased because of observers' inability to detect individual birds.

Predictor Models - I used an information-theoretic approach (Burnham and Anderson 2002) to investigate the effects that patch size and habitat variables had on Bachman's sparrows and eastern meadowlarks. Because I was interested in examining the effect of parameters that were acting at different biological scales, I constructed hierarchical models using the PROC NLMIXED procedure of SAS software (SAS Institute 2003). PROC NLMIXED employs the use of mixed models, allowing the incorporation of both fixed and random effects; or in the case of my study, lower level predictors (habitat variables) that were nested within an upper level predictor (patches). By incorporating the random effect of each patch on the intercept and slope of the lower level predictors, I also avoided the pseudoreplication that would have occurred by

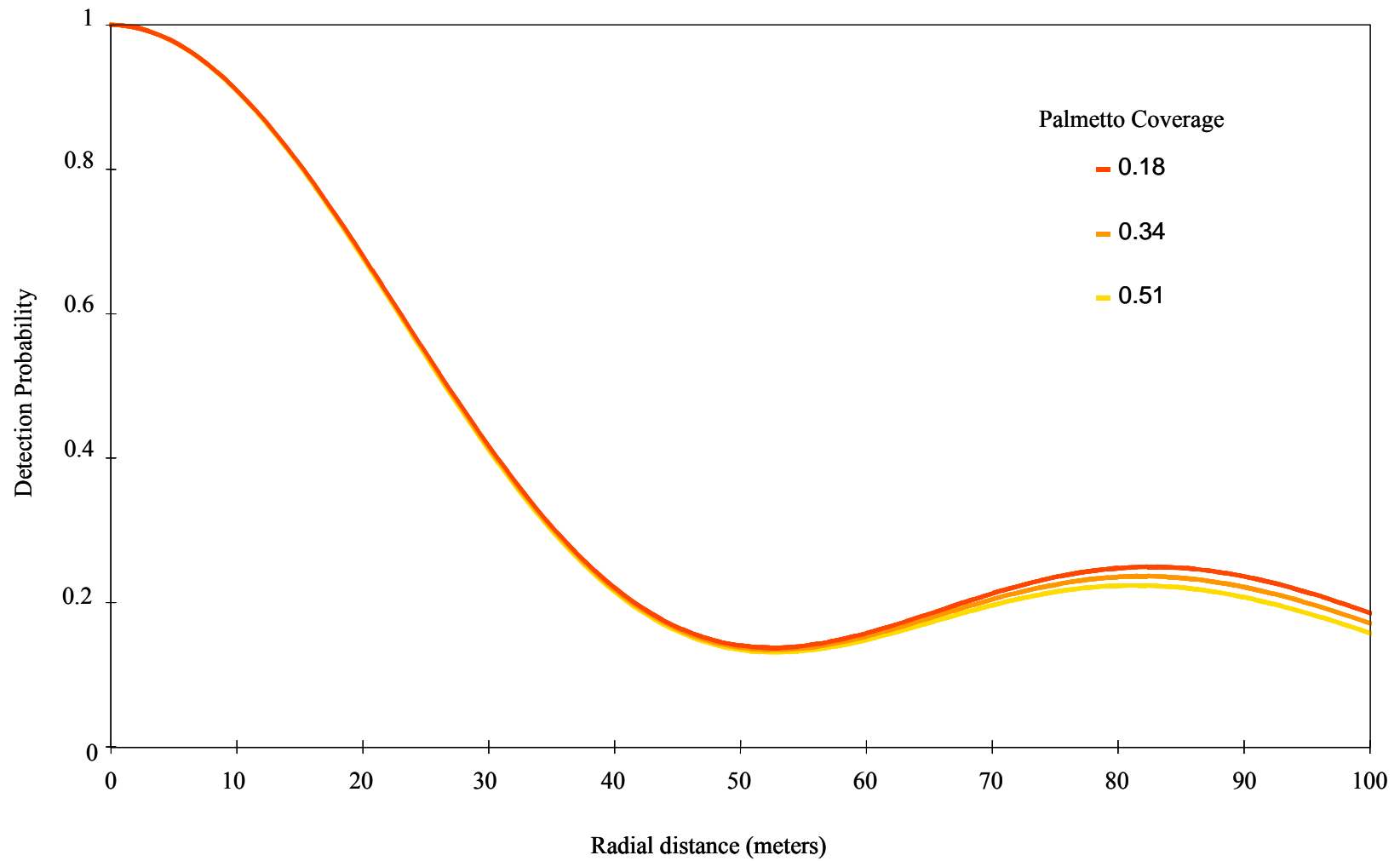


Figure 2.3. Global detection functions for Bachman’s sparrow (*Aimophila aestivalis*) in relation to three levels of percent coverage by saw palmetto (*Serenoa repens*) on south Florida dry prairie, 2006.

failing to acknowledge that the multiple sub-samples (points) within each patch were not independent samples (Hurlbert 1984, Bryk and Raudenbush 1992).

For Bachman's sparrow, count data fit a Poisson distribution; therefore I modeled the relationship between abundance (maximum of the 2 trials) and predictor variables using Poisson regression with a log link function. This approach was not appropriate for eastern meadowlarks because they were often observed in groups containing several individuals, resulting in poor fit of count data to the Poisson distribution. Instead, I created logistic regression models in which the response variable was the occurrence of eastern meadowlarks in either of the 2 trials, and the relationship between occurrence and predictor variables was modeled using a binomial distribution with the logit link.

Fifty-four total patches were surveyed; 22 of which were sampled in both years. Land management activities between years sometimes created substantial year-to-year variation in the habitat characteristics of those patches; therefore, I analyzed years separately. Based on biological hypotheses, I created 10 candidate models for each species which I believed would best account for variation in the abundance of Bachman's sparrows and occurrence of eastern meadowlarks. Models included combinations of patch size (SIZE), 4 habitat characteristic measurements (saw palmetto coverage [PALM], bunch-grass coverage [BUGR], vegetative litter coverage [LIT], mean Robel measurement [ROBEL]), and a binary variable to describe the previous year's fire history (BURN). Patch size measurements were standardized to aid in model convergence by dividing the actual size of each patch by 100. Explanations and biological interpretations of all model covariates can be found in Table 2.2.

I suspected that birds may not show a linear response to all habitat variables. Therefore, prior to conducting the hierarchical analysis, I ran base Poisson and logistic models with each

Table 2.2. Explanation of variables used to predict abundance of Bachman’s sparrows (*Aimophila aestivalis*) and occurrence of eastern meadowlarks (*Sturnella magna*) on Florida dry prairie, 2005 – 2006.

Variable	Explanation
SIZE	Describes the area of each patch in hectares. See <i>Delineation of Dry Prairie Patches</i> section for description of how patches were delineated. To aid in model convergence, the covariate used to describe size within predictor models was the actual patch size / 100.
PALM	Describes the estimate of the percent of coverage by saw palmetto (<i>Serenoa repens</i>) along the bird transect. Measured by the number times saw palmetto was contacted at least once by pins dropped at half-meter intervals along the 2, 25 m vegetation survey transects.
BUGR	Describes the estimate of the percent of coverage by bunch grasses along the bird transect. A few examples of bunch grasses on south Florida dry prairies include: wiregrass (<i>Aristida beyrichiana</i>), bluestems (<i>Andropogon</i> spp.), and panic grasses (<i>Dichantheium</i> spp.). Measured by the number times any type of bunch grass was contacted at least once by pins dropped at half-meter intervals along the 2, 25 m vegetation survey transects.
ROBEL	Describes a measure of visual obstruction, or total biomass within survey plots. Measured using a 1.5 m pole demarcated at half-decimeter intervals. Observers moved approximately 5 m away from the pole and recorded the last half-decimeter segment that was not obscured by vegetation from each of the 4 cardinal directions.
LIT	Describes the estimate of the percentage of the ground covered by vegetative litter of any kind. Measured by the number times that pins dropped at half-meter intervals along the 2, 25 m vegetation survey transects were intercepted by vegetative litter before coming in contact with the ground.
BURN	A binary variable (1 / 0) describing whether a survey point fell within an area that had been burned within the previous year.

habitat variable alone, and examined plots of residuals for signs of a nonlinear response. Bachman's sparrow response to saw palmetto coverage appeared curvilinear; therefore, I included the quadratic form of the variable in all predictor models for that species. To avoid multicollinearity, I calculated correlation coefficients between all possible pairs of data. Saw palmetto coverage and mean Robel measurement were correlated in both years (2005: $r^2 = 0.3783$, 2006: $r^2 = 0.3708$); therefore, I first ran models with saw palmetto and then ran the same model substituting mean Robel measurement for saw palmetto, and compared model fit and likelihood. These substitutions were conducted because I believed that though the covariates were correlated, they did not necessarily represent the same biological phenomenon. Model likelihood was substantially improved when ROBEL was substituted for PALM in all the eastern meadowlark models; thus the final candidate model set for meadowlarks included mean Robel measurement rather than saw palmetto coverage.

To compare the relative plausibility of predictor models, I used the information-theoretic approach suggested by Akaike (1973) and later supported by Burnam and Anderson (2002). This approach, commonly known as AIC and is derived from likelihood theory. I calculated AIC values with the small sample size adjustment (AIC_c ; Burnam and Anderson 2002) for each predictor model; models with lower AIC_c values are considered to be better approximations than higher valued models.

To further facilitate comparisons, a ΔAIC_c value was then obtained by subtracting the AIC value of each model from that of the model with the lowest AIC_c . This value was then normalized to obtain a relative weight, or Akaike weight, for each model. This value, w_i , is interpreted as being the weight of evidence for, or probability in favor of, model i , given the data and the candidate set of models. Goodness-of-fit was assessed for the global model and each

model in the candidate through visually examining plots of the residuals to check for lack-of-fit and obvious outliers.

Though some inference can be gained from model selection, I was most interested in the effect individual covariates had on the abundance and occurrence of the 2 species. The level of effect of covariates can differ among candidate models, but use of model-averaging as a means of incorporating this uncertainty into parameter estimates is not suitable with mixed models (J.T. Peterson, D.B. Warnell School of Forest Resources, University of Georgia, Athens, GA 30605, *personal communication*). Therefore, for inference I report the parameter estimates from all models contained within the confidence set of models. I define the confidence set for predictor models as those models with an Akaike weight of greater than 10 % of the weight of the most parsimonious model (Burnham and Anderson 2002).

RESULTS

I conducted counts for breeding birds at 235 points within 30 patches during 2005, and 228 points within 46 patches during 2006. Patch size ranged from 7.4 – 1392.2 ha in both years, however due to the addition of a study site containing numerous small patches (< 40 ha) in the study's second year, the median patch size was substantially lower in 2006 (median = 108.5 and 39.7 ha in 2005 and 2006, respectively). Forty-two species were observed across both years, and eastern meadowlarks and Bachman's sparrows were the fourth and fifth most abundant species, respectively (see Appendix 2A). Bachman's sparrows were observed at 40.8 % of points during 2005 and 30.7 % of points during 2006. Eastern meadowlarks were observed at 35.7 and 33.7 % of points during 2005 and 2006, respectively. Response to patch size and habitat variables varied between species, and within species between years. During 2006, the 95 % confidence

intervals for all parameters across all models spanned 0, suggested highly variable biological effects existed within that year.

Patch Size

Each species exhibited some level of response to patch size during at least 1 year, but neither was affected by patch size in both years. Eastern meadowlarks responded to patch size during 2005, but not in 2006. In 2005, the confidence set for eastern meadowlarks included 4 of the 5 more inclusive models, 2 of which contained SIZE. In the best-fitting model (SIZE + ROBEL + BUGR + LIT; Akaike weight = 0.3898; Table 2.3), the parameter estimate for SIZE suggested that increases in patch size had a positive influence on probability of occurrence of meadowlarks (Table 2.4 and Figure 2.4). There was a large amount of uncertainty surrounding selection of the best-fitting, most parsimonious model in 2006. Nine of the 10 candidate models were contained in the confidence set, including the SIZE model, as well as models that contained SIZE in conjunction with other parameters. However, the SIZE model only received 9.31 % of the overall model weight, which only comprised $9.31 / 28.77 = 32.4$ % of the weight of the best-fitting model (Table 2.3). The parameter estimates from the SIZE model, as well other models within the confidence set that contained SIZE, all exhibited weak and highly variable effects of the covariate, suggesting that patch size was unrelated to meadowlark occurrence during that year (Table 2.4 and Figure 2.4).

Bachman's sparrows did not seem influenced by patch size during 2005. During that year, the model containing only SIZE was not included within the confidence set, and was the second least-likely model (Akaike weight = 0.0014; Table 2.5). Furthermore, parameter estimates from the 2 models within the confidence set that included the additive effects of SIZE with other variables suggested patch size had a non-significant effect on abundance (Table 2.6).

Table 2.3. Predictors, number of parameters (K), Akaike's Information Criterion with small-sample size adjustment (AIC_c), ΔAIC_c , and Akaike weights (w_i) for the set of candidate models used to predict the probability of occurrence of eastern meadowlark (*Sturnella magna*) on south Florida dry prairie, 2005 – 2006.

Model	K	AIC_c	ΔAIC_c	w_i
2005				
SIZE + ROBEL + BUGR + LIT	6	267.44	0.0000	0.3898
ROBEL + BUGR	4	268.13	0.6964	0.2752
SIZE + ROBEL + BUGR + LIT + BURN (<i>Global</i>)	7	269.06	1.6267	0.1728
ROBEL + BUGR + LIT	5	270.08	2.6375	0.1043
BUGR	3	272.27	4.8363	0.0347
ROBEL	3	273.55	6.1123	0.0183
SIZE	3	277.46	10.0231	0.0026
SIZE + BURN	4	279.53	12.0905	0.0009
LIT	3	280.22	12.7811	0.0007
BURN	3	280.46	13.0225	0.0006
2006				
ROBEL	3	289.46	0.0000	0.2877
ROBEL + BUGR	4	290.47	1.0063	0.1740
BUGR	3	290.96	1.4891	0.1366
SIZE	3	291.72	2.2572	0.0931
LIT	3	292.27	2.8002	0.0709
BURN	3	292.27	2.8035	0.0708
ROBEL + BUGR + LIT	5	292.35	2.8856	0.0680
SIZE + ROB + BUGR + LIT	6	293.01	3.5434	0.0489
SIZE + BURN	4	293.79	4.3253	0.0331
SIZE + ROBEL + BUGR + LIT + BURN (<i>Global</i>)	7	295.14	5.6699	0.0169

Table 2.4. Estimates, standard error, 95 % confidence interval, and p-value ($\alpha = 0.05$) for parameters included within models contained within the confidence set of models used to predict probability of occurrence of eastern meadowlark (*Sturnella magna*) on south Florida dry prairie, 2005 – 2006.

Parameter	Estimate	SE	95% Confidence Interval		P-value
			Lower	Upper	
2005					
SIZE + ROBEL + BUGR + LIT					
Intercept	-2.0936	0.7934	-3.7163	-0.4709	0.0132
SIZE	0.1563	0.0754	0.0020	0.3106	0.0473
ROBEL	-0.2777	0.1063	-0.4950	-0.0603	0.0141
BUGR	2.1911	0.7939	0.5673	3.8149	0.0099
LIT	0.3121	0.7986	-1.3212	1.9455	0.6988
ROBEL + BUGR					
Intercept	-1.5064	0.6919	-2.9214	-0.0913	0.0377
ROBEL	-0.2451	0.1027	-0.4552	-0.0350	0.0238
BUGR	2.0974	0.7860	0.4898	3.7051	0.0123
SIZE + ROBEL + BUGR + LIT + BURN (<i>Global</i>)					
Intercept	-2.1875	0.8135	-3.8514	-0.5236	0.0118
SIZE	0.1744	0.0815	0.0078	0.3410	0.0408
ROBEL	-0.2799	0.1062	-0.4970	-0.0627	0.0133
BUGR	2.3099	0.8180	0.6369	3.9828	0.0085
LIT	0.3783	0.8070	-1.2721	2.0287	0.6427
BURN	-0.3562	0.5130	-1.4054	0.6930	0.4930
ROBEL + BUGR + LIT					
Intercept	-1.5817	0.7192	-3.0526	-0.1108	0.0360
ROBEL	-0.2536	0.1048	-0.4679	-0.0393	0.0220
BUGR	2.1220	0.7878	0.5108	3.7332	0.0116
LIT	0.3081	0.7950	-1.3178	1.9340	0.7012

Table 2.4 continued.

Parameter	Estimate	SE	95% Confidence Interval		P-value
			Lower	Upper	
2006					
ROBEL					
Intercept	-0.4861	0.3208	-1.1323	0.1600	0.1367
ROBEL	-0.1531	0.0932	-0.3407	0.0345	0.1073
ROBEL + BUGR					
Intercept	-0.9237	0.5320	-1.9952	0.1478	0.0894
ROBEL	-0.1479	0.0942	-0.3377	0.0419	0.1235
BUGR	0.6624	0.6304	-0.6072	1.9320	0.2990
BUGR					
Intercept	-1.3475	0.4727	-2.2996	-0.3954	0.0066
BUGR	0.7321	0.6281	-0.5329	1.9972	0.2499
SIZE					
Intercept	-1.0058	0.2984	-1.6068	-0.4049	0.0015
SIZE	0.0380	0.0525	-0.0678	0.1438	0.4730
LIT					
Intercept	-0.8644	0.3830	-1.6358	-0.0929	0.0289
LIT	-0.0255	0.5698	-1.1733	1.1222	0.9644
BURN					
Intercept	-0.8780	0.2548	-1.3912	-0.3647	0.0012
LIT	-0.0003	0.3789	-0.7634	0.7628	0.9994
ROBEL + BUGR + LIT					
Intercept	-1.0143	0.5637	-2.1497	0.1210	0.0787
ROBEL	-0.1641	0.1001	-0.3658	0.0375	0.1082
BUGR	0.6311	0.6325	-0.6428	1.9050	0.3237
LIT	0.2878	0.6007	-0.9220	1.4976	0.6342

Table 2.4 continued.

Parameter	Estimate	SE	95% Confidence Interval		P-value
			Lower	Upper	
SIZE + ROBEL + BUGR + LIT					
Intercept	-1.1773	0.5835	-2.3525	-0.0021	0.0496
SIZE	0.0006	0.0005	-0.0004	0.0016	0.2409
ROBEL	-0.1895	0.1028	-0.3964	0.0175	0.0718
BUGR	0.6968	0.6368	-0.5858	1.9794	0.2797
LIT	0.2720	0.5984	-0.9332	1.4772	0.6516
SIZE + BURN					
Intercept	-1.0161	0.3225	-1.6656	-0.3665	0.0029
SIZE	0.0004	0.0005	-0.0007	0.0014	0.4694
BURN	0.0321	0.3848	-0.7429	0.8072	0.9339

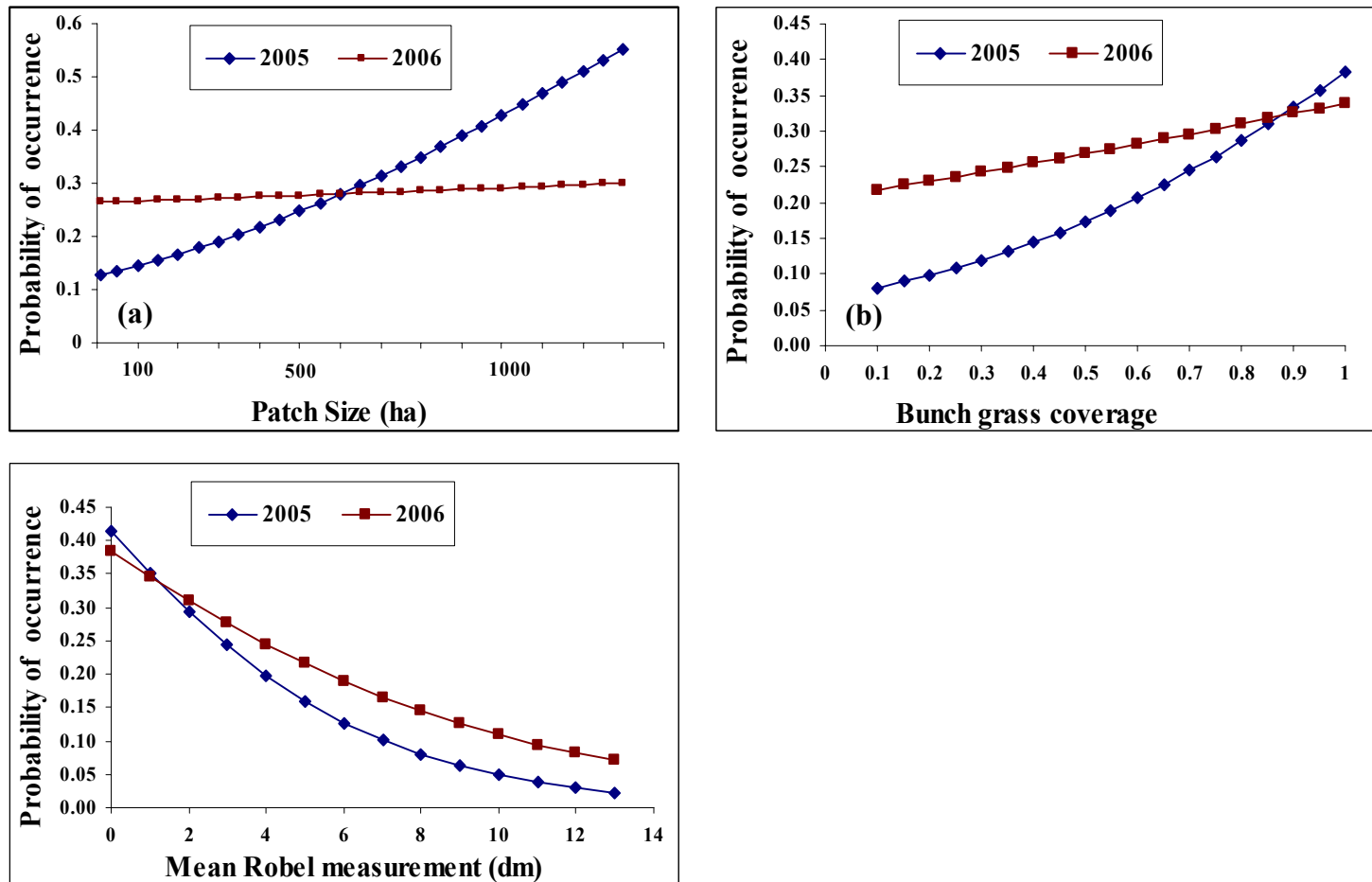


Figure 2.4. Predicted relationship between eastern meadowlark (*Sturnella magna*) occurrence and (a) patch size, (b) bunch grass coverage, and (c) mean Robel measurement on south Florida dry prairie, 2005 - 2006. Trend lines can be interpreted as the mean fixed effect averaged across all confidence set models containing SIZE, BUGR, or ROBEL. For models that contained additional variables, all other covariates were held constant at their mean.

Table 2.5. Predictors, number of parameters (K), Akaike's Information Criterion with small-sample size adjustment (AIC_c), ΔAIC_c , and Akaike weights (w_i) for the set of candidate models used to predict the abundance of Bachman's sparrow (*Aimophila aestivalis*) on south Florida dry prairie, 2005 – 2006.

Model	K	AIC_c	ΔAIC_c	w_i
2005				
PALM ² + BUGR	5	473.72	0.0000	0.5204
PALM ² + BUGR + LIT	6	475.80	2.0813	0.1838
PALM ²	9	477.52	3.8027	0.0777
SIZE + PALM ² + BUGR + LIT + BURN (<i>Global</i>)	8	477.63	3.9123	0.0736
SIZE + PALM ² + BUGR + LIT	7	477.79	4.0766	0.0678
BUGR	3	477.86	4.1442	0.0655
BURN	3	482.84	9.1261	0.0054
SIZE + BURN	4	484.11	10.3965	0.0029
SIZE	3	485.53	11.8149	0.0014
LIT	3	485.56	11.8452	0.0014
2006				
SIZE	3	357.40	0.0000	0.3023
LIT	3	358.48	1.0809	0.1761
BUGR	3	358.87	1.4713	0.1449
BURN	3	358.94	1.5439	0.1397
SIZE + BURN	4	359.08	1.6797	0.1305
PALM ²	4	360.70	3.3004	0.0581
PALM ² + BUGR	5	362.55	5.1455	0.0231
SIZE + PALM ² + BUGR + LIT	7	363.98	6.5836	0.0112
PALM ² + BUGR + LIT	6	364.32	6.9197	0.0095
SIZE + PALM ² + BUGR + LIT + BURN (<i>Global</i>)	8	365.81	8.4133	0.0045

Table 2.6. Estimates, standard error, 95 % confidence interval, and p-value ($\alpha = 0.05$) for parameters included within models contained within the confidence set of models used to predict abundance of Bachman's sparrows (*Aimophila aestivalis*) on south Florida dry prairie, 2005 – 2006.

Parameter	Estimate	SE	95% Confidence Interval		P-value
			Lower	Upper	
2005					
PALM ² + BUGR					
Intercept	-2.1124	0.4613	-3.0558	-1.1690	< 0.0001
PALM	5.2612	1.9746	1.2227	9.2997	0.0125
PALM ²	-6.4576	2.5401	-11.6527	-1.2626	0.0166
BUGR	1.1134	0.4600	0.1725	2.0542	0.0220
PALM ² + BUGR + LIT					
Intercept	-2.0936	0.4757	-3.0664	-1.1207	0.0001
PALM	5.2492	1.9749	1.2101	9.2883	0.0127
PALM ²	-6.4121	2.5530	-11.6335	-1.1907	0.0178
BUGR	1.1159	0.4609	0.1733	2.0585	0.0219
LIT	-0.0718	0.4527	-0.9977	0.8541	0.8751
PALM ²					
Intercept	-1.5170	0.3909	-2.3164	-0.0718	0.0006
PALM	5.7195	1.9567	1.7176	9.7213	0.0067
PALM ²	-7.2193	2.5042	-12.3410	-2.0977	0.0073
SIZE + PALM ² + BUGR + LIT + BURN (<i>Global</i>)					
Intercept	-1.9821	0.4838	-2.9716	-0.9926	0.0003
SIZE	-0.0321	0.0363	-0.1063	0.0420	0.3830
PALM	5.0734	1.9531	1.0789	9.0679	0.0146
PALM ²	-6.0881	2.5330	-11.2686	-0.9076	0.0229
BUGR	1.0578	0.4605	0.1159	1.9996	0.0290
LIT	-0.1441	0.4531	-1.0709	0.7827	0.7527
BURN	0.4006	0.2582	-0.1285	0.9297	0.1323

Table 2.6 continued.

Parameter	Estimate	SE	95% Confidence Interval		P-value
			Lower	Upper	
SIZE + PALM ² + BUGR + LIT					
Intercept	-2.0486	0.4885	-3.0476	-1.0496	0.0002
SIZE	-0.0131	0.0361	-0.0869	0.0607	0.7193
PALM	5.2347	1.9739	1.1976	9.2717	0.0128
PALM ²	-6.3572	2.5588	-11.5905	-1.1238	0.0190
BUGR	1.1205	0.4610	0.1776	2.0633	0.0215
LIT	-0.0767	0.4527	-1.0025	0.8490	0.8666
BUGR					
Intercept	-1.3416	0.2934	-1.9416	-0.7416	0.0001
BUGR	1.2345	0.4436	0.3272	2.1419	0.0094
2006					
SIZE					
Intercept	-0.8734	0.1726	-1.2210	-0.5257	< 0.0001
SIZE	-0.0437	0.0325	-0.1091	0.0217	0.1854
LIT					
Intercept	-1.2121	0.2783	-1.7727	-0.6515	< 0.0001
LIT	0.3248	0.4195	-0.5201	1.1696	0.4428
BUGR					
Intercept	-1.1698	0.3400	-1.8546	-0.4851	0.0013
BUGR	0.2094	0.4599	-0.7169	1.1357	0.6511
BURN					
Intercept	-0.9999	0.1637	-1.3295	-0.6703	0.0000
BURN	-0.1017	0.2760	-0.6576	0.4541	0.7141
SIZE + BURN					
Intercept	-0.8123	0.1936	-1.2022	-0.4225	0.0001
SIZE	-0.0458	0.0316	-0.1094	0.0178	0.1540
BURN	-0.1628	0.2691	-0.7049	0.3793	0.5483

Table 2.6 continued.

Parameter	Estimate	SE	95% Confidence Interval		P-value
			Lower	Upper	
PALM ²					
Intercept	-1.1273	0.3060	-1.7437	-0.5109	0.0006
PALM	0.1664	1.7883	-3.4355	3.7683	0.9263
PALM ²	0.2772	2.3446	-4.4452	4.9995	0.9064

Conversely, the model containing only SIZE was best-fitting (Akaike weight = 0.3023) for Bachman's sparrows during 2006, and was $0.3023 / 0.1761 = 1.71$ times more likely than the second ranked model containing only LIT (Akaike weight = 0.1761; Table 2.5). Parameter estimates from SIZE model suggest that Bachman's sparrows generally exhibited a slightly negative, though not statistically significant ($p = 0.1854$), response to increasing patch size in that year (Table 2.6).

Habitat Characteristics

The ROBEL variable improved likelihood and model fit when it was interchanged with PALM in candidate models for Eastern meadowlark. In 2005, 3 of the 4 models in the confidence set for Eastern meadowlark contained ROBEL. In 2006, the model containing only ROBEL best fit the meadowlark data (Akaike weight = 0.2877), and was $0.2877 / 0.1740 = 1.65$ times more likely than the second best-fitting model, which contained the additive effect of ROBEL and BUGR (Akaike weight = 0.1740; Table 2.3). During both years, meadowlarks exhibited an inverse relationship to ROBEL measurements; as the mean Robel measurement increased, the probability of occurrence of meadowlarks decreased, though the effect was stronger during the first year (Table 2.4 and Figure 2.4).

High levels of bunch grass were important to both species. The model containing the additive effect of BUGR and PALM best fit the Bachman's sparrow data in 2005 (Akaike weight = 0.5204), and every model that contained BUGR was included within the confidence set of models (Table 2.5). Furthermore, BUGR had a significant positive effect on the slope of each model in which it was contained (Table 2.6 and Figure 2.5). In 2006, the BUGR model received the third-highest amount of support (Akaike weight = 0.1449), which was approximately half of the weight in favor of the top model (Akaike weight = 0.3023; Table 2.3). However, the effect

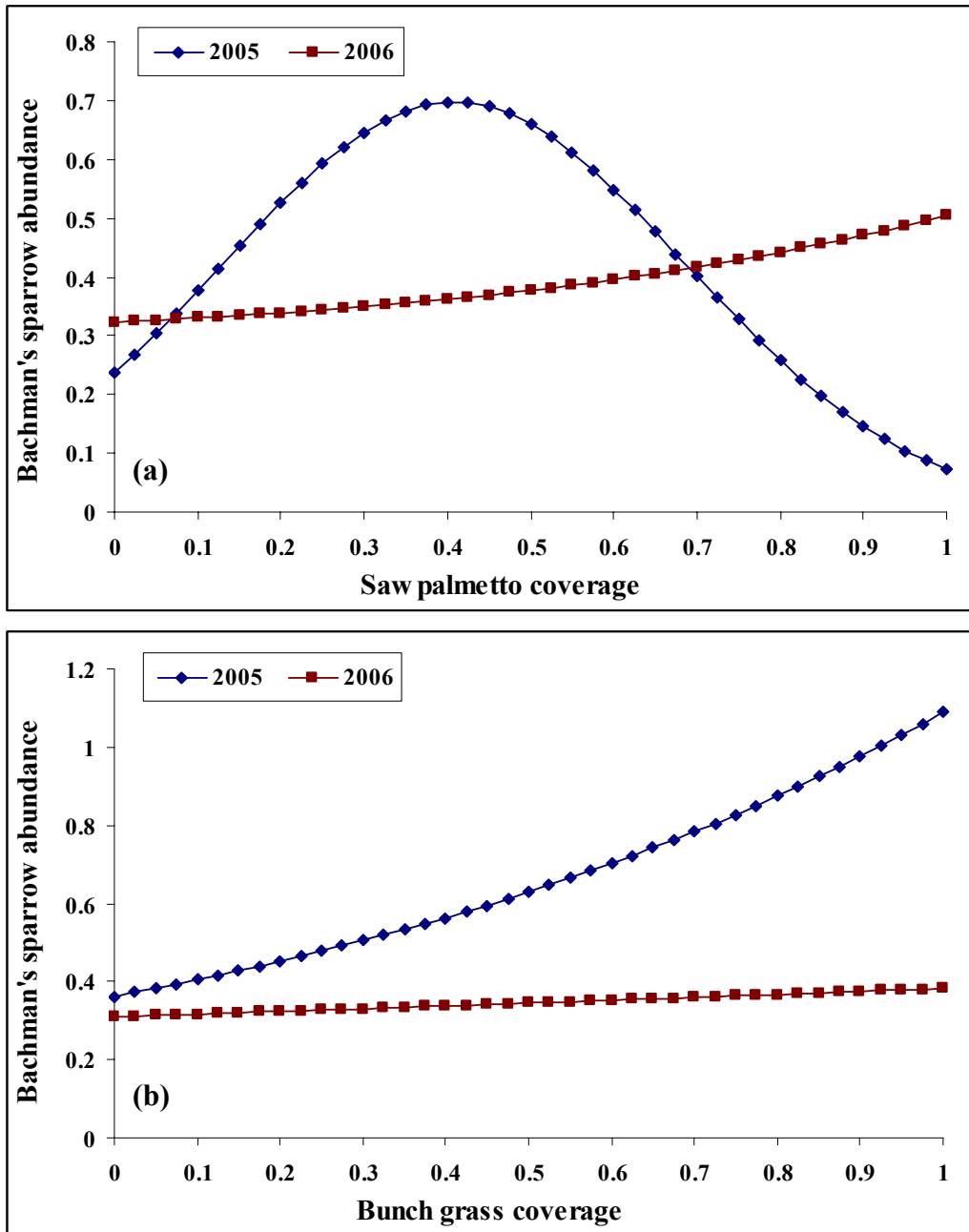


Figure 2.5. Predicted relationship of Bachman's sparrow (*Aimophila aestivalis*) abundance to coverage by (a) saw palmetto, and (b) bunch grass on south Florida dry prairie, 2005 - 2006.

Trend lines can be interpreted as the mean fixed effect averaged across all confidence set models containing PALM² or BUGR. For models that contained additional variables, all other covariates were held constant at their mean.

of bunch grass was not significant in any model within the confidence set that contained the variable (Table 2.6). BUGR was included in all 4 of the models of the confidence for eastern meadowlark in 2005 (Table 2.3), and the parameter estimates from each model suggests that increases in BUGR had a significant positive relationship with occurrence of meadowlarks (Table 2.4). In 2006, BUGR was included in the second and third best-fitting models, which respectively carried 60.4 and 47.4 % of the weight of the top model (Table 2.3), although the 95 % confidence intervals of the parameter estimates from the models did span 0 (Table 2.4).

Because the ROBEL variable provided better fit in the eastern meadowlark models, I only investigated the effect of PALM on Bachman's sparrows. Comparison of the base models and their residuals containing the linear and quadratic form of the PALM suggested that Bachman's sparrows likely responded to saw palmetto in a non-linear fashion (Figure 2.5). Therefore, all candidate models predicting Bachman's sparrow abundance contained the curvilinear form of the variable. In 2005, all the models that contained PALM were included in the confidence set of models, and the model for PALM + BUGR was best-fitting and received over half of the overall model Akaike weight (0.5204; Table 2.5). During 2005, Bachman's sparrow abundance increased with saw palmetto coverage until coverage reached approximately 40 %, at which point abundance became inversely related to saw palmetto coverage and steadily declined (Figure 2.5). PALM did not appear to have an effect on Bachman's sparrow abundance in 2006. The model containing only PALM was the only model in the confidence set that contained the variable (Akaike weight = 0.0581), and it only had approximately 20 % of the weight of the best-fitting model (Akaike weight = 0.3023; Table 2.5). Parameter estimates of PALM from that model spanned 0 (Table 2.6), and suggested that during 2006 the effect of PALM on Bachman's sparrow abundance was negligible (Figure 2.5).

DISCUSSION

A substantial amount of variability existed within the relationship of each species to both patch size and habitat variables. This was not only true for comparisons between species, but also within species across years. For example, eastern meadowlarks and Bachman's sparrows seemed to differ in their relationship to patch size, and the effect of patch size was not consistent between years for either species. Furthermore, the magnitude of the effect of the habitat variables varied between years, although the direction typically did not, and some similarities did exist between species.

Though variability between species was expected, the degree of temporal variation I observed within species was somewhat surprising. However, previous research has suggested that high levels of variability may be the norm when investigating the relationship between grassland birds and their environment. Winter et al. (2005) provided evidence that grassland bird response to habitat characteristics is highly variable, both spatially and temporally. They reported that considerable variation in the magnitude and response to habitat variables existed between species, and in some cases, from region-to-region for the same species. They reasoned that this variability was likely due to the dynamic nature of grassland environments, i.e., because grasslands are highly disturbance dependent, habitat conditions are constantly changing and species that inhabit these areas must remain highly adaptable. In an ensuing study, Winter et al. (2006) suggested that the effects of patch and landscape level variables, such as patch size, may also vary among species, regions, and even years. Similar findings were reported by both Johnson and Igl (2001) and Davis (2003) from research on grassland systems in the Great Plains. My results lend support to the conclusions of these authors, and provide an example of this phenomenon for a grassland system within the Southeast.

Patch Size

Results of this study suggest that eastern meadowlarks on Florida dry prairie may exhibit area sensitivity in certain years. During 2005, eastern meadowlarks were not observed within patches that were less than 40 ha in size, even though this is nearly 7 times greater than the largest territory size reported for the species (Lanyon 1995). Previous studies on the effects of patch size on Eastern meadowlarks have yielded conflicting results. In Missouri and New York density and abundance of eastern meadowlarks was not related to patch size (Bollinger 1995, Winter and Faaborg 1999). However, Herkert (1994) found that eastern meadowlarks were sensitive to area in Illinois, and were rarely found on grassland patches that were less than twice the size of the species' average territory size. Herkert et al. (2003) suggested that patterns of area sensitivity in eastern meadowlarks may be correlated with declines in nest success that occur as patch size decreases. This relationship has also been suggested for eastern meadowlarks on Florida dry prairie; Perkins and Vickery (2007) observed abnormally low nest success in dry prairie fragments, which they believed was attributed to the small and fragmented nature of their study sites.

Bachman's sparrows seemed to show an inverse relationship to patch size during the study's second year. Little other empirical research exists directly examining the response of Bachman's sparrows to patch size, although some studies have inadvertently offered clues into this relationship. Krementz and Christie (2000) included Bachman's sparrows within a larger study relating clear-cut size to richness of early successional species, and found that abundance of Bachman's sparrows did not vary with patch size. However, evidence does suggest that populations of Bachman's sparrows may be limited by dispersal, a life-history trait that can be influenced by the size of remnant patches within the context of the overall landscape (Forman

and Gordon 1986). Pulliam et al. (1992) and Dunning et al. (1995) both investigated patch isolation and source-sink dynamics of the species and found that these population processes may potentially explain the lack of Bachman's sparrows in areas of seemingly suitable habitat - a phenomenon that has puzzled many ornithologists studying the decline of the species (Dunning 2006).

Dispersal and immigration have also been cited as issues regulating populations of Bachman's sparrows on dry prairie within south Florida. Perkins et al. (2003) studied reproductive success of Bachman's sparrows on dry prairie sites and found that within their large study patches (>700 ha), reproduction was not sufficient to maintain populations, prompting them to suggest that large dry prairie patches serve as a sink, and Bachman's sparrow persistence on Florida dry prairie may be dependent on immigration from nearby pine flatwoods. However, these conclusions seem somewhat erroneous because little ecological difference exists between south Florida pine flatwoods and adjoining dry prairie, other than the presence of low densities of scattered trees in the former, and lack thereof in the latter (Abrahamson and Harnett 1990). Vegetative structure and composition varies little between the two, and it is these features to which Bachman's sparrows would likely respond (Dunning 2006). Perkins et al. (2003) made their conclusion regarding the importance of flatwoods by assuming that nest success and clutch size within south Florida pine flatwoods were equivalent to piney woods in Arkansas; this assumption may or may not be reasonable. Additionally, they calculated nest success from admittedly low sample sizes that could have led to false inferences about the quality of dry prairie for nesting.

Yet, the preference that Bachman's sparrows showed for smaller patches within my study seems somewhat supportive of the conclusions of Perkins et al. (2003), though the variability of

my results make a definitive answer on the issue somewhat premature. The model including patch size effects was only selected as a member of the confidence set of models within 1 year and the estimate from that model suggested a high level of variability surrounding the effect of patch size. If the assumption of Perkins et al. (2003) was correct, then the effect of patch size that I observed in 2006 would be expected; Bachman's sparrows should concentrate in smaller patches when those patches were adjacent to flatwoods because large blocks of dry prairie were undesirable habitat and were avoided. Again, however, this theory seems unrealistic because ecologically, there are few differences between dry prairie and pine flatwoods.

I believe that more plausible explanations exist for the results of both Perkins et al. (2003) and my study. The dry prairie fragments that Perkins et al. (2003) used as study sites showed abnormally low reproductive success not only for Bachman's sparrow, but also for other species (Perkins et al. 2003, Perkins and Vickery 2007). In a subsequent paper, they attributed the abnormally low nest success they found in eastern meadowlark and common ground dove (*Columbina passerina*) on the fragmented nature of the landscape surrounding their study sites. These were the same sites that they used in their study of Bachman's sparrow nest success. The range of nest success they reported for those species was similar to the range they reported for Bachman's sparrow, and there is little reason to doubt that the fragmented nature of their sites could also have contributed to the abnormally low nest success they observed in Bachman's sparrows, making their assertion of the value of dry prairie false.

Additionally, it is possible that the results I found could be attributed to differences in the probability of dispersal among study sites, rather than the actual effects of patch size. The year in which Bachman's sparrows showed signs of inverse area sensitivity coincided with the year in which I included the Avon Park Air Force Range (APAFR) as a study site. The APAFR is a

42,941 ha landscape composed of a primarily dry prairie matrix; however many of the actual patches of dry prairie are separated by small, linear blocks of planted slash pine (*Pinus caribaea*). The relatively minute widths of the planted pine stands do not likely serve as a hindrance to dispersing Bachman's sparrows. Therefore, APAFR is a landscape that is dominantly dry prairie, yet composed of small patches, and dispersal between patches can easily occur. At most of my other study sites, dry prairie patches were often separated by large expanses of exotic grass pasture, creating a landscapes that were inhospitable to dispersing Bachman's sparrows. If the landscape of APAFR was more conducive to dispersal than my other sites, the smaller patches of APAFR would have likely held higher densities of Bachman's sparrows than patches at other sites, and indeed, this seemed to be the case. A post hoc analysis of the density estimates from program DISTANCE revealed that 7 of the 10 highest densities of Bachman's sparrows were recorded in patches at APAFR, and the average size of these patches was relatively small ($\bar{x} = 46.7$ ha, SD = 22.8; A.B. Butler, D.B. Warnell School of Forestry and Natural Resources, University of Georgia, *unpublished data*). Therefore the inclusion of a large number of small patches (nearly half of all survey patches in 2006 were located at APAFR) from a landscape that offered high dispersal could have yielded patch size effects that were not representative of the true biological response that would be seen across all dry prairie sites in south-central Florida.

Habitat Characteristics

Eastern meadowlarks responded relatively strongly in both years to the mean Robel measurement within survey plots. Inclusion of the Robel variable improved model likelihood and fit when interchanged with saw palmetto, signifying that meadowlarks were responding to the overall density of dry prairie vegetation rather than the specific effects of saw palmetto

overabundance. Eastern meadowlark occurrence was inversely related to mean Robel measurement, suggesting that within my dry prairie sites meadowlarks preferred less dense vegetation. This is similar to the findings of others. For instance, McCoy et al. (2001) found higher mean abundance of eastern meadowlarks in shorter, less dense Conservation Reserve Program (CRP) fields. In Nebraska, eastern meadowlarks avoided CRP fields in favor of native prairies, possibly because CRP was too tall and dense (King and Savidge 1995). Schroeder and Sousa (1982) created a habitat suitability model for eastern meadowlarks that predicted that habitat quality for meadowlarks would decline as density of shrub coverage increased.

A threshold value for coverage by saw palmetto appeared to exist for Bachman's sparrows; low levels of saw palmetto seemed favorable and abundance increased with coverage, but abundance began to decline once saw palmetto coverage reached approximately 40 %. On the same sites, a similar non-linear pattern in which abundance declines after saw palmetto coverage rises above approximately 35 % has also been seen in northern bobwhite (*Colinus virginianus*; J.A. Martin, D.B. School of Forestry and Natural Resources, University of Georgia, *unpublished data*). My results support the work of others who have suggested that low levels of saw palmetto is advantageous to Bachman's sparrows because it may offer protection from predators (Dean and Vickery 2002), but high coverage is detrimental, likely because of the limiting effect that it can have on other important plant species (Watts 2002). Due to changes in the historical burning regimes, saw palmetto currently covers a much greater proportion of dry prairies than it once did (Yarlett 1965), and results from this study suggest that these changes may play a role in regional population declines.

I did not observe a strong influence of saw palmetto on Bachman's sparrows during the study's second year. Only 1 of the models within the confidence set contained saw palmetto and

the estimates from that model suggested a weak, non-significant effect. This is surprising when considering the degree of strength that saw palmetto had on Bachman's sparrow abundance during the first year. However, the lack of effect of saw palmetto from the hierarchical models during the second year coincided with the inclusion of saw palmetto within the DISTANCE models predicting detection probability. Therefore, I cannot be certain whether the lack of response to saw palmetto was a function of an actual biological reaction or simply an artifact decreased detection rates.

Coverage by bunch-grasses was important to both eastern meadowlarks and Bachman's sparrows. Haggerty (2000) found that Bachman's sparrows preferred dense herbaceous ground cover composed of high proportions of bunch grasses. This was believed to aid in the capture of prey items by providing a good substrate for arthropods near the ground, yet still allowing high visibility and maneuverability. Furthermore, clumps of bunch grasses seem to be preferred for nest sites for sparrows (Haggerty 1995). Likewise, eastern meadowlarks have been shown to prefer areas with high coverage of grasses (Roseberry and Klimstra 1970, Hull 2000). Habitat suitability models suggest that the bunch grasses are extremely important to eastern meadowlarks and are critical for providing nesting habitat (Schroeder and Sousa 1982).

Conclusions

As suggested by Winter et al. (2005) and (2006), I observed a high level of variability in the relationship of birds to patch and habitat variables, across both years and species. Each of the 2 species I investigated have markedly different life histories, therefore their differences in response to habitat variables was expected. However, the variability within species between years was not expected. Few clear patterns emerged across both years for either species; furthermore, no model yielded a statistically significant parameter effect during 2006. In that

year, breeding activity seemed to be suppressed throughout most of the period in which I sampled birds, possibly because of abnormal drought conditions. Throughout the breeding season of 2006, year-to-date rainfall for much of southern Florida was only 50 – 60 % of the long-term average (SFWMD 2007). Some passerine species have been shown to exhibit lower breeding activity and reproductive success during years of extreme drought conditions (Christman 2002), and this seemed to be the case on my dry prairie sites during 2006. Fewer males of all species appeared to be calling; this was particularly true with Bachman's sparrows. Although nearly the same number of points was sampled in both years, in 2006 the total observations of Bachman's sparrows was nearly half that of the previous year. Additionally, a lower proportion of birds were detected aurally. The suppression of breeding activity could have potentially affected my results in two ways: (1) it could have lead to a decrease in territoriality by males, causing them to abandon specific preferences and be observed within more general habitat conditions, or (2) the suppression of calling lowered the observer's ability to detect birds, possibly masking certain patterns of habitat use. The former explanation is supported by the much lower detection rate for Bachman's sparrow that I observed during the second year. Due to the lower breeding activity during the second year, I suspect that the results from that year may not accurately reflect the conditions to which birds normally respond.

Some models in the candidate set for both species included a covariate describing whether sample points had been burned during the previous year. This variable seemed relatively unimportant and was rarely included within the confidence set of models, nor was the effect of the covariate ever statistically significant. Grassland birds have been shown to be extremely dependent on fire to maintain habitat conditions (Brawn et al. 2001), and the dry prairie is considered an extremely pyrogenic ecosystem (United States Fish and Wildlife Service

1999), which made the lack of effect of the BURN variable somewhat surprising. However, I believe that the lack of effect of burning was due to the relatively short fire interval that the BURN variable represented. Most of my field sites were private lands, and managers rarely recorded burning history. Therefore, I was not able to include a covariate explaining a more detailed account of the fire history at each point. Most likely, the species in my study would have responded to the effects of frequent, long-term burning, rather than the short-term effects of a single burn.

Although I was unable to include covariates to account for long-term fire history, I believe that some of the habitat variables that I included likely serve as indicators of the fire history. For instance, on south Florida prairies and flatwoods, saw palmetto coverage can increase dramatically in the absence of frequent burns (Watts 2002). Therefore, points with high palmetto coverage likely had received less frequent burning than points with lower levels of coverage. The same can be said of the mean Robel measurement; it is highly probable that areas with high Robel readings were denser because they were not burned as often. It can therefore be assumed that the short term effect of burning may be unimportant, but many of the conditions to which birds showed a favorable response were representative of the long-term fire history at the survey points.

The effects of patch size differed between years for each species. Previous studies have shown that effects of patch size can be highly variable both spatially (Johnson et al. 2001), and temporally (Winter et al. 2006). The effects of patch size are also likely to vary in relation to the degree of fragmentation and overall matrix of the landscapes in which patches are embedded (Donovan et al. 1997). The landscape context surrounding dry prairie patches differed greatly between my study sites, and due to the relatively few landscapes (study sites) that were contained

within my study, I was unable to measure the influence that landscape may have had on bird response to patch size. Although my data do suggest that some members of the avian community of the dry prairie may be influenced by patch size, a more critical, long-term evaluation of the patch size phenomenon, which is able to incorporate other landscape-level information, is needed before definitive inference can be made regarding the effects of patch size on members of the dry prairie bird community.

My use of abundance and occurrence to gauge habitat preference could have possibly introduced bias regarding the importance of vegetation and patch variables. Van Horne (1983) warned against using abundance and density as measures of habitat quality. She concluded that demographic parameters such as reproduction and survival should be included in any assessment of habitat quality, because without that information, inference about the causes of abundance could be misleading. Vickery et al. (1992) illustrated Van Horne's (1983) point by showing that density was not correlated with breeding success for three grassland sparrows. However, subsequent authors have argued that density and abundance can be meaningful measures of habitat quality in many cases (Bock and Jones 2004). Regardless, my lack of data on the relationship between habitat conditions and demographics could have potentially biased my interpretation of the importance of those parameters for the species I studied.

MANAGEMENT IMPLICATIONS

Results from my study suggest that habitat conditions for some savannah species on Florida dry prairie can be improved by maintaining a low-density herbaceous layer, composed of low to moderate levels of saw palmetto, and high proportions of bunch grasses. Although I did not specifically address the effect that frequent fire can have on habitat conditions, previous

research on dry prairie has suggested that the habitat conditions which I found to be optimal can be promoted and maintained through short fire-return intervals (Fitzgerald and Tanner 1992, Watts 2002). Historic burn intervals on Florida dry prairie ranged from 1 – 3 years, though current frequencies for land management are slightly longer, ranging from 2 – 4 years (United States Fish and Wildlife Service 1999, Main and Richardson 2002). Within a highly pyrogenic environment such as dry prairie, a 1-year difference in fire-return can have dramatic effects on vegetation over long time periods (Watts 2002). The relative intolerance that I observed for some vegetative characteristics that result from infrequent burning suggests that land managers within the dry prairie should try to mimic natural fire intervals and apply prescribed burns at 1 – 3 year intervals. Due to the difficulty of managing large properties, this frequency of burning may best be accomplished by targeting a biennial burning regime. In the event that environmental or logistical constraints do not allow adequate areas to be burned on the 2-year rotation, burning the following year will still fall within the natural 1 - 3 year interval.

Although the effects of patch size deserves further investigation, inference gained from this study does suggest that efforts to maintain large prairie patches and increase connectivity between both dry prairie patches and adjacent flatwoods should be promoted within south Florida. Loss and fragmentation of existing dry prairie remain the greatest threats to the ecosystem's conservation (Kautz 1998, United States Fish and Wildlife Service 1999), and other research has suggested that these may be major hurdles in maintaining viable populations of some species of grassland birds (Shriver and Vickery 1999). My research supports the conclusions of Perkins and Vickery (2007) regarding the eastern meadowlark, suggesting that fragmentation may play a role in regional declines of the species.

Confounding factors in my analysis prevented definitive conclusions regarding the role of patch size on Bachman's sparrows, but did suggest that dispersal ability may be important in maintaining populations within the region. Many of the dry prairie patches that I surveyed were embedded within landscapes dominated by exotic grass pastures, and for Bachman's sparrows this landcover type may be inhospitable and limit dispersal between prairie fragments. Further research incorporating other landscape metrics, specifically the potential for dispersal, should be conducted on Bachman's sparrows. Nonetheless, in lieu of more definitive evidence, I suggest that promoting connectivity between remnant dry prairie fragments as well as nearby pine flatwoods may aid in conserving Bachman's sparrows within south Florida. In addition, when management for Bachman's sparrows is an objective, the value of small patches should not be overlooked. These areas should be managed with the same intensity as larger patches, particularly if they occur adjacent to possible population sources.

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Appendix 2A. Abundance of all species observed on point counts conducted on the dry prairie south-central Florida, 2005 – 2006.

Species	Scientific name	2005	2006	Total
Common yellowthroat	<i>Geothlypis trichas</i>	557	217	774
Eastern towhee	<i>Pipilo erythrophthalmus</i>	422	165	587
Red-winged blackbird	<i>Agelaius phoeniceus</i>	175	184	359
Eastern meadowlark	<i>Sturnella magna</i>	167	144	311
Bachman's sparrow	<i>Aimophila aestivalis</i>	189	104	293
Northern bobwhite	<i>Colinus virginianus</i>	34	17	51
White-eyed vireo	<i>Vireo griseus</i>	43	4	47
Mourning dove	<i>Zenaida macroura</i>	32	14	46
Red-bellied woodpecker	<i>Melanerpes carolinus</i>	31	11	42
Northern mockingbird	<i>Mimus polyglottos</i>	26	9	35
Carolina wren	<i>Thryothorus ludovicianus</i>	23	9	32
Northern cardinal	<i>Cardinalis cardinalis</i>	25	6	31
Common grackle	<i>Quiscalus quisqualis</i>	27	0	27
Unknown	----	1	26	27
Common ground dove	<i>Columbina passerina</i>	13	7	20
American crow	<i>Corvus brachyrhynchos</i>	4	6	10
Brown-headed cowbird	<i>Molothrus ater</i>	5	4	9
Boat-tailed grackle	<i>Quiscalus major</i>	6	1	7
Great-crested flycatcher	<i>Myiarchus crinitus</i>	6	1	7
Loggerhead shrike	<i>Lanius ludovicianus</i>	7	0	7
Unknown grackle	----	3	4	7
Eastern bluebird	<i>Sialia sialis</i>	3	3	6
Pine warbler	<i>Dendroica pinus</i>	4	0	4
Red-shouldered hawk	<i>Buteo lineatus</i>	1	3	4
Unknown woodpecker	----	3	1	4
Blue jay	<i>Cyanocitta cristata</i>	2	1	3
Brown thrasher	<i>Toxostoma rufum</i>	3	0	3
Florida scrub jay	<i>Aphelocoma coerulescens</i>	0	3	3
Grey catbird	<i>Dumetella carolinensis</i>	3	0	3
Pileated woodpecker	<i>Dryocopus pileatus</i>	2	1	3
Red-headed woodpecker	<i>Melanerpes erythrocephalus</i>	1	2	3
Black vulture	<i>Coragyps atratus</i>	2	0	2
Downy woodpecker	<i>Picoides pubescens</i>	1	1	2
Eastern kingbird	<i>Tyrannus tyrannus</i>	0	2	2
Osprey	<i>Pandion haliaetus</i>	0	2	2
Sandhill Crane	<i>Grus canadensis</i>	1	1	2
Tufted titmouse	<i>Baeolophus bicolor</i>	2	0	2
Black-bellied whistling duck	<i>Dendrocygna autumnalis</i>	1	0	1
Blue-gray gnatcatcher	<i>Poliophtila caerulea</i>	0	1	1
Fish Crow	<i>Corvus ossifragus</i>	1	0	1
Hairy woodpecker	<i>Picoides villosus</i>	0	1	1
Northern parula	<i>Parula americana</i>	1	0	1
Total		1827	955	2782

Appendix 2B. Mean and standard deviation for covariates used in models to predict eastern meadowlark (*Sturnella magna*) occurrence and Bachman's sparrow (*Aimophila aestivalis*) abundance on south Florida dry prairie, 2005 – 2006.

	2005		2006	
	Mean	SD	Mean	SD
Patch size (ha)	249.12	333.70	155.89	287.64
Saw palmetto coverage	0.3440	0.2048	0.3275	0.2173
Bunch grass coverage	0.5745	0.2432	0.6518	0.2559
Vegetative litter coverage	0.3182	0.2331	0.5634	0.2823
Mean Robel measurement	4.01	2.21	2.65	1.77

CHAPTER 3
HABITAT USE BY TWO MIGRANT GRASSLAND PASSERINES OVER-WINTERING
ON SOUTH FLORIDA'S DRY PRAIRIE

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CHAPTER 3

HABITAT USE BY TWO MIGRANT GRASSLAND PASSERINES OVER-WINTERING ON SOUTH FLORIDA'S DRY PRAIRIE

INTRODUCTION

Grassland birds have witnessed population declines that surpass those exhibited by any other suite of avian species in North America (Samson and Knopf 1994, Peterjohn and Sauer 1999, Brawn et al. 2001, Brennan and Kuvlesky 2005). Research has tended to focus on the breeding grounds of most species, where issues such as overgrazing (Saab et al. 1995), suppression of fire (Johnson 1997), encroachment by woody stemmed plants and exotic grasses (Grant et al. 2004), habitat loss (Samson and Knopf 1994), fragmentation (Herkert et al. 2003), and brood parasitism by brown-headed cowbirds (*Molothrus ater*; Johnson and Temple 1990) seem to be fueling declines. However, little effort has gone into understanding the wintering ecology of most grassland birds, and consequently this aspect of the life cycle of nearly all grassland species remains poorly understood (Herkert and Knopf 1998, Vickery et al. 1999, Vickery and Herkert 2001), although demographics during this period may have serious implications for overall population viability (Marra et al. 1998, Basili and Temple 1999).

Many of North America's grassland birds are short-distance migrants that winter in the Southeastern United States. Within this region, the dry prairie of south-central Florida possibly provides the largest remaining area of contiguous grassland habitat. Like many grassland systems across North America, the dry prairie has witnessed severe habitat loss and degradation over the last century. These changes have resulted from anthropogenic conversion of dry prairie

to other land-uses, such as exotic grass pastures, citrus groves, and residential developments. Simultaneously, shifts in the natural plant community have resulted from alterations to the timing and frequency of natural fire regimes (United States Fish and Wildlife Service 1999). The effect that these changes have had on migrant grassland birds that rely on the dry prairie is unknown, largely because few quantitative studies have described the habitat preferences of grassland birds over-wintering within the region.

Increased insight into the wintering ecology of grassland passerines has been put forth by several authors (Herkert and Knopf 1998, Vickery and Herkert 2001, Brennan and Kuvlesky 2005) as a major research objective needed to advance grassland bird conservation, particularly on southern Florida's dry prairie (United States Fish and Wildlife Service 1999). Therefore, I investigated the winter ecology of 2 migrant species of high conservation priority, the grasshopper sparrow (*Ammodramus savannarum pratensis*) and the sedge wren (*Cistothorus platensis*). I believed that focusing on these 2 species, which are both grassland-obligates yet exhibit distinctly different life-history strategies, would yield insight into how the entire suite of migrant grassland passerines may be responding to environmental changes on the dry prairie.

My objectives were to explain how variation in the structure of the dry prairie's vegetation, fire history, and distance to landscape variables influenced habitat usage by the 2 selected species. These characteristics have been shown to influence habitat use by grassland birds in other regions, yet knowledge about their effects on grassland species in the dry prairie is lacking. Land managers in the region rarely incorporate the needs of migrant grassland birds into management plans, possibly because of the uncertainty surrounding their requirements and response to management. Increased understanding of the relationship between habitat

management and use by this guild could yield more informed land management decisions that provide prairie conditions that are more suitable for grassland birds.

METHODS

Study Site

This study was conducted within the historic range of the dry prairie in Desoto, Manatee, and Sarasota Counties, Florida (Figure 3.1), from January - February of 2006 - 2007. The dry prairie is a treeless, pyrogenic landscape, dominated by wiregrass (*Aristida beyrichiana*), scattered saw palmetto (*Serenoa repens*), and low-growing runner oak (*Quercus minima*) (United States Fish and Wildlife Service 1999). During the winter months, the region's climate is mild and extremely dry. During the months of December through February, there is an average of 16 cm of rainfall, and mean daily temperature is approximately 17.7°C.

Research was conducted on 4 study sites; 3 of which were privately-owned ranches managed for cattle production and landowner recreation, and the fourth was a state park managed to promote biodiversity and maintain natural ecosystems. Despite the differences in objectives, land management was relatively similar across the dry prairie fragments on all 3 sites, and involved a combination of prescribed burning and roller chopping. The only major difference in management activities between sites was periodic or continuous grazing by cattle on the private ranches and total exclusion of cattle grazing at the state park.

Bird Surveys

During winter, most grassland passerines remain hidden below the vegetation's canopy, which renders passive sampling techniques ineffective. To overcome this problem, I conducted "flush" transects modified from the methodology suggested by Fletcher et al. (2000). Using this

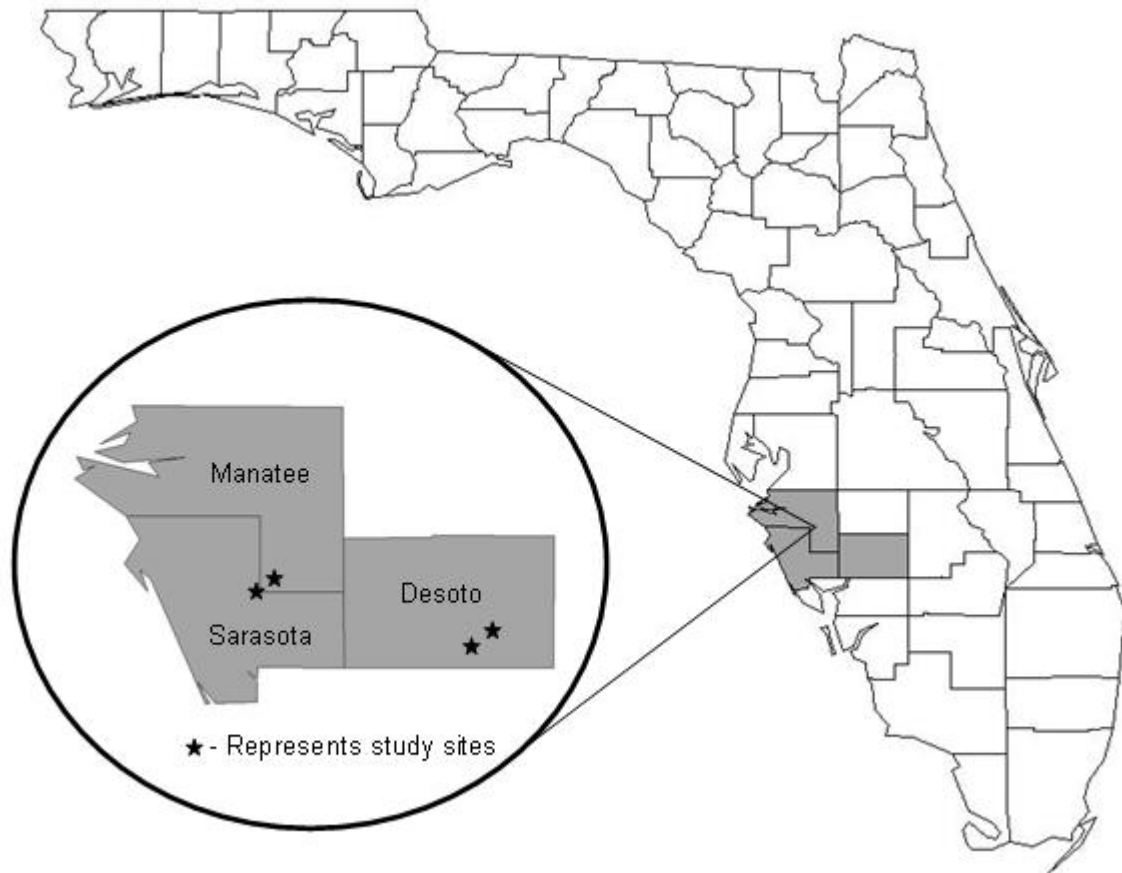


Figure 3.1. Study site locations in Desoto, Manatee, and Sarasota counties, Florida.

technique, 2 observers dragged a 25 m rope fitted with noise makers (2 L plastic bottles filled with rocks) across the top layer of the prairie vegetation. A third observer was positioned near the rope's center, roughly halfway between the other 2 observers. Flushed birds were identified to species while in flight through observation of distinctive flight patterns, call notes, or coloration. Others have had success sampling inconspicuous species in prairie environments using similarly proactive methods (Higgins et al. 1969).

Within each study site, dry prairie patches were identified by on-site visitation or through inspection of satellite imagery. The location and orientation of each transect within patches was then randomly selected. Each patch contained multiple transects, ranging from 2 – 10 depending upon patch size. In an effort to improve species identification, the direction of travel along transects was selected to prevent observers from facing directly into the sun. Transect dimensions were 25 X 100 m, encompassing an area of approximately 0.25 ha, and were covered in a single pass. The mean duration for transects (\pm SD) was 3.7 ± 1.9 minutes. Transects were spaced ≥ 200 m apart to minimize the probability of double counting birds. Surveys began at sunrise and continued until 1300, and were only conducted during favorable conditions with no precipitation and winds < 10 km/hour (Robbins 1981).

If flushed birds landed on a transect portion that had not been covered, observers noted their location to prevent repeated counts of individuals. Additionally, the re-flush behavior of these known individuals was recorded in an attempt to estimate their flushing probability (i.e., detection rate; but see *Data Analysis*). A total of 148 transects were established, and each was surveyed twice within years. The occurrence of grasshopper sparrows and sedge wrens in either trial were treated as the response variables for analysis. If transects were severely altered by land

management activities (e.g., burned) between subsequent bird surveys, data from that transect was not used for analysis.

Vegetation Surveys

Vegetation measurements were taken using a combination of the methods outlined in Herrick et al. (2005) and visual obstruction methodology proposed by Robel et al. (1970). The vegetative community was divided into 8 major functional groups: saw palmetto, bunch grass, broad-leafed woody, forbs, sod-forming grass, vines, legumes, and standing-dead woody. The percent of coverage by each major vegetative functional group was then assessed using 2, 25 m lines which were extended along the length of each bird transect; the first at the starting point, and the second at approximately the 75 m mark. At half-meter intervals along each line, a metal pin-flag was dropped and all functional classes touching the pin were recorded. I also noted whether the pin-flag struck leaf litter or bare soil. A measurement of vegetation height was taken every 2.5 m. At the 10 and 20 m interval of the vegetation transect, the density of the vegetation was estimated using methodology suggested by Robel et al. (1970). For data analysis, I averaged the percent coverage of each functional class, height, litter, bare soil, and Robel measurements across the 2 vegetation transects in order to obtain a vegetation mean for the bird transect.

Data Analysis

I used logistic regression to investigate factors that influenced the occurrence of grasshopper sparrows and sedge wrens. Many patches of dry prairie were significantly altered by land management between years, therefore I analyzed years separately. Models used in the regression analysis were constructed based on biological hypotheses, and contained combinations of vegetation variables, amount of litter, and distance of transects to patch edges. The length of time since burning was also included as a predictor and was broken into 3 intervals

based on time since burning: 1-year, 2-year, and 3 or more years. In 2007, there were no observations of sedge wrens on transects with 1-year burns, so for the sedge wren model in 2007, I created a binary variable by grouping observations in 1- and 2-year-old burns together. Explanations and biological interpretations of all model covariates can be found in Table 3.1.

I suspected *a priori* that bird response to certain vegetation variables was possibly non-linear; therefore I ran base logistic models with each vegetation variable alone and visually examined a plot of the residuals for signs of a quadratic response. I calculated correlation coefficients between all possible pairs of data. If two variables were strongly correlated ($r^2 \geq 0.30$) I chose only the variable that I believed to be most biologically meaningful for inclusion in regression models.

I initially fit a global (i.e., included all predictors) logistic regression model for each species and examined plots of the residuals grouped by dry prairie patch. The plots revealed a significant amount of spatial autocorrelation among dry prairie patches, suggesting that surveys within patches were more alike than surveys between patches. To account for this lack of independence, I analyzed the data by acknowledging that transects were nested within patches, and allowed the intercept and slopes of the models to vary randomly by patch. This is known as a mixed model because it contains both random and fixed effects. For my study, the random effect was the individual effect of each patch, which was unable to be accounted for using regular logistic regression. Fixed effects included the mean effect of transect-specific characteristics, such as vegetative coverage, across all patches. All mixed models were fit using the NLMIXED procedure of SAS software (SAS Institute 2003).

To compare the relative plausibility of models, I used the information-theoretic approach suggested by Akaike (1973) and later supported by Burnam and Anderson (2002). This

Table 3.1. Explanation of variables used in logistic regression models to predict occupancy of transects by grasshopper sparrow (*Ammodramus savannarum pratensis*) and sedge wrens (*Cistothorus platensis*) over-wintering in the dry prairie of southern Florida, 2006 – 2007.

Variable	Explanation
BURN	Describes the lapse of time between the last prescribed burn and the bird survey. Within models, three intervals were acknowledged: 1, 2, and ≥ 3 -year burns. For ease of comparison, models related the odds of occupancy in 1 and 2 year burns against odds for ≥ 3 -year burns.
PALM	Describes the estimate of the percent of coverage by saw palmetto (<i>Serenoa repens</i>) along the bird transect. Measured by the number times saw palmetto was contacted at least once by pins dropped at half-meter intervals along the 2, 25m vegetation survey transects.
BUGR	Describes the estimate of the percent of coverage by bunch grasses along the bird transect. A few examples of bunch grasses on south Florida dry prairies include: wiregrass (<i>Aristida beyrichiana</i>), bluestems (<i>Andropogon</i> spp.), and panic grasses (<i>Dichanthelium</i> spp.). Measured by the number times any type of bunch grass was contacted at least once by pins dropped at half-meter intervals along the 2, 25m vegetation survey transects.
BLWD	Describes the estimate of the percent of coverage by broad-leafed, woody-stemmed shrubs along the bird transect. Common examples of broad-leafed, woody-stemmed shrubs on south Florida dry prairies are wax myrtle (<i>Myrica cerifera</i>), runner oak (<i>Quercus minima</i>), and gallberry (<i>Ilex glabra</i>). Measured by the number times any type of species from this group was contacted at least once by pins dropped at half-meter intervals along the 2, 25m vegetation survey transects.
FORB	Describes the estimate of the percent of coverage by forbs along the bird transect. Common examples of forbs on south Florida dry prairie include varieties of goldenrod (<i>Euthamia</i> spp), bachelor's button (<i>Polygala</i> spp.), and yellow-eyed grass (<i>Xyris</i> spp.). Measured by the number times any type of species from this group was contacted at least once by pins dropped at half-meter intervals along the 2, 25m vegetation survey transects.
LIT	Describes the estimate of the percentage of the ground covered by vegetative litter of any kind. Measured by the number times that pins dropped at half-meter intervals along the 2, 25m vegetation survey transects were intercepted by vegetative litter before coming in contact with the ground.
EDGE	Describes the length in meters that the center of the bird transect is from the nearest patch edge. Abrupt changes in landcover or roads and trails > 10m in width were considered edges.
WET	Describes the length in meters that the center of the bird transect is from the nearest permanent or ephemeral wetland.

approach is commonly known as Akaike's Information Criterion (AIC) and is derived from likelihood theory. I calculated AIC values with the small sample size adjustment (AIC_c; Burnham and Anderson 2002) for each predictor model; models with lower AIC_c values are considered to be better approximations than higher valued models.

To further facilitate comparisons, a ΔAIC_c value was then obtained by subtracting the AIC_c value of each model from that of the model with the lowest AIC_c. This value was then normalized to obtain a relative weight, or Akaike weight, for each model via the equation:

$$W_i = \exp(-0.5 \cdot \Delta_i) / \sum_{r=1}^R \exp(-0.5 \cdot \Delta_r)$$

where Δ_i is the ΔAIC_c value for the i^{th} model in the set of r candidate models. This value, w_i is interpreted as being the weight of evidence for, or probability in favor of, model i , given the data and the candidate set of models. Goodness-of-fit was assessed for the global model and each model in the candidate set through visually examining plots of the residuals to check for lack-of-fit and obvious outliers.

Though some inference can be gained from model selection, I was most interested in the effect individual covariates had on the occurrence of grassland passerines. The effect size of covariates can differ among candidate models, but use of model-averaging to incorporate this uncertainty into parameter estimates is not suitable with mixed models (J.T. Peterson, D.B. Warnell School of Forest Resources, University of Georgia, Athens, GA 30605, *personal communication*). Therefore, for inference I report the parameter estimates from all models contained within the confidence set of models. I define the confidence set of models as those models with an Akaike weight of greater than 10 % of the weight of the most parsimonious model (Burnham and Anderson 2002). To ease interpretation, I calculated scaled odds ratios for each parameter by exponentiation of the product of the parameter estimate and a scaling value which I deemed biologically relevant.

Prior to building predictor models, I investigated the efficacy of the “flush” transect technique by modeling the probability of reflush by known individuals of each species using closed capture models in program MARK (White and Burnham 1999). If flushed birds landed on a transect portion that had not been covered, the reflushing behavior of that individual was recorded. Reflushes were treated as recaptures in the Huggin’s closed capture model. I believed that this recapture rate was analogous to detection probability under the following assumptions: (1) behavior by individual birds was comparable between initial and subsequent flushes, and (2) the probability of observers detecting a flush was equal between initial flushes of unknown individuals and reflushes of known individuals.

Often individuals were reflushed multiple times over a transect’s length. This was particularly true for sedge wrens because of their weak flying ability. I believed that the assumption of comparable behavior between initial flush and subsequent reflushes was not reasonable as an individual became increasingly disturbed. Therefore, the detection probability was calculated using only the first reflush after each individual bird’s initial encounter. I believed that assumption 2 was realistic considering that 3 observers, stationed at both ends and the rope’s center, were spaced roughly 12 m apart and were focused on the transect at all times. I believed that employing 3 observers in such close proximity ensured that if birds did flush, they were not missed.

I created models in program MARK to predict the detection probability for both grasshopper sparrows and sedge wrens. Models were parameterized to include the effects of species, year, and all vegetation covariates that I suspected might influence flushing rates. Vegetation covariates included the coverage of saw palmetto, bunch grass, broad-leafed woody vegetation, and the interactions between each pair of variables. The plausibility of each model

was determined using AIC_c and the relative belief in each model was evaluated by their respective AIC_c weights. I calculated a model-averaged parameter estimate and standard error for detection probability across all models within the confidence set (Burnham and Anderson 2002).

The model-averaged parameter estimate for detection probability was high ($\bar{x} = 0.8958$, $SE = 0.0346$). Therefore, I felt that more robust methods to account for detectability in presence / absence surveys were not warranted, and proceeded to model occurrence of each species using the raw, unadjusted data. The output and detection estimates from program MARK can be found in Table 3.2.

RESULTS

I surveyed 140 transects within 25 dry prairie patches, and 126 transects within 24 dry prairie patches, respectively, during the winters of 2006 and 2007. Twenty-three species were observed over both years, and nearly half were migrants that only reside on dry prairie during the winter (a complete listing of all species observed can be found in Appendix 3A). Grasshopper sparrows were the most abundant species observed in both years, and were observed on 50 and 34 % of transects during 2006 and 2007, respectively. Sedge wrens were observed on 15 % of transects in 2006 and 29 % in 2007. In the following sections, model results are reported independently for both grasshopper sparrows and sedge wrens.

Grasshopper sparrow

In both years, the models containing only BURN were ranked the highest, although the strength of support for those models did vary between years (2006: Akaike weight = 0.5585, 2007: Akaike weight = 0.2382; Table 3.3). In 2006, the burn only model was over 0.5585 /

Table 3.2. Recapture probability (i.e., detection rate), standard error, 95 % confidence interval, number of parameters (K), and Akaike Information Criterion with small sample size adjustment (AIC_c) results from models used in program MARK to investigated efficacy of flush transects on southern Florida’s dry prairie, 2006 – 2007.

Model	Detection (SE)	95% Confidence Interval		K	AICc	ΔAICc	wi
		Lower	Upper				
c + BLWD	0.8863 (0.0208)	0.8385	0.9213	4	190.9552	0.0000	0.3076
c constant by year & species	0.8775 (0.0206)	0.8310	0.9124	2	192.2230	1.2678	0.1632
c + SERE+ BLWD	0.8868 (0.0209)	0.8389	0.9218	5	192.7388	1.7836	0.1261
c + SERE+ height	0.8863 (0.0209)	0.8385	0.9213	5	192.9781	2.0229	0.1119
c + BLWD + height + BLWD*height	0.8892 (0.0209)	0.8411	0.9241	6	193.1625	2.2073	0.1020
c + SERE + BLWD+ SERE*BLWD	0.8912 (0.0214)	0.8415	0.9267	6	193.5878	2.6326	0.0825
c + BUGR	0.8778 (0.0206)	0.8313	0.9128	4	196.0478	5.0926	0.0241
c + SERE	0.8776 (0.0206)	0.8311	0.9126	4	196.1945	5.2393	0.0224
c + height	0.8771 (0.0206)	0.8306	0.9122	4	196.2000	5.2448	0.0223
c different by year	0.8775 (0.0206)	0.8310	0.9125	4	196.2623	5.3071	0.0217
c + SERE + height	0.8778 (0.0206)	0.8313	0.9128	5	198.0699	7.1147	0.0088
c + SERE + height + SERE*height	0.8791 (0.0206)	0.8324	0.9141	6	199.2990	8.3438	0.0047
c different by species	mean = 0.8856 (0.0589)	mean = 0.7021	mean = 0.9528	9	200.4492	9.4940	0.0027

Table 3.3. Predictors, number of parameters (K), Akaike's Information Criterion with small-sample size adjustment (AIC_c), ΔAIC_c, and Akaike weights (w_i) for the set of candidate models used to predict the occupancy of grasshopper sparrows (*Ammodramus savannarum pratensis*) over-wintering on south Florida dry prairie, 2006 – 2007.

Model	K	AIC _c	ΔAIC _c	w _i
2006				
BURN	4	179.52	0.0000	0.5585
BURN + PALM + BUGR + BLWD + FORB + LIT + EDGE (<i>Global</i>)	10	181.08	1.5561	0.2565
BURN + PALM + BUGR + BLWD + FORB + LIT	9	181.87	2.3518	0.1723
PALM	3	189.47	9.9525	0.0039
PALM + BUGR + BLWD + FORB + LIT	7	189.81	10.2877	0.0033
LIT	3	190.77	11.2446	0.0020
PALM + BLWD	4	191.37	11.8520	0.0015
PALM + BLWD + PALM*BLWD	5	192.23	12.7099	0.0010
FORB	3	193.63	14.1139	0.0005
EDGE	3	194.02	14.5038	0.0004
BUGR	3	195.51	15.9881	0.0002
2007				
BURN	4	165.37	0.0000	0.2382
PALM + BLWD	4	165.98	0.6059	0.1759
FORB	3	166.09	0.7185	0.1663
LIT	3	167.04	1.6698	0.1034
PALM + BLWD + PALM*BLWD	5	167.74	2.3705	0.0728
PALM	3	167.84	2.4630	0.0695
EDGE	3	167.92	2.5479	0.0666
BUGR	3	167.95	2.5747	0.0657
BURN + PALM + BUGR + BLWD + FORB + LIT	9	170.29	4.9170	0.0204
PALM + BUGR + BLWD + FORB + LIT	7	170.98	5.6104	0.0144
BURN + PALM + BUGR + BLWD + FORB + LIT + EDGE (<i>Global</i>)	10	172.52	7.1438	0.0067

$0.2565 = 2.1$ times more likely than the second best-fitting model, and $0.5584 / 0.1723 = 3.2$ times more likely than the third ranked model (Table 3.3).

During 2007, there was less certainty surrounding support for the model containing only BURN, and no single model garnered a majority of the weight. For instance, the amount of support in favor of the models for PALM + BLWD (Akaike weight = 0.1759), FORB (Akaike weight = 0.1663), and LIT (Akaike weight = 0.1034) was 73, 70, and 43 %, respectively, of that of the BURN model (Table 3.3).

Although there were differences in the relative strength of the BURN model between years, the parameter estimates were nearly identical in both years (Table 3.4). In both years, the models that contained the burn parameter showed a significant effect of 1-year burns. The probability of encountering a grasshopper sparrow was much higher in 1-year burns than in areas that had not been burned in 2 or more years (Figure 3.2).

No other parameters shared consistent effects between years. In 2006, 2 models in the confidence set contained the covariate for LIT, and in both models the parameter had a significant negative effect. In 2007, the FORB covariate in the third ranking model was statistically significant and suggested a slightly negative effect of FORB, which contrasted with the slightly positive, yet highly variable effect of FORB in the 2006 models (Table 3.4).

Sedge wren

In 2006, the model containing only LIT was best-fitting (Akaike weight = 0.3638) and was $0.3638 / 0.2103 = 1.72$ times more likely than the second ranked model containing only BUGR (Akaike weight = 0.2103), and > 2 times more likely than the third ranked model (PALM + BUGR + BLWD + FORB + LIT, Akaike weight = 0.1586; Table 3.5). Percent coverage by LIT had a negative effect on sedge wren occupancy and was relatively strong for all models in

Table 3.4. Estimates, standard error, 95 % confidence interval, standardized coefficients, biologically relevant unit scaler, and scaled odds ratio for fixed parameters contained within the confidence set of models used to predict occurrence of wintering grasshopper sparrows (*Ammodramus savannarum pratensis*) on transects within southern Florida's dry prairie, 2006 - 2007. Scaled odds were calculated by exponentiation of the product of the parameter estimate and the unit scaler, and can be interpreted as the number of times more / less likely occurrence was after an increase / decrease in the parameter equivalent to the value of the unit scaler.

Parameter	Estimate	SE	95% Confidence Interval		Standardized Coefficient	Unit Scaler	Scaled Odds
			Lower	Upper			
2006							
BURN only							
Intercept	-1.0678	0.3495	-1.7528	-0.3828	----	----	----
BURN							
One Year	1.8945	0.4366	1.0388	2.7502	6.65	----	6.65
Two Year	0.6325	0.5214	-0.3894	1.6544	1.88	----	1.88
≥ Three Years	----	----	----	----	----	----	----
EDGE + BURN + PALM + BUGR + BLWD + FORB + LIT (<i>Global</i>)							
Intercept	0.9367	1.3108	-1.6325	3.5059	----	----	----
EDGE	0.0021	0.0012	-0.0002	0.0044	1.00	100	1.23
BURN							
One Year	1.6646	0.4909	0.7024	2.6268	5.28	----	5.28
Two Year	0.1049	0.5905	-1.0525	1.2623	1.11	----	1.11
≥ Three Years	----	----	----	----	----	----	----
PALM	-1.8787	1.1630	-4.1582	0.4008	0.15	0.1	0.83
BUGR	-1.4167	1.0940	-3.5609	0.7275	0.24	0.1	0.87
BLWD	-0.2079	2.1612	-4.4439	4.0281	0.81	0.1	0.98
FORB	3.2584	3.4081	-3.4215	9.9383	26.01	0.025	1.08
LIT	-1.8356	0.9790	-3.7544	0.0832	0.16	0.25	0.63

Tables 3.4 continued.

Parameter	Estimate	SE	95% Confidence Interval		Standardized Coefficient	Unit Scaler	Scaled Odds
			Lower	Upper			
BURN + PALM + BUGR + BLWD + FORB + LIT							
Intercept	1.2472	1.3406	-1.3804	3.8748	----	----	----
BURN							
One Year	1.6123	0.5171	0.5988	2.6258	5.01	----	5.01
Two Year	0.2164	0.6296	-1.0176	1.4504	1.24	----	1.24
≥ Three Years	----	----	----	----	----	----	----
PALM	-1.8616	1.2793	-4.3690	0.6458	0.16	0.1	0.83
BUGR	-1.1571	1.0884	-3.2904	0.9762	0.31	0.1	0.89
BLWD	0.2323	2.1962	-4.0723	4.5369	1.26	0.1	1.02
FORB	3.5980	3.4971	-3.2563	10.4523	36.53	0.025	1.09
LIT	-2.0509	1.0299	-4.0695	-0.0323	0.13	0.25	0.60
2007							
BURN only							
Intercept	-0.9029	0.3061	-1.5029	-0.3029	----	----	----
BURN							
One Year	1.8192	0.8909	0.0730	3.5654	6.17	----	6.17
Two Year	0.2542	0.3999	-0.5296	1.0380	1.29	----	1.29
≥ Three Years	----	----	----	----	----	----	----
PALM + BLWD							
Intercept	-1.6642	0.6934	-3.0233	-0.3051	----	----	----
PALM	0.6618	1.0416	-1.3797	2.7033	1.94	0.1	1.07
BLWD	2.6524	1.3510	0.0044	5.3004	14.19	0.1	1.30
FORB only							
Intercept	-1.6642	0.6934	-3.0233	-0.3051	----	----	----
FORB	-4.7419	0.3360	-5.4005	-4.0833	0.01	0.025	0.89

Tables 3.4 continued.

Parameter	Estimate	SE	95% Confidence Interval		Standardized Coefficient	Unit Scaler	Scaled Odds
			Lower	Upper			
LIT only							
Intercept	-1.1751	0.6488	-2.4467	0.0965	----	----	----
LIT	0.9771	1.1084	-1.1954	3.1496	2.66	0.25	1.28
PALM + BLWD + PALM*BLWD							
Intercept	-2.2069	1.1356	-4.432676	0.018876	----	----	----
PALM	1.7277	2.0080	-2.2080	5.6634	5.6277	0.10	1.1886
BLWD	4.6696	3.5593	-2.3066	11.6458	106.6551	0.10	1.5951
PALM*BLWD	-4.1113	6.5818	-17.0116	8.7890	----	----	----
PALM only							
Intercept	-0.8187	0.5186	-1.8352	0.1978	----	----	----
PALM	0.3343	0.9989	-1.6235	2.2921	1.40	0.1	1.03
EDGE only							
Intercept	-0.6977	0.3060	-1.2975	-0.0979	----	----	----
EDGE	0.0002	0.0011	-0.0020	0.0024	1.00	100	1.02
BUGR only							
Intercept	-0.6753	0.6911	-2.0299	0.6793	----	----	----
BUGR	0.0260	0.9828	-1.9003	1.9523	1.03	0.1	1.00

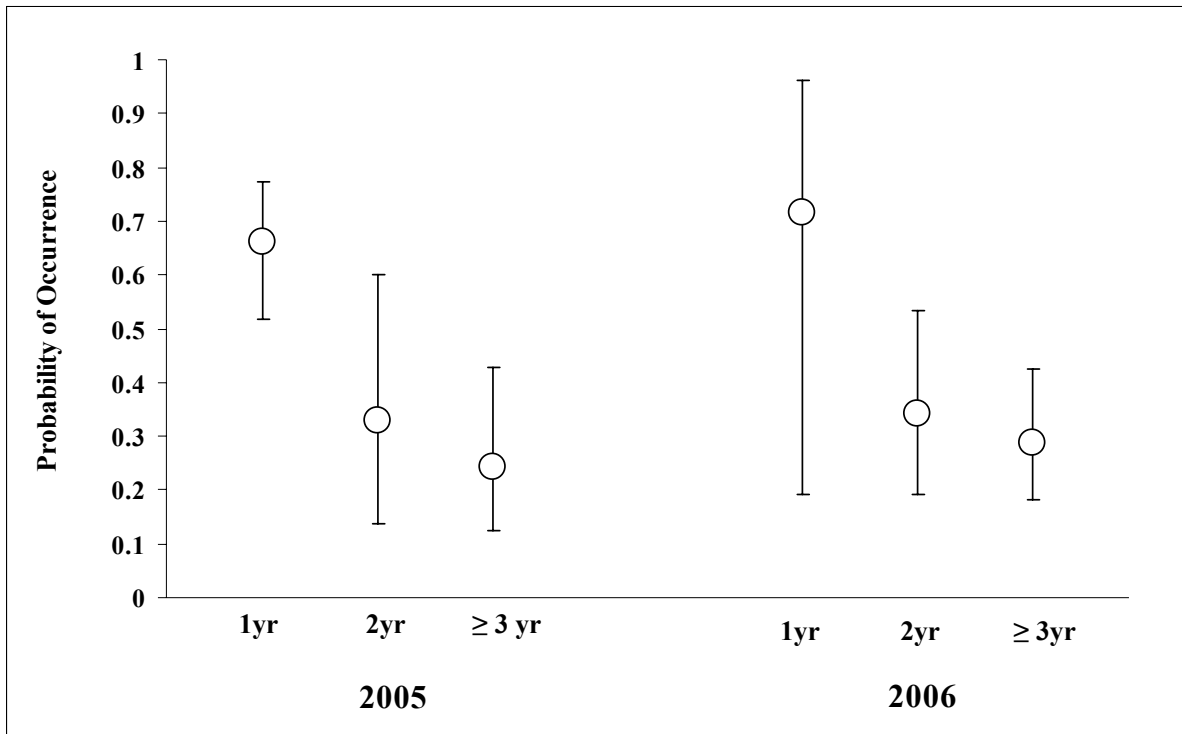


Figure 3.2. Mean probability of occurrence \pm 95 % confidence intervals of over-wintering grasshopper sparrows (*Ammodramus savannarum pratensis*) on transects that had been burned 1, 2, and \geq 3 years prior to survey within the dry prairie of southern Florida, 2006 - 2007. The mean probability was calculated by averaging the estimated probabilities for each model within the confidence set that contained the BURN parameter. For models that contained multiple covariates, all covariates except BURN were held at their mean.

Table 3.5. Predictors, number of parameters (K), Akaike's Information Criterion with small-sample size adjustment (AIC_c), ΔAIC_c , and Akaike weights (w_i) for the set of candidate models used to predict the occupancy of sedge wrens (*Cistothorus platensis*) over-wintering on south Florida dry prairie, 2006 – 2007.

Model	K	AIC_c	ΔAIC_c	w_i
2006				
LIT	3	115.74	0.0000	0.3638
BUGR	3	116.83	1.0957	0.2103
PALM + BUGR + BLWD + FORB + LIT	7	117.40	1.6608	0.1586
EDGE	3	118.81	3.0750	0.0782
BURN + PALM + BUGR + BLWD + FORB + LIT + EDGE + WET (<i>Global</i>)	11	118.87	3.1275	0.0762
WET	3	119.42	3.6840	0.0577
BURN + PALM + BUGR + BLWD + FORB + LIT	9	121.21	5.4749	0.0235
FORB	3	122.12	6.3841	0.0149
PALM	3	123.46	7.7205	0.0077
BURN	4	124.13	8.3956	0.0055
PALM + BLWD	4	125.54	9.8034	0.0027
PALM + BLWD + PALM*BLWD	5	127.61	11.8710	0.0010
2007				
BURN	3	152.76	0.0000	0.4328
LIT	3	154.72	1.9659	0.1620
EDGE	3	155.70	2.9460	0.0992
BUGR	3	155.74	2.9787	0.0976
WET	3	156.50	3.7427	0.0666
FORB	3	156.86	4.1058	0.0556
PALM	3	156.93	4.1731	0.0537
PALM + BLWD	4	158.99	6.2296	0.0192
PALM + BLWD + PALM*BLWD	5	160.65	7.8897	0.0084
PALM + BUGR + BLWD + FORB + LIT	7	162.53	9.7680	0.0033
BURN + PALM + BUGR + BLWD + FORB + LIT + EDGE + WET (<i>Global</i>)	10	164.48	11.7207	0.0012
BURN + PALM + BUGR + BLWD + FORB + LIT	8	167.13	14.3705	0.0003

which it was contained. For instance, the estimate from the top model suggests that sedge wrens were $1 / 0.3865 = 2.58$ times less likely to occur with every 25 % increase in coverage by vegetative litter. BUGR was positively related to occurrence of sedge wrens, though the effect was not as strong as that of LIT. Sedge wren occurrence was 1.53 times more likely for every 10 % increase in coverage by bunch grasses (Table 3.6). Also in 2006, distance to the nearest wetland (WET) played a role in occupancy by sedge wrens. Sedge wrens were $1 / 0.4155 = 2.4$ times less likely to be present on transects for every 100 m that the distance to the nearest wetland increased.

In 2007, the model containing only BURN was best-fitting (Akaike weight = 0.4328) and was >2.5 times more likely than the second best-fitting model containing only LIT (Akaike weight = 0.1620; Table 3.5). Sedge wren occurrence on transects that had been burned in the previous 2 years was $1 / 0.4414 = 2.26$ times less likely than on transects that had not been burned in 3 or more years (Table 3.6).

DISCUSSION

Results of this study suggest that the role of conservation and management of dry prairie in southern Florida may serve a much greater importance to migrant passerines than previously suspected, because over-winter abundances of migratory species was relatively high when compared with residents, and some species seem highly sensitive to management practices. This seems to support the claims of Hunter (1990), Lymn and Temple (1991), and others who have suggested that degradation of grassland habitats in the Southeastern United States has decreased the value of many areas as wintering habitat for some migrant grassland species. Other studies that investigated grassland birds on the dry prairie during winter, such as Fitzgerald

Table 3.6. Estimates, standard error, 95 % confidence interval, standardized coefficients, biologically relevant unit scaler, and scaled odds ratio for fixed parameters contained within the confidence set of models used to predict occurrence of wintering sedge wrens (*Cistothorus platensis*) on transects within southern Florida's dry prairie, 2006 - 2007. Scaled odds were calculated by exponentiation of the product of the parameter estimate and the unit scaler, and can be interpreted as the number of times more / less likely occurrence was after an increase / decrease in the parameter equivalent to the value of the unit scaler.

Parameter	Estimate	SE	95% Confidence Interval		Standardized Coefficient	Unit Scaler	Scaled Odds
			Lower	Upper			
2006							
LIT only							
Intercept	-0.1067	0.5836	-1.2506	1.0372	----	----	----
LIT	-3.8030	1.3839	-6.5154	-1.0906	0.02	0.25	0.3865
BUGR only							
Intercept	-4.9280	1.4645	-7.7984	-2.0576	----	----	----
BUGR	4.3051	1.8182	0.7414	7.8688	74.08	0.10	1.5380
PALM + BUGR + BLWD + FORB + LIT							
Intercept	-3.7260	2.0185	-7.6823	0.2303	----	----	----
PALM	-0.0544	1.7876	-3.5581	3.4493	0.95	0.10	0.9946
BUGR	3.5146	1.8356	-0.0832	7.1124	33.60	0.10	1.4211
BLWD	1.9304	3.0998	-4.1452	8.0060	6.89	0.10	1.2129
FORB	6.4714	4.9131	-3.1583	16.1011	646.39	0.025	1.1756
LIT	-3.4109	1.5476	-6.4442	-0.3776	0.03	0.25	0.4263
EDGE only							
Intercept	-2.5196	0.4992	-3.4980	-1.5412	----	----	----
EDGE	0.0029	0.0012	0.0005	0.0053	1.00	100	1.3318

Table 3.6 continued.

Parameter	Estimate	SE	95% Confidence Interval		Standardized Coefficient	Unit Scaler	Scaled Odds
			Lower	Upper			
BURN + PALM + BUGR + BLWD + FORB + LIT + EDGE + WET (<i>Global</i>)							
Intercept	-2.2962	2.1461	-6.5026	1.9102	----	----	----
BURN							
One Year	-0.1629	0.7129	-1.5602	1.2344	0.85	----	0.85
Two Year	0.2761	0.7802	-1.2531	1.8053	1.32	----	1.32
Three Years +	----	----	----	----	----	----	----
PALM	-0.5380	1.6505	-3.7730	2.6970	0.58	0.1	0.95
BUGR	2.9271	1.7545	-0.5117	6.3659	18.67	0.1	1.34
BLWD	1.0449	3.1505	-5.1301	7.2199	2.84	0.1	1.11
FORB	4.6682	5.0668	-5.2627	14.5991	106.51	0.025	1.12
LIT	-3.3752	1.7015	-6.7101	-0.0403	0.03	0.25	0.43
EDGE	0.0012	0.0014	-0.0016	0.0039	1.00	100	1.12
WET	-0.0095	0.0050	-0.0193	0.0004	0.99	100	0.39
WET only							
Intercept	-0.7568	0.5000	-1.7368	0.2232	----	----	----
WET	-0.0088	0.0044	-0.0175	-0.0001	0.99	100	0.4155
2007							
BURN only							
Intercept	-0.4700	0.2850	-1.0286	0.0886	----	----	----
BURN	-0.8179	0.4012	-1.6043	-0.0315	0.44	----	0.4414
LIT only							
Intercept	-1.7259	0.5983	-2.8986	-0.5532	----	----	----
LIT	1.5378	1.0466	-0.5135	3.5891	4.65	0.25	1.4688

Table 3.6 continued.

Parameter	Estimate	SE	95% Confidence Interval		Standardized Coefficient	Unit Scaler	Scaled Odds
			Lower	Upper			
EDGE only							
Intercept	-0.6193	0.3321	-1.2702	0.0316	----	----	----
EDGE	-0.0014	0.0013	-0.0041	0.0012	1.00	100	0.8661
BUGR only							
Intercept	-0.1600	0.7030	-1.5379	1.2179	----	----	----
BUGR	-1.1327	1.0231	-3.1380	0.8726	0.32	0.10	0.8929

and Tanner (1992), have reported much lower abundances of migratory grassland passerines than I observed, possibly as a result of survey techniques that were ineffective at sampling inconspicuous wintering grassland species.

Habitat characteristics and fire-return intervals had varying effects between the 2 species and, in some cases, between years. However, my results generally support the theory that grassland birds prefer to over-winter on sites that have characteristics that are similar to their breeding grounds (Johnston and Odum 1956, Grzybowski 1976, Best et al. 1998). Furthermore, my results seem to suggest that grassland passerine response to management practices such as prescribed fire is comparable between wintering habitat in south Florida and breeding grounds in more northerly latitudes.

Similar to the findings of others (Herkert 1994, Madden et al. 1999), grasshopper sparrow occurrence seemed strongly influenced by burn frequency and was consistently highest on transects with 1-year burns. This association seemed more important than, and acted irrespective of, the relationship between most vegetation characteristics and sparrow occurrence. During winter, grasshopper sparrows are primarily granivorous, and require open conditions with exposed soil at ground level to effectively forage (Vickery 1996). In the absence of fire, prairie undergrowth can become thick and matted while plant litter accumulates and reduces soil exposure; the combination of these factors may act to restrict foraging opportunities for sparrows. Frequent burning promotes more open structural conditions as well as removing dead vegetation and reducing accumulated plant litter, allowing the birds to forage more effectively. Furthermore, seed production of many plant species has been shown to increase following fire (Buckner and Landers 1979), and seed availability can have serious implications on populations of granivorous sparrows (Pulliam and Dunning 1987).

Other studies of grasshopper sparrows have reported tolerance for much longer burning regimes than I observed (Bock and Bock 1987, Johnson 1997), however, most previous work was conducted within ecosystems that have much longer natural fire-return intervals than the dry prairie. The dry prairie evolved as a highly pyrogenic environment where lightning-strike, growing season fires burnt most upland sites every 1 – 3 years (FNAI and DNR 1990). In such a highly fire-dependent ecosystem, successional changes occur rapidly in the absence of fire, and foraging conditions near the ground quickly deteriorate and likely become unusable by grasshopper sparrows within a few years.

Previous studies have documented the effects of prescribed burning on the Florida grasshopper sparrow (*Ammodramus savannarum floridanus*), an endangered, non-migratory race that is endemic to southern Florida dry prairies. Shriver and Vickery (2001) investigated the effects of prescribed winter burns on Florida grasshopper sparrows and found that sparrow densities were higher on recently burned plots when compared to plots that had not been burned in ≥ 2.5 years. They also reported that grass and grass litter had a negative effect on densities of sparrows, possibly as a result of limited foraging opportunities. During the study's first year I observed a similar, though not necessarily statistical, response to those variables. During the study's second season, the effect of both variables was negligible; I do not have a biological explanation for the lack of influence of those variables during that year. However, overall, my results suggest that response of the northern, migratory race of grasshopper sparrows responds to dry prairie management in a similar way to that of the Florida grasshopper sparrow.

Tucker and Bowman (2006) investigated characteristics of Florida grasshopper sparrow habitat and found that proportion of coverage by "runways" was an important indicator of habitat quality. They defined runways as open corridors near the ground that were free of obstruction

from vegetation, offered overhead protection from avian predators, and were not covered by vegetative litter. These areas were maintained through frequent burning, and were believed to be important because they increased foraging efficiency. Although I did not measure coverage by runways directly, it is likely that the preference that grasshopper sparrows showed for 1-year burns was closely correlated with the increase in runways that would have been a result of the recent burning. This also suggests that the lack of response exhibited by grasshopper sparrows toward many of the vegetation variables that I measured could have been a result of my failure to measure vegetation at the scale to which sparrows were responding. The measurements for most vegetation variables in my study were taken from the canopy; which occurs ~0.5 m above the soil on the dry prairie. Therefore, if Tucker and Bowman's (2006) biological interpretation of runways can be extended to migrant grasshopper sparrows, individuals in my study likely responded to vegetation characteristics within centimeters of the ground, rather than the level I measured.

Sedge wren occupancy was strongly influenced by distance to wetlands during the study's first year, suggesting that wren habitat preference while over-wintering in south Florida may be similar to their breeding grounds, where they are typically associated with mesic grasslands (Herkert et al. 2001). This relationship did not exist during the second season; this lack of effect could potentially be explained by abnormally dry conditions that existed between the 2 years, rendering many ephemeral wetlands non-existent during 2007.

I found that sedge wren occupancy was positively related to coverage by bunch grass, and that the effect of litter was strongly negative during the first year and negligible during the second. This is somewhat contradictory with results from the Great Plains that suggest that areas occupied by sedge wrens had high levels of litter (Renken and Dinsmore 1987). Evidence

suggests that sedge wrens are insectivorous on winter ranges in southern Florida (Howell 1932); therefore, large amounts of vegetative litter could potentially serve as a hindrance toward the capture of insects. This is a potential explanation for the negative effect of leaf litter observed during the first year, though I lack a biological explanation for the lack of effect during the second year. It should be noted, however, that the same pattern of a strong negative effect of litter during the first year and negligible effects during the second was also shown by grasshopper sparrows, suggesting that both species may have been responding to the same environmental phenomenon.

Burning only influenced sedge wren occupancy during 1 year, so inferences about the effects of fire on sedge wrens from my results are inconclusive. Previous studies on the effect of burning on sedge wrens have also yielded conflicting results, but generally sedge wrens seem to prefer longer fire-return intervals than most other grassland passerines. For instance, Herkert (1994) and Westemeir and Buhnerkempe (1983) both found that sedge wrens reached their highest abundances 3-years post-burn on Midwestern prairies. During the second year, the total lack of sedge wrens on transects in 1-year burns suggests that under certain conditions burning may have serious short-term consequences on individuals. This effect was not seen in the first year, as wrens did not seem to show a strong preference for any particular burn category. This suggests that some unmeasured environmental factor likely caused the birds to become much more sensitive to alterations during the first year after burning. Continuing research is needed in order to understand the implications of fire intervals on sedge wrens on the dry prairie.

I limited inference from this study to 2 of the more common winter residents. However, each species occupies opposite ends of the continuum of habitat preferences seen in grassland birds, thus I believe they may serve as adequate indicators from which generalizations about

habitat preference can be made toward the entire suite of grassland passerines that over-winter on south Florida dry prairie. For example, savannah sparrows (*Passerculus sandwichensis*) are another common winter resident on Florida dry prairie. They are a member of the family Emberizidae, making them closely related to the grasshopper sparrow, and may likely respond to fire in a similar manner. This assumption is supported by previous research on both their breeding grounds and over-wintering areas in Texas where they have shown strong affinity to 1-year-old burns (Johnson 1997, Baldwin et al. 2007). Conversely, some other species such as the swamp sparrow (*Melospiza georgiana*), that over-winter on Florida dry prairie prefer dense cover and may respond in a manner similar to that of the sedge wren.

Both species showed a high level of variability in relation to habitat parameters, but responded relatively consistently to the effects of burn interval. This suggests that some winter migrants on the dry prairie may respond to the short-term effects of fire, or lack thereof, rather than the structure of the habitat itself. Granivorous species such as the grasshopper sparrow could potentially select areas that were burned recently, because on south Florida dry prairie, seed production by many member of the plant community is likely highest immediately following burning (Buckner and Landers 1979). Conversely, a large proportion of the sedge wren's winter diet is composed of insects and fire may temporarily reduce some insect populations (Swengel 2001), decreasing the quality of those areas for insectivorous species such as the sedge wren.

MANAGEMENT IMPLICATIONS

Results from this study suggest that the dry prairie of southern Florida is likely a major wintering area for many of North America's short-distance migrant grassland birds. Over the

last half-century, the dry prairie has lost huge expanses of its former range through conversion to anthropogenic land uses (United States Fish and Wildlife Service 1999). Many of these alterations provide very little suitable habitat for grassland birds, and problems on the wintering grounds have been implicated in fueling continental declines of some grassland species (Basili and Temple 1999). Considering the level of conservation concern that is garnered by the suit of grassland species that over-winter in Southeastern savannahs, it would be wise for land managers in the region to give strong consideration to wintering grassland species in management plans aimed at promoting biodiversity.

Fire plays a major role in maintaining quality habitat for many grassland species. Current burn intervals on managed prairies in south Florida can range from 2 - 4 years (United States Fish and Wildlife Service 1999, Main and Richardson 2002). Results from this study suggest that the upper end of this interval may be too long to maintain habitat for many grassland species. Species presence and abundance are not valid surrogates for demographic parameters when assessing habitat quality (Van Horne 1983), but other studies suggest that habitat preferences in winter may be strongly tied to over-winter survival. For instance, Henslow's sparrows (*Ammodramus henslowii*) have been shown to prefer to winter in prairies and pitcher plant bogs that were burned during the previous year, possibly because these areas ensure higher survival rates (Tucker Jr. and Robinson 2003, Thatcher et al. 2006).

I recommend that managers burn dry prairie on 2-year intervals to maintain suitable habitat for wintering passerines. Glitzenstein et al. (2003) recommended that to promote maximum plant species composition and richness in the longleaf pine ecosystem of north Florida, which has herbaceous groundcover similar to that of the dry prairie, burns should be conducted as often as fuels will allow. Burning at this frequency will likely result in "patchy"

burns where fire is unable to be carried in certain areas due to discrepancies in fine fuel loads (Wade and Lunsford 1989). Areas that are left unburned will provide denser cover benefiting species such as sedge wren, while frequently burned portions will create quality habitat grasshopper sparrows and others that very fire dependent.

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Appendix 3A. Abundance of all species observed on flush transects conducted on the dry prairie of south-central Florida during late winter, 2005 – 2006. Residency status relates only to south Florida and is represented by the following: year-round non-migratory (YRNM), year-round migratory (YRM), and migrant winter-only resident (MWR). Habitat classification refers to the guild to which the species primarily belongs during the breeding season and are represented by: grassland-obligate (GR), early-successional (ES), shrub / scrub (SS), wetland (WT), woodland (WD), and generalist (GN).

		Observed	Frequency of Occurance	Residency Status	Habitat Classification
Grasshopper Sparrow	Total	216	---	MWR	GR
<i>Ammodramus</i> <i>savannarum</i>	2006	148	50%		
	2007	68	34%		
Bachman's Sparrow	Total	96	---	YRM	ES, WD
<i>Aimophila</i> <i>aestivalis</i>	2006	51	30%		
	2007	45	24%		
Common Yellowthroat	Total	77	---	YRM	ES, SS, GR
<i>Geothlypis</i> <i>trichas</i>	2006	38	19%		
	2007	39	22%		
Savannah Sparrow	Total	73	---	MWR	GR
<i>Passerculus</i> <i>sandwichensis</i>	2006	41	9%		
	2007	32	14%		
Sedge Wren	Total	70	---	MWR	GR
<i>Cistothorus</i> <i>platensis</i>	2006	25	15%		
	2007	45	29%		
Palm Warbler	Total	43	---	MWR	SS
<i>Dendroica</i> <i>noveboracensis</i>	2006	26	12%		
	2007	17	7%		
Swamp Sparrow	Total	41	---	MWR	WT
<i>Melospiza</i> <i>georgiana</i>	2006	32	15%		
	2007	9	5%		
Wilson's Snipe	Total	28	---	MWR	WT
<i>Gallinago</i> <i>delicata</i>	2006	13	4%		
	2007	15	5%		

Appendix 3A continued.

		Observed	Frequency of Occurance	Residency Status	Habitat Classification
Yellow-rumped Warbler	Total	28	---	MWR	WD
<i>Dendroica coronata</i>	2006	11	1%		
	2007	17	6%		
Northern Bobwhite	Total	22	---	YRNM	ES
<i>Colinus virginianus</i>	2006	13	4%		
	2007	9	<1%		
Eastern Towhee	Total	16	---	YRM	SS
<i>Pipila erythrophthalmus</i>	2006	9	5%		
	2007	7	3%		
Eastern Meadowlark	Total	14	---	YRM	GR
<i>Sturnella magna</i>	2006	8	3%		
	2007	6	3%		
Common Ground-Dove	Total	11	---	YRNM	SS
<i>Columbina passerina</i>	2006	9	2%		
	2007	2	<1%		
American Robin	Total	9	---	MWR	GN
<i>Turdus migratorius</i>	2006	0	0%		
	2007	9	2%		
Gray Catbird	Total	5	---	YRM	SS, GN
<i>Dumetella carolinensis</i>	2006	0	0%		
	2007	5	2%		
Carolina Wren	Total	3	---	YRNM	GN
<i>Thryothorus ludovicianus</i>	2006	1	<1%		
	2007	2	1%		
Henslow's Sparrow	Total	3	---	MWR	GR
<i>Ammodramus henslowii</i>	2006	3	2%		
	2007	0	0%		
Red-winged Blackbird	Total	2	---	YRM	WT, GN
<i>Aimophila carpalis</i>	2006	1	<1%		
	2007	1	<1%		
Northern Mockingbird	Total	2	---	YRNM	GN, SS
<i>Mimus polyglottos</i>	2006	0	0%		
	2007	2	1%		
Vesper Sparrow	Total	2	---	MWR	GR
<i>Poocetes gramineus</i>	2006	2	1%		
	2007	0	0		

Appendix 3A continued.

		Observed	Frequency of Occurance	Residency Status	Habitat Classification
Eastern Bluebird <i>Sialia sialis</i>	Total	1	---	YRM	SS, GN
	2006	0	0%		
	2007	1	<1%		
White-eyed vireo <i>Vireo griseus</i>	Total	1	---	YRM	SS
	2006	1	<1%		
	2007	0	0%		
Yellow Rail <i>Coturnicops noveboracensis</i>	Total	1	---	MWR	WT
	2006	1	<1%		
	2007	0	0		
Unknown	Total	64	---	---	---
	2006	30	---		
	2007	34	---		
Unknown Sparrow	Total	31	---	---	---
	2006	26	---		
	2007	5	---		

Appendix 3B. Mean and standard deviation for covariates used in models to predict probability of occupancy by grasshopper sparrows (*Ammodramus savannarum pratensis*) and sedge wrens (*Cistothorus platensis*) on transects within south Florida dry prairie, 2006 – 2007.

	2006		2007	
	Mean	SD	Mean	SD
Saw palmetto coverage	0.3763	0.1872	0.4801	0.1896
Bunch grass coverage	0.6668	0.1915	0.6763	0.1920
Broad-leafed woody coverage	0.1613	0.0966	0.2516	0.1449
Forb coverage	0.0792	0.0618	0.0580	0.0583
Vegetative litter coverage	0.4816	0.2173	0.5137	0.1972
Distance to edge	235.94	177.58	215.71	168.18
Distance to wetland*	79.87	177.57	126.3	69.01

* - used only in sedge wren models

CHAPTER 4

CONCLUSIONS, RECOMMENDATIONS, AND FUTURE RESEARCH

The dry prairie of south-central Florida comprises one of the largest remaining grassland ecosystems in the Southeastern United States. Though the area is home to several species of concern, surprising little research has addressed the relationship between the anthropogenic induced changes and subsequent population declines in many regional wildlife species. Through this study, I attempted to address how some of these current conditions may affect certain members of the bird community. I focused on select members of both the breeding and wintering community using a combination of point counts and “flush” transects. My aim was to assess how members of each group responded to habitat conditions, patch features, and management. Specifically, I investigated the influence of patch size, habitat structural characteristics, and burning within the previous year on both Bachman’s sparrows (*Aimophila aestivalis*) and eastern meadowlarks (*Sturnella magna*) during a 2-year span. For the overwintering species, I limited inference to 2 species, grasshopper sparrows (*Ammodramus savannarum pratensis*) and sedge wrens (*Cistothorus platensis*), and focused on more explicitly on the effect of fire regime, in addition to the influence of many of the same habitat features.

Breeding birds showed considerable variability regarding the some of the covariates in my study. Neither species responded strongly to differences in patch size in both years. Eastern meadowlarks exhibited a positive relationship between patch size and abundance in the first year. Conversely, patch size did not seem to be an influencing factor for that species during the following year. Bachman’s sparrows only showed a response to patch size during the study’s

second year. However, I believed that the inverse relationship between Bachman's sparrow abundance and patch was more likely attributable to differences in dispersal probability between study sites, rather than a response to patch size.

Both of the breeding bird species seemed to respond to habitat conditions that were a result of long-term frequent fire. Bachman's sparrows were negatively affected by saw palmetto (*Serona repens*) after it reached approximately 40 % coverage. Eastern meadowlarks consistently preferred areas in which vegetation was less dense. High levels of bunch grasses seemed beneficial to both species. The short-term effect of burning within the previous year seemed negligible to both species, and lack of information on some study sites prevented a more detailed glimpse into the effect that fire may have on the 2 species. However, previous research has shown that the habitat characteristics to which the 2 species responded can be promoted and maintained through the use of frequent fire.

The 2 over-winter migrants consistently showed the strongest response to time since prescribed burning. Grasshopper sparrow occurrence was much higher in transects that had been burned within the previous year, while sedge wrens seemed to prefer areas that had not been burned in several years. These differences were likely related to differences in diet between the 2 species and the changes in those food resources following burns.

MANAGEMENT RECOMMENDATIONS

Results of this study suggest that fire plays a critical role on the dry prairie. Maintenance of habitat conditions for many resident species will require frequent prescribed burning. On prairie fragments that have not been burned in many years, mechanical treatment such as roller-drum chopping may quickly create favorable conditions which are then able to be maintained

through use of burning alone. Previous research has suggested that prescribed burning and mechanical treatments can shift the structure and composition of the dry prairie towards the conditions that I found to be optimal for grassland and savannah species (Fitzgerald and Tanner 1992). My research suggests that burning intervals on dry prairie should be kept to less than 3 years, and this interval corresponds to historical estimates of fire return on dry prairie (United States Fish and Wildlife Service 1999). In order to ensure that this rate of burning is carried out, I recommend that land managers target a biennial burning cycle. Targeting this frequency will ensure that in the event that environmental or logistical constraints do not allow adequate areas to be burned on the 2-year rotation, burning the following year will still fall within the natural 1 - 3 year time span. The long-term effects of burning at this frequency will create habitat conditions that benefit resident savannah species, while short-term effects such as increased seed production will benefit wintering migrants.

Fragmentation and outright habitat loss are potentially the 2 greatest factors facing the future of many species that utilize the dry prairie (United States Fish and Wildlife Service 1999). Results of this study suggest that some members of the dry prairie avian community may be negatively affected by the reduction in size and subsequent isolation of prairie fragments. This highlights the need to conserve large blocks of dry prairie and promote connectivity between remnant patches.

FUTURE RESEARCH NEEDS

Given the lack of knowledge surrounding avian ecology on the dry prairie, many questions still need to be addressed to advance wildlife conservation within the region. My research has focused on some of the basic habitat preferences of 4 common, yet declining,

species that utilize dry prairie. However, several limitations of my study reduced the level of inference that can be garnered. Therefore, further investigation into the ecology of savannah birds on the dry prairie is warranted. Particularly, longer duration studies are needed which will be able to find more definitive patterns, and overcome the variability that I observed in relation to some habitat parameters. Additionally, my use of characteristics such as abundance and occurrence to make inference about the value of certain habitat conditions is somewhat troublesome; incorporation of demographics such as survival and fecundity would yield a more certain assessment of habitat quality (Van Horne 1983). This would be particularly useful in understanding the influence of landscape fragmentation and patch size, as most published research on the subject suggests that nest success and fecundity are most affected.

A large portion of the remaining dry prairie is devoted to livestock production; this is particularly true for privately owned parcels. No empirical research has ever investigated the relationship between cattle grazing and birds within these areas. Research from other grasslands across North America suggest variable effects of grazing on grassland birds, though overgrazing typically is detrimental to grassland birds (Askins et al. 2007). Given the economic importance of livestock on south Florida rangelands, understanding how to successfully strike a balance between cattle production and grassland birds is critical to successful wildlife conservation within the region.

Within my study, I did not differentiate between dormant and growing season burns, and a more detailed account of the relationship that this aspect of fire ecology has on vegetative response, and thus habitat quality, for many bird species is needed. Historically, most of the dry prairie would have burned during the spring and early summer due to a preponderance of lightning strikes (United States Fish and Wildlife Service 1999). Modern landowners have

tended to burn south Florida rangelands during the late winter, and little is known about the effect of this shift away from the natural burning period. A manipulative study comparing treatments plots in which fire had been applied at different seasons would help address some of the uncertainty surrounding this relationship.

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