Fall 12-20-2013

Effects of Prescribed Burns on Grassland Breeding Birds at Mississippi Sandhill Crane National Wildlife Refuge

Elizabeth A. Ruffman
University of New Orleans, eruffman@gmail.com

Recommended Citation

Ruffman, Elizabeth A., "Effects of Prescribed Burns on Grassland Breeding Birds at Mississippi Sandhill Crane National Wildlife Refuge" (2013). *University of New Orleans Theses and Dissertations*. 1776.
https://scholarworks.uno.edu/td/1776

Effects of Prescribed Burns on Grassland Breeding Birds at Mississippi Sandhill Crane National Wildlife Refuge

A Thesis

Submitted to the Graduate Faculty of the
University of New Orleans
in partial fulfillment of the
requirements for the degree of

Master of Science
In
Biological Sciences

By

Elizabeth A. Ruffman

B.S. Tulane University New Orleans, 2008

December 20, 2013

Dedication

This work is dedicated to the memory of my father, Joel Ruffman, whose passion for science and nature continue to guide me.

Acknowledgements

I would like to thank my advisor, Dr. Jerry J. Howard for his support, patience and advice throughout. I would also like to thank my committee members, Dr. Simon Lailvaux and Dr. Peter Yaukey for their guidance and assistance. I received training and support from Scott Hereford, Supervisory Wildlife Biologist at Mississippi Sandhill Crane National Wildlife Refuge. Thanks go to you and your staff for granting me access to the refuge and sharing your insights on this project.

Special thanks to lab mate Cara Nighohossian and fellow graduate students Lauren Gonzalez and Lyndon Coghill for providing encouragement and advice these last two years. Lastly, thank you to my mom, Martha, and sister, Rachel, and Darrell Davis, for their love, patience and support.

Table of Contents

Figures and Tables

Abstract

There has been a critical decline in grassland bird populations due to habitat fragmentation and deterioration, and suppression of natural fires. Alteration of the disturbance cycle may lead to changes in vegetation structure and thus habitat suitability for breeding grassland birds. Management practices at the Mississippi Sandhill Crane National Wildlife Refuge, including the use of prescribed fire, are in need of evaluation. My study asked what frequency of prescribed burns is necessary to support breeding grassland birds and whether vegetation structure varies among burn units. In this study, bird abundance and species richness did not differ significantly among burn units and vegetation cover-type was not a strong predictor of these factors either. There was evidence of site utilization by breeding grassland birds immediately following a burn, which suggests that the bird community is able to recovery quickly post-fire and these units may serve as viable habitat for breeding grassland birds.

Keywords: Longleaf pine savanna, grassland birds, prescribed fire, vegetation structure

Introduction

North American Grassland Bird Population Trends

Grassland birds include any bird that is reliant upon grassland habitats for all or part of its life cycle (Vickery et al. 1999). Longleaf pine (*Pinus palustris*) savannas, a type of grassland habitat found in the southeastern United States, support a large variety of avifauna; Engstrom (1993) identified 86 species of birds that are characteristically found in longleaf pine forests. Thirty-five of these are considered permanent residents, 29 are breeders and 22 are winter visitors. Brawn et al. (2001) found that of 27 species of grassland birds studied, 56% had experienced a significant decrease in population from 1966-1998. Among the many grassland birds adapted to southeastern pine savanna habitat, there are three essentially endemic species, including the Brown-headed Nuthatch (*Sitta pusilla*), the Bachman's Sparrow (*Peucaea aestivalis*), and the Red-cockaded Woodpecker (*Picoides borealis*). The latter two are heavily dependent upon fire disturbance to maintain an optimal habitat (Brawn et al. 2001, Jackson 1988).

North American grassland bird populations have seen substantial declines in recent decades (Vickery and Herkert 2001, Vickery et al. 1999). In fact, they have sustained a greater, more consistent decline than any other ecological or behavioral guild (Knopf 1994). Between 1966 and 1996, 13 grassland species experienced a significant decline, while only 3 species were known to have increased in population size (Peterjohn & Sauer 1999). Grassland birds have experienced declines due to several factors including the use of pesticides, livestock grazing, urbanization, habitat loss and fragmentation, forestry practices, invasion by woody vegetation, and the conversion of grasslands for agriculture (Askins et al. 2007, Veech 2006).

Grassland birds and their response to fire regimes

Despite the conservation concern for grassland birds, the relationship between bird diversity and fire regime (the key natural disturbance in grassland habitats) remains largely undefined. According to Connell (1978), highest community diversity is expected to occur at intermediate levels of disturbance. If disturbances are too frequent, too large, or too intense, diversity will be low because only a few species will be able to colonize the area. If disturbances are too infrequent, too small or too low in intensity, diversity will decline because a few species will outcompete others for resources resulting in competitive exclusion. Highest diversity is expected at intermediate levels of disturbance because it allows sufficient time for multiple species to establish in the ecosystem and share resources.

Grassland birds are sensitive to disturbance and may react variably to prescribed burn based upon the frequency, size and intensity of the burn as well as its effect on the structure of the vegetation. Breeding birds use certain environmental cues to discern which habitat is best suited to their needs. These cues may include vegetation structure, the availability of food, nest site availability and others. It is clear that disturbance of any kind will affect these variables and thus a bird's habitat preference by altering its foraging, or nesting behavior (Brawn et al. 2001, Provencher et al. 2002). Fire is a dominant ecological process in prairies and savannas that alters vegetation structure, breeding habitat, and resource availability and evidence suggests that grassland birds are highly sensitive to fire disturbance (Reinking 2005, Engstrom et al. 2005). It is estimated that about one third of the bird species found in this ecosystem "forage on or close to the ground, or in shrubs in mature, fire-maintained woodlands" (Engstrom et al. 2005). Because of this fact, it is a research priority to understand how disturbance affects habitat suitability for a guild of birds so closely associated with groundcover.

Previous studies have investigated the response of grassland birds to fire management practices, however the exact impacts of fire to longleaf pine groundcover and avian communities have not been clarified (Rutledge and Conner 2002). In some cases, species richness and density of birds were greater in recently burned sites than on un-burned sites. A review on bird communities in the south by Dickson (2000), found that some birds were attracted to smoke or fire and had been detected on recently burned sites including wild turkeys (*Meleagris gallopavo*), mourning doves (*Zenaida macroura*), eastern bluebirds (*Sialia sialis*), pine warblers (*Setophaga pinus*) and some woodpeckers. Reduction of hardwoods via prescribed and natural burns has been seen to have a positive effect for many species. This review cited instances in which fire applications increased populations of birds that are associated with grass-forb and low shrub vegetation found on pine savannas such as common yellowthroat (*Geothlypas trichas*), Bachman's sparrow, indigo bunting (*Passerina cyanea*), eastern towhee (*Pipilo erythrophthalmus*), yellow-breasted chat (*Icteria virens*), and prairie warbler (*Setophaga discolor*). It also indicated that as the savannas developed and were increasingly composed of hardwood species, populations of these same species diminished (Dickson 2000).

In another study of grassland bird response to intense fire and fire exclusion, Allen et al. (2006) found that certain species were found more often on fire-intense sites, while others favored unburned sites. Species associated with the longleaf pine assemblage were found in greater numbers on fire-intense study sites. Listed among those were Bachman's sparrow, prairie warbler, brown-headed nuthatch, eastern wood-pewee (*Contopus virens*) and red-cockaded woodpecker. Species normally associated with fire-suppressed assemblage, such as wood thrush (*Hylocichla mustelina*) and tufted titmouse (*Baeolophus bicolor*), were more common in fire-suppressed study sites. Those species categorized as a generalist assemblage had varying responses to fire-intense and fire-suppressed study sites. While Carolina chickadees (*Poecile carolinensis*), summer tanagers (*Piranga rubra*) and great-crested flycatchers (*Myiarchus crinitus*) were found in greater abundances on fire-suppressed areas, blue jays (*Cyanocitta cristata*), northern flickers (*Colaptes auratus*), and red-bellied woodpeckers (*Melanerpes carolinus*) were in greater abundance on fire-intense sites (Allen et al. 2006). This demonstrates the varied response of guilds and species to fire treatments and the need for greater investigation of the particular habitat requirements for grassland breeding birds.

Comparison of fire regimes and avifauna response in a variety of ecosystems

The longleaf pine savanna ecosystem is just one of several that supports the avifauna of North America. Longleaf pine savannas are fire-mediated ecosystems. They are affected by the intensity and frequency of burns, and as a result of these burning regimes they may experience a shift towards grassland or forest (Beckage et al. 2009). Following either a natural or prescribed burn, there is a growth response in the vegetation community. Frequent, low-intensity fires that reduce fuel loads of ground-layer vegetation characterize this grassland ecosystem. However, they generally do not affect overstory trees and "do not substantially change the structure of the dominant vegetation" (Saab and Powell 2005). Furthermore, owing to the fact that the majority of nutrient storing tissues of grassland ecosystems are stored underground, the impacts on nutrient cycling processes are greatly reduced when compared with pine forests and chaparrals (Boerner 1982). This may affect how quickly longleaf pine savanna vegetation recovers post-fire.

Different ecosystems respond variably to fire management practices. They may vary in vegetation return time, which may directly affect vegetation structure and thus breeding bird habitat preferences. Saab and Powell (2005) found that the greatest changes in fire regimes were in ecosystems that had short return-intervals; those with long fire-return intervals such as those found in the Pacific Northwest, had experienced the least change. They also found that the spatial scale of fires has generally increased in the western United States and decreased in eastern and central United States. In addition they and other authors have characterized the response of avian communities to fire regimes among different ecosystems. Their review found the response of avifauna to prescribed burns varied greatly among guilds, species, and ecosystems. However they did discover that ground, bark and aerial insectivores were found in greater densities on recently burned areas. In addition, species with closed nests, such as cavity nesters, preferred recently burned areas. In contrast, foliage gleaners, ground and canopy nesting birds, and those with open-cup nests preferred unburned habitats (Saab and Powell 2005). These findings suggest that birds are highly sensitive to alterations of their habitat by fire and may respond variously depending upon local conditions.

Much like the longleaf pine savanna of the southeastern Coastal Plain, frequent, low-intensity fires characterize northeastern deciduous forests. It is thought that the forests experienced a 5-7 year fire-return interval pre- and post-European settlement. However, little is known about the effects of prescribed burns on birds in eastern deciduous forests (Artman 2005). In a study by Artman et al. (2001), the effect of prescribed burns on bird populations in southern Ohio was measured. Plots ranging from 20-30 ha were burned either every year or every two years over a four-year study period. They found that low-intensity surface fires did result in a significant decrease in populations of species of ground- and low- shrub-nesting birds. Populations of canopy and mid-story nesters did not significantly decrease after repeated prescribed burns. This may be due to the fact that these low-intensity fires do not significantly disturb the canopy and mid-story and thus, nests were not threatened (Artman et al. 2001, Artman et al. 2005).

The tallgrass prairies of the Great Plains represent another ecosystem that supports a wide variety of avifauna. It is understood that fires play a historical role in the maintenance of tallgrass prairies and that both vegetation and birds have become adapted to these disturbances. Fire is a necessary management tool and without it tallgrass prairies experience ecosystem shifts and may become dominated by woody vegetation (Reinking 2005). Ideal fire frequency in this ecosystem is of some debate. While regular burns are needed, burning too frequently could lead to diminished vegetation structural complexity and reduce suitable breeding habitat for grassland birds (Zimmerman 1992). Reinking (2005) suggests that a pre-settlement fire-return interval of 3-4 years with a maximum of 10 years probably existed and that a 3-year fire-return interval is sufficient to produce vegetation structural complexity. Askins et al. (2007) points out that certain grassland species respond negatively and some respond positively to fire. To accommodate the needs of the greatest number of tallgrass prairie birds, a "rotational management program" should be considered in which small portions of prairie are burned with 3-5 years between burns (Askins et al. 2007). This would create a heterogeneous, patchy environment in varying stages of recovery from burns.

Sagebrush (*Artemisia spp.*) communities are another example of a fire-mediated ecosystem that occurs primarily in western North America. Sagebrush ecosystems support a variety of birds, but typically fewer species than other ecosystems. It has been observed that bird species diversity increases when sagebrush ecosystems comprise a shrub-grassland mosaic (Knick et al. 2005) but decreases when converted to annual grasslands owing to reduced vegetation structural complexity. At the other extreme, bird diversity has been observed to increase moderately in tree composition, but begins to decrease with excessive tree dominance as the shrub and grass layer becomes excluded (Medin et al. 2000, Miller et al. 2000).

Like most temperate ecosystems in North America, sagebrush ecosystems have experienced an alteration in their fire regime in the past century. Due to increased fire suppression and livestock grazing fire frequency has increased drastically which, in turn, has altered the composition of the landscape. Sagebrush probably experienced a fire frequency of 30 to >100 years between fire events. However, due to the invasion of exotic cheatgrass (*Bromus tectorum*), the fire frequency has been reduced to as little as 10-20 years in sites that contained greater grassy composition (Knick et al. 2005). The increase in extensive, high-severity fires in this ecosystem has made it unsuitable for the bird community that uses sagebrush as its primary habitat. It has been noted that burns which reduce sagebrush by <50% do not have a significantly negative impact on the avifauna, however, those that alter >50%, do. This was seen in loggerhead shrike (*Lanius ludovicianus*) populations which rely on this habitat for nesting and experienced a 50% decrease in population (Knick et al. 20005) Cheatgrass is able to out-compete native grasses because it sets abundant seed annually and before native grasses. It promotes the spread of fires and leads to a more patch distribution of sagebrush. Prescribed burns have been used to restore sagebrush, however they have not been successful in eradicating the invasion of exotic grasses and in some instances may promote their establishment.

There is limited information on how birds respond to prescribed burns in sagebrush. Previous studies on bird response to fire may not have taken into account the fact that long-term effects may not be accurately captured by short-term studies (Knick et al. 2005). In a study by Peterson and Best (1987), prescribed burns were initiated on experimental plots of 6.25 ha each which resulted in approximately 50% reduction in sagebrush cover. Songbird censuses were conducted on these plots for five years and they found prescribed burns did not significantly reduce songbird density. This may be due in part to the fact that the prescribed burns resulted in a patchy distribution of sagebrush at a larger landscape scale and thus was still able to provide enough structural complexity for nesting songbirds (Peterson and Best 1987).

Longleaf Pine Savanna Grasslands

Range and Composition

Southeastern pine savannas historically extended from southeastern Virginia to Florida, and west to areas within Louisiana and Texas (Engstrom et al. 2005). Longleaf pine savannas are known for having a continuous grassy, herbaceous layer with trees interspersed sparsely. They have among the highest plant diversity levels in North America, some estimating as many as 50 species m-^{2} (Loudermilk et al. 2011). While there is a great variety of plant species in the understory layer, it is the herbaceous layer that contributes the greatest biodiversity. It is dominated by graminoids such as bluestem (*Andropogon sp.*) and wiregrass (*Aristrida sp.*) (Frost 2006). In addition, longleaf pine savannas are home to a great number of rare and endangered species. This may be due in part to the wide geographic range of longleaf pine savannas and the variety of environmental gradients that provide niches for populations of rare plants and animals (Noss 1995). In addition, to this, the effect of disturbance has contributed to high biodiversity in the longleaf pine savanna ecosystem (Van Lear et al. 2005).

Longleaf Pine Decline

It is estimated that more than 80% of the total area of grassland ecosystems in North America have been destroyed since the mid-1800's and since European settlement, there has been >97% loss of longleaf pine savannas in the southeastern coastal plain (Knopf 1994, Noss et al. 1995). Longleaf pine savannas once covered approximately 37 million hectares of the southeastern Coastal Plain (Frost 2006). Current estimates are that 1.2 million ha (<3%) of longleaf pine savanna ecosystem remain, making it the third most endangered ecosystems in the United States (Outcalt and Sheffield 1996, Noss et al. 1995). The declines are largely due to settlement, agriculture and grazing of livestock, suppression of natural fires, and logging (Frost 2006).

Longleaf Pine Fire Ecology

The longleaf pine forest is considered to be a pyroclimax ecosystem (Frost 2006). Fire is an essential component of the longleaf pine savanna in that it maintains species diversity, excludes woody species and encourages extensive herb and graminoid ground layer growth (Garren 1943). Longleaf pine savannas are characterized by high plant diversity found in the ground layer that is maintained by fire. Although the frequency of natural fires caused by lightning strikes pre-European settlement is a subject of much debate, current estimates point to a 1-5 year interval (Askins 2002). Fire frequency among pine savannas may vary depending upon local climate and vegetation community composition (Engstrom et al. 2005). Fire frequency may be proportional to the size of the fire compartment; larger fire compartments (>1000km^2) are expected to experience fires at a higher frequency (Frost 2006). These fires are characterized as frequent, low-intensity ground fires that do little damage to soil and overstory (Van Lear and Harlow 2002). They are fueled by grasses and occur most often in the summer months. Fire plays a vital role in maintaining a grass-dominated landscape interspersed with longleaf pines and devoid of woody shrubs (Askins et al. 2007, Shepherd et al 2012).

Many plant species in this ecosystem are not only fire-tolerant, but are fire dependent. Longleaf pines are considered a pioneer species and are highly tolerant of fire but do not compete well with introduced woody species (Frost 2006). The ground layer is composed of graminoids that act as fuel for fires. These graminoids, such as wiregrass, require fire to shed seeds and growing season burns to produce seeds. The close relationship between longleaf pines and graminoids has developed because they provide fuel to fires. These graminoids burn quickly and support relatively cool, low intensity fires that characterize the longleaf pine savanna ecosystem. Likewise, longleaf pines are reliant upon fire and have evolved adaptations to aid in survival of fires. Longleaf pines are able to survive fires due to several adaptations. The terminal bud is protected by a layer of needles that deflect the heat of the fire by releasing water stored within the needle. Seedlings have a well-developed taproot that remains un-burned and provides food for the seedling to grow post-fire (Askins 2002). Longleaf pines require fire to generate this growth spurt from seedling to young tree, making it more likely to survive the next fire. They remain in a low-growth "grass stage" which may last between 10-25 years until they experience a fire of the correct intensity to generate a growth spurt (Jose et al. 2006).

The interval between fires today has been lengthened since pre-European settlement. This is due to human suppression and exclusion of natural and prescribed fire in this ecosystem (Engstrom et al. 2005). Fire suppression has been practiced in the United States for over 150 years. It has interrupted the natural fire cycle of habitats where fire was historically responsible for the maintenance of biodiversity and suppression of woody encroachment (Frost 1998). Fire suppression may negatively impact both xeric and mesic longleaf pine savannas. Among the most noticeable changes is the reduction in herbaceous plant diversity coupled with the invasion of woody species. When natural fires are suppressed this leads to an encroachment of fire intolerant woody species that replace native herbaceous vegetation unable to compete for resources. Fire exclusion and suppression also lead to an increase in litter which make longleaf pine seed germination difficult (Dickson 2000, Jose et. al 2006). Today, an estimated 74% of the remaining longleaf pine ecosystem experiences fire suppression and is in danger of woody encroachment (Askins 2002).

Prescribed Burns

Although exact knowledge of historical fire regimes for the longleaf pine savanna ecosystem is not known, scientists have reviewed the subject extensively. Fires have naturally occurred via lightning strike in this ecosystem for thousands of years; this historical co-occurrence of fire may have shaped the fire-tolerant nature of vegetation characterized in this ecosystem (Komarek 1974, Frost 2006). The use of fire as a management tool has its roots in Native American practices, who utilized it to clear closed forests and open territory for hunting (Frost 2006). Fire suppression and exclusion was practiced widely in the United States post-European settlement.

Managers previously thought that fires interfered with longleaf pine restoration and growth, and that suppression should be practiced (Mattoon 1922). However, the use of fire as a management tool gained wider acceptance in the 1980's (Saab and Powell 2005) when land managers began to incorporate prescribed burns into management plans. Today, the practice of prescribed burn fire regimes is widely used as a management tool for the restoration of pine savannas. The goal of this practice is to mimic the natural ecological processes of this ecosystem by replicating the timing, intensity, and season of burns (Vickery et al. 1999). Managers are using this tool to restore longleaf pine savannas, in particular the herbaceous layer, and for wildlife management purposes. The effects of fire frequency, season of fire, and intensity of fire have been studied to determine effects on breeding grassland birds. However, optimal frequency of prescribed burns is a continuing debate amongst managers as response to fire varies among different guilds of avifauna.

The current consensus is that the longleaf pine savanna ecosystem is expected to reach its highest diversity of plants with a fire-return interval of 2-3 years. Although fire is expected to influence bird community diversity, fire frequency in managed longleaf pine habitats is generally determined primarily by its impact on the plant community. Annual fires are not an optimal management regime to restore or maintain longleaf pine savanna ecosystem because they generally do not allow the establishment of longleaf pine seedlings that are too fragile to survive a fire in their first year. Interspersed fire-free years are important as they allow growth of first-year longleaf pine seedlings along with the regeneration of established seedlings and groundcover (Garren 1943). However, the effect of fire return intervals of 2-3 years on bird communities has not be explicitly examined for its effect on bird diversity.

Research Objectives

The objective of this study was to determine the effect of prescribed burns on grassland breeding bird habitat preferences on the Mississippi Sandhill Crane National Wildlife Refuge (MSCNWR). My study focused on time since last burn and vegetation structure as factors that may influence these preferences that include comparing compartments differing in time since last burn and their vegetation structure. While literature suggests that high vegetation diversity in the ground layer of longleaf pine ecosystems is achieved using a 2-3 year burn regime (Garren 1943); the fire regime to optimize species richness and abundance of grassland birds has not been explicitly evaluated. Prescribed burns done at the correct frequency, intensity, and size should create a habitat mosaic that supports a wide variety of avifauna (Peterson and Best 1987, Enstrom et al. 2005). If the current size and frequency of burns successfully establishes a habitat mosaic, no significant difference in species richness or abundance would be expected among management compartments that varied in time since burn between 0-3 years.

Methods

Study Site

The study was conducted at the Mississippi Sandhill Crane National Wildlife Refuge in Jackson County, MS. The refuge was established in 1975 under the Endangered Species Act to protect the endangered Mississippi Sandhill Crane and pine savanna habitat. The refuge employs prescribed burns for habitat management both during the dormant and growing seasons. In order to simulate a natural fire cycle current practice seeks to maintain a 2 to 3 year burn rotation among management compartments on the refuge (Figure 1). The refuge supports a wide variety of grassland birds during the spring breeding season. A major goal of the refuge is to optimize prescribed burn regimes to provide suitable breeding habitat for these grassland bird species. The refuge has identified several species of birds that are of a high conservation priority to the East Gulf Coast Plain. These include: Mississippi sandhill crane (*Grus canadensis pulla*), prairie warbler, sedge wren (*Cistothorus platensis*), northern bobwhite (*Colinus virginianus*), red-cockaded woodpecker, Bachman’s sparrow, brown-headed nuthatch, Henslow’s sparrow (*Ammodramus henslowii*), and American kestrel (*Falco sparverius*) (MSCNWR Comprehensive Conservation Plan 2007).

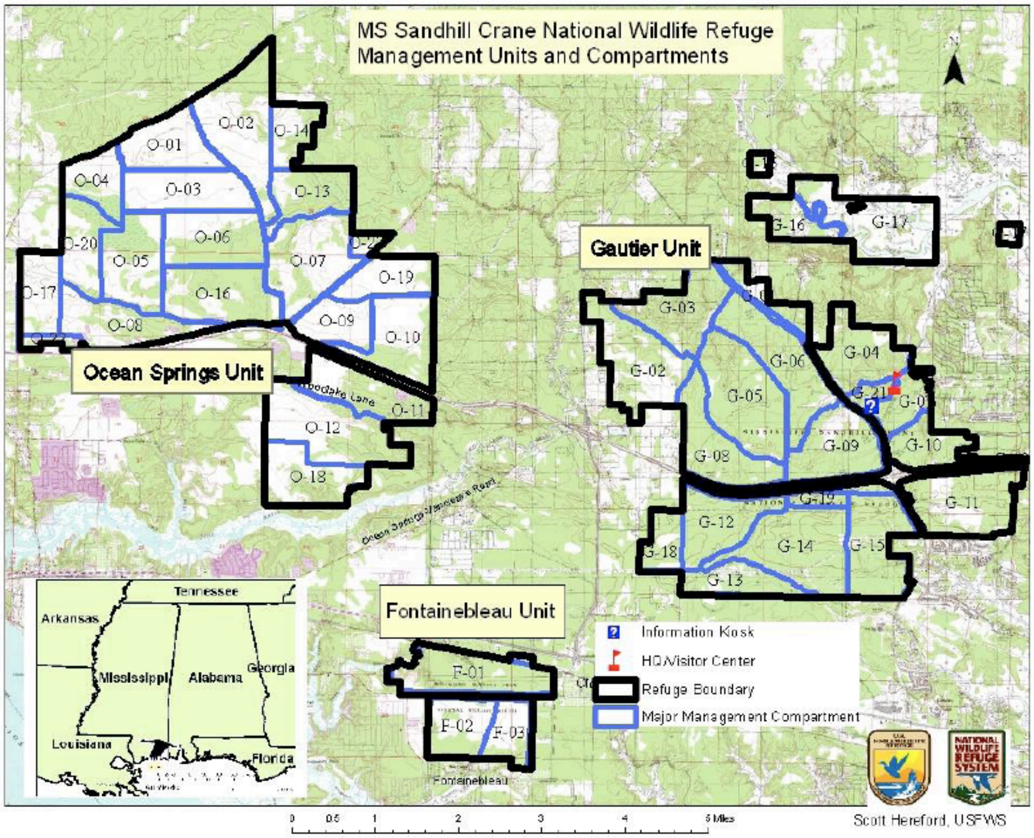

Figure 1. Map of the Mississippi Sandhill Crane National Wildlife Refuge showing management compartments

Sampling Design

To assess bird assemblage on the refuge I selected management compartments that had been burned within the last year (Age-0), those with one previous growing season (Age-1), two growing seasons (Age-2) and three growing seasons (Age-3) since the last prescribed burn. I used three replicate compartments for each burn age category, with the exception of Age-2 in which two compartments were surveyed, for a total of 11 management compartments (Table 1). Within each compartment, 6 points were randomly chosen to conduct both bird point-count surveys and vegetation surveys.

Table 1. Management compartments used to study spring breeding grassland bird communities at MSCNWR

Compartment	Last Burn Date	Hectares Burned	Burn Treatment
G-15S	12-Mar	62	0
G-05S	12-Mar	148	0
G-05N	12-Mar	148	0
G-07	11-Apr	205	1
F-03	11-Jan	95	1
G-11	11-Feb	143	1
O-07	10-Apr	259	2
O-10	10-Mar	90	2
O-02N	9-Feb	36	3
G-3	9-Apr	206	3
G-14	9-Mar	293	3

I conducted bird point-count surveys from April 13, 2012 to May 19, 2012. Point count stations were located at least 100 meters from any edge to ensure that grassland birds were being sampled. An edge was defined as a boundary to grassland habitat consisting of either a road or densely forested area. I established point-count stations at least 250 meters apart to reduce re-sampling of birds from previously surveyed point count stations. Vegetation surveys were conducted between June 15, 2012 and August 12, 2012 during the summer growing season.

Bird Surveys

I surveyed grassland birds on the 11 management compartments using the standard point count survey method for grassland birds in the southeastern United States (Hamel et al. 1996). Surveys were conducted during a time period of one half hour prior to sunrise until 10am. Point count surveys were not conducted under unfavorable weather conditions, including fog, smoke or rain, and winds > 13km/hr. During each survey, one observer stood at the point count station and recorded the time of first detection for each individual identified. Detections were classified into one of five categories (0-1min, 1-2min, 2-3min, 3-4min, 4-5min). The observer spent time facing each of the cardinal directions to optimize detection of birds. Birds were detected by sight and aurally. Distance and direction of each bird to the observer were estimated. When necessary, a rangefinder was used to accurately estimate distances. Detections were placed in distance categories of 0-25m, 25-50m, and 50-100m. Each point-count survey lasted for a total of 5 minutes. This length of time was chosen based on the assumption that travel time between consecutive point-count stations would require less than 15 minutes and thus reduce the likelihood of re-sampling previously recorded birds (Hamel et al. 1996). The observer also noted sky conditions, wind speed, and air temperature.

I calculated bird species abundance and species richness for each burn age category. I also calculated the Shannon diversity (*H*) and equitability (E_H) across the burn age categories. The Shannon diversity index (*H*) is commonly used in ecology to measure species diversity and corrects for relative abundance of species in a community (Equation 1). A high value of H represents community with high species diversity, while a value of 0 represents a community with only one species.

$$H = -\sum_{i=1}^{S} p_i \ln p_i$$

Equation 1: Shannon diversity index

Shannon equitability index (E_H) was used to compute equitability (Equation 2). Its value ranges from 0 to 1, with a value of 1 indicating complete evenness.

$$E_H = H / H_{\max} = H / \ln S$$

Equation 2: Shannon equitability index

Vegetation Surveys

I assessed vegetation structure surrounding each point-count station where a bird survey was conducted using the Point-Quarter Center Method (Barbour 1987). I constructed a cover pole made from 1m of bamboo and partitioned its length into height class intervals of 5cm from 0-20cm and intervals of 10cm up to 1 m.

Vegetation was sampled using the cover pole at 9 points at each bird point-count station: at the center, where the original bird point-count had been conducted, and in the four cardinal directions around the center at distances of 2 and 4 meters.

I used the pole method to estimate vegetation density and cover type at each vegetation sample point. I classified vegetation into six types: graminoid (grasses and sedges), forb, shrub, carnivorous plant, tree or saw palmetto (*Seronoa repens*). Vegetation density was estimated using the number of contacts of each vegetation type in each height interval from 0-1m. Cover type at each sample point was determined based on the vegetation type with the highest contact point on the cover pole.

Statistical Analysis

Statistical analysis was performed in SYSTAT version 11.

I pooled vegetation density scores into three height classes: 0-30cm, 31-70cm, and 71- over 100cm. To assess density of each category of vegetation, I calculated the average density of the nine sample points at each point-count station and generated the average density of each height class over the four burn age categories. Because much of the vegetation density data did not conform to a normal distribution, I ranked the vegetation density scores prior to any further analysis. I then ran a one-way ANOVA and Tukey's HSD post hoc analysis for each of the height classes on the ranked average density scores.

To understand the relationship between abundance of birds with respect to time since burn and cover type at each bird point-count station, I ran a two-way ANOVA with Tukey's HSD post-hoc analysis. I repeated the process to analyze the relationship of species richness with respect to time since burn and cover type.

Bird Survey Analysis

I performed a one-way ANOVA on the diversity index (H), equitability (E_h), species richness, and abundance for each burn age category. This data was tested for normality using Shapiro-Wilk's normality test and was found to have normal distributions.

For each species of bird I calculated percent of presence at the 66 point-count stations. For the group of species with percent present of >20%, I calculated what percentage they made up of all detections.

I characterized each bird species by nest microhabitat preference (Poole 2005). For each category of nest microhabitat preference, I ran a two-way ANOVA for the corresponding bird species' abundance over the four burn age categories. The nest microhabitat categories identified were: cavity, tree, ground, shrub, and burrow.

Estimation of Bird Density

I used Program Distance version 6.0 to generate estimates of relative density for each bird species from the raw bird detection data. To analyze the results of the bird point-count survey, I used common distance sampling (CDS) methods that are regularly used in ecological studies to estimate the relative density of avian populations. Distance sampling differs from finite population sampling in that it accounts for the fact that many objects may be undetected and the sample area size may not be known (Buckland et al. 1993). Distance sampling theory dictates that randomly placed points are chosen and the sighting (radial) distances (r_i) are measured from the observer to those objects detected (Buckland et al. 1993). CDS requires observers to meet the following basic assumptions: objects on the point are detected, objects do not move in response to the observer prior to detection, and distances are measured accurately. CDS allows the observer to account for objects missed because they were not detected and compensates for the fact that detectability generally decreases with increasing distance from the observer. Distance sampling allows the observer to estimate the detection function g(y) which is the probability of detecting an object given that it is at distance y from the point: $g(y)=\{\text{detection}|\text{distance } y\}$ (Buckland et al., 1993).

Using Program DISTANCE 6.0, I fit a half-normal key function to each species encountered in each burn age categories. Program DISTANCE produced several models that included a half-normal key function with varying number of adjustments. The final model was chosen based on the minimum AIC value. The model generated expected density (D) per hectare for each bird species detected in each burn age category and their probability of detection P (a).

Results

Vegetation Structure

Graminoids were the dominant cover type in 53 of the 66 points sampled. Shrubs were identified as the cover type in 13 points (Figure 2). Trees, saw palmetto, and carnivorous plants were not identified as the dominant cover type at any of the sampled points.

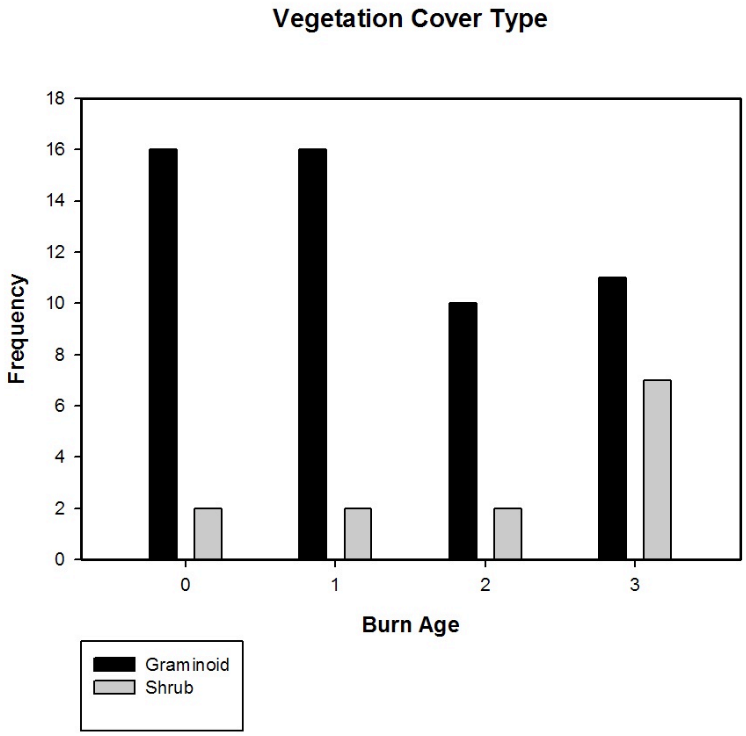

Figure 2- Cover type frequency for burn age categories 0-3.

I analyzed vegetation structure among burn age categories. Average densities for each vegetation type per burn age are summarized (Appendix). In the 0-30cm height class, graminoids had the greatest average density in the Age-2 category (257.86±77.10). In the 31-70cm height class, graminoids had the greatest average density in the Age-1 category (46.27±16.58). In the 71- over 100cm height class, graminoids had the greatest average density in the Age-1 category (4.80±4.09).

Shrubs were present in relatively high densities primarily in the 0-30cm and 31-70cm height categories. The remaining vegetation types were present in comparatively reduced densities among burn age categories and height classes. (See Table 2, Appendix)

Graminoid density

Two-way ANOVA on ranked graminoid density measures showed significant main effects of height class ($P < 0.001$) and burn age ($P < 0.001$), as well as a significant interaction between height and age ($P = 0.002$; Table 2). Overall, mean graminoid density was greater between 0-30 cm (216.54 ± 91.65 contacts) than 31-70 cm (32.46 ± 19.92) or 71 -100cm(2.29 ± 3.26) (Figure 3). Post hoc analyses using Tukey's HSD multiple comparisons test indicated that all pairwise comparisons among height classes were significantly different ($P < 0.001$). Graminoid density was lower in Age-0 plots (62.44 ±75.48 contacts) than in plots of Ages-1 (92.71 ± 104.91), 2 (86.06± 110.17), or 3(94.6 ± 137.53), but these differences were not significant on post-hoc tests. The significant interaction term arose because the greatest density between 0-30 cm was found in Age 3 plots, while the greatest densities at 31-70 cm and 71-100 cm were found in Age-1 plots (Figure 3).

Table 2. Analysis of variance results for average graminoid density among height classes and burn year.

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
HEIGHT	558253.644	2	279126.822	899.027	0.000
YEAR	13416.176	3	4472.059	14.404	0.000
HEIGHT*YEAR	6734.441	6	1122.407	3.615	0.002
Error	57748.625	186	310.476		

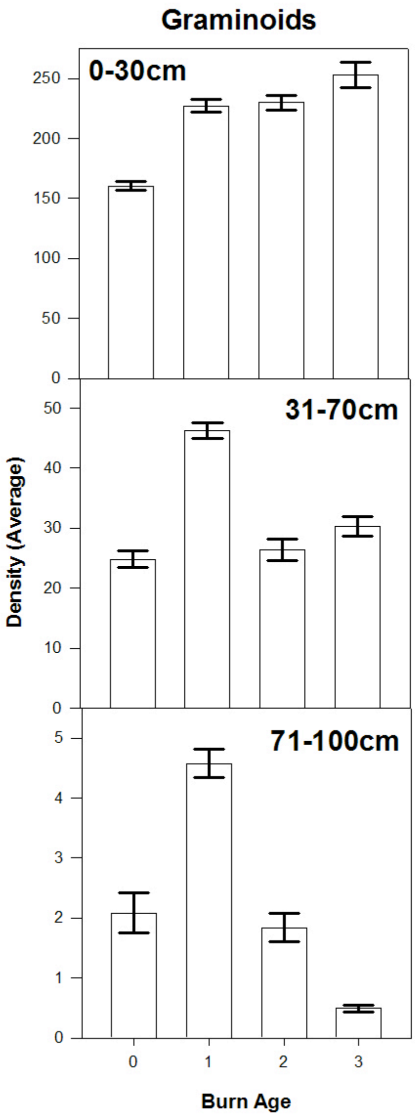

Figure 3. Average graminoid density per burn age category for each height class

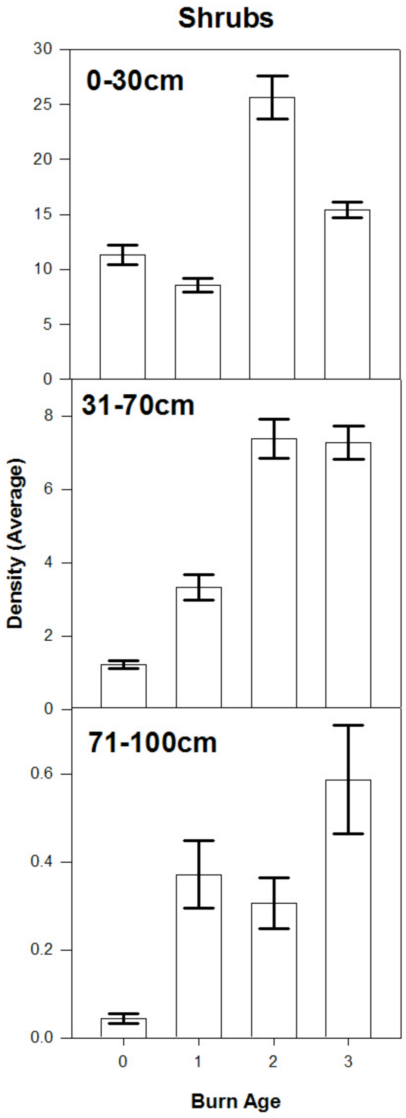

Figure 4. Average shrub density per burn age category for each height class

Shrub density

Two-way ANOVA on ranked shrub density measures showed significant main effects of height class ($P < 0.001$) and burn age ($P < 0.001$; Table 3). Mean shrub density was greater between 0-30 cm (14.27±13.33 contacts) than 31-70 cm (4.57±5.11) or 71-100cm (0.33± 1.00) (Figure 4). Post hoc analyses using Tukey's HSD multiple comparisons test indicated that all pairwise comparisons among height classes were significantly different ($P < 0.001$).

Shrub density was lower in Age-0 (4.20±8.12) than Age-1 (4.08±6.33), plots than in Age-2 (11.09±16.10) or Age-3 (7.75±8.61) plots. Post hoc tests also showed that shrub density was significantly lower in Age-0 and Age-1 plots than in Age-2 plots (P = 0.006 and P=0.005 respectively).

Table 3. Analysis of variance results for average shrub density among height classes and burn years.

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
YEAR	42919.563	3	14306.521	11.446	0.000
HEIGHT	340594.383	2	170297.192	136.247	0.000
YEAR*HEIGHT	14233.087	6	2372.181	1.898	0.083
Error	232483.944	186	1249.914		

Forb density

Two-way ANOVA on ranked forb density measures showed significant main effects of height class (P<0.001) and burn age (P=0.001; Table 4). Mean forb density was greater between 0-30cm (0.73±1.42) than 31-70cm (0.48±1.38) or 71-100cm(0.12±0.61) (Figure 5). Post hoc analyses using Tukey's HSD multiple comparisons test indicated that density in the 71-100cm height class was significantly lower than both the 0-30cm and 71-100cm (P<0.001 for both contrasts.) Forb density was lower in Age-1 (0.20±0.70) than Age-3 (0.29±0.91), Age-0 (0.5±1.07), and Age-2 (0.95±2.03). Post hoc tests showed that forb density was significantly lower in Age-1 than Age-2 (P=0.001).

Table 4. Analysis of variance results for forb densities among height classes and burn age categories.

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
YEAR	29451.216	3	9817.072	5.514	0.001
HEIGHT	49890.534	2	24945.267	14.010	0.000
YEAR*HEIGHT	10822.821	6	1803.804	1.013	0.418
Error	331170.660	186	1780.487		

Carnivorous plant density

Two-way ANOVA on ranked carnivorous plant measures showed significant main effects for height (P<0.001) and burn age (P<0.001; Table 5). Mean carnivorous plant density was greater between 0-30cm (2.18±2.27) than 31-70cm (0.26±0.54) or 71-100cm (0±0) (Figure 6). Post hoc analyses using Tukey's HSD multiple comparisons test indicated that all pairwise comparisons among height classes were significantly different: 0-30cm was greater than 31-70cm (P=0.008), and 71-100 cm was significantly less than both 0-30cm and 31-70cm (P<0.001 for both). Carnivorous plant density was lower in Age-3 (0.38±1.41) than Age-0 (0.48±1.46), Age-2 (0.86±2.09) or Age-1 (1.56±3.14). As the figure suggests Age-1 is higher in density than both Age-0 (P=0.008) and Age-3 (P=0.004).

Table 5. Analysis of variance results for carnivorous plant densities among height classes and burn age categories.

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
YEAR	33129.722	3	11043.241	7.638	0.000
HEIGHT	99549.266	2	49774.633	34.428	0.000
YEAR*HEIGHT	18188.380	6	3031.397	2.097	0.056
Error	268907.625	186	1445.740		

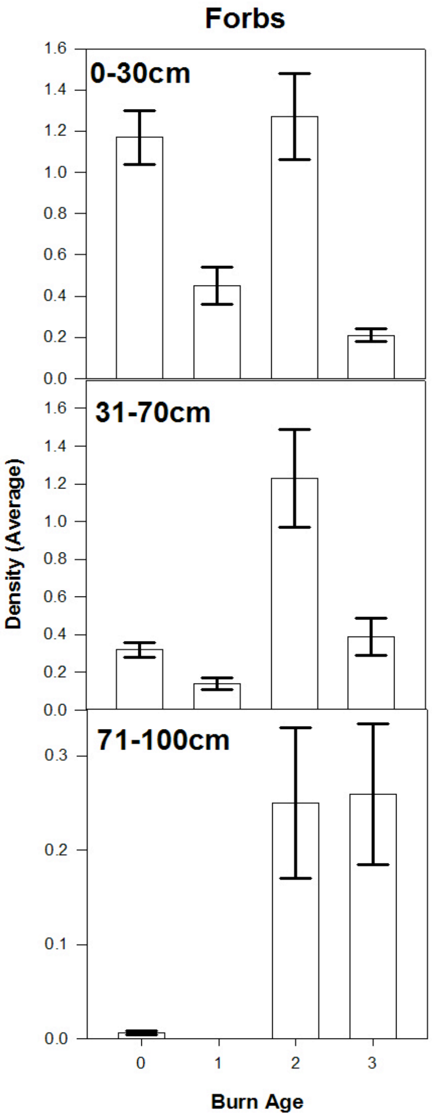

Figure 5. Average forb density per burn age category for each height class.

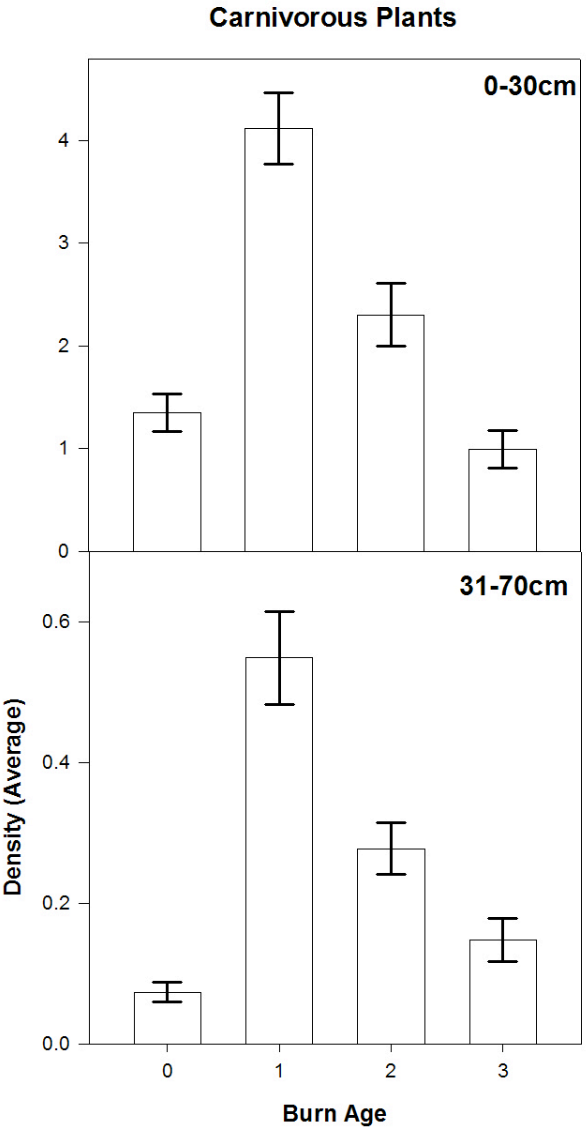

Figure 6. Average carnivorous plant density per burn age category for each height class.

Saw palmetto density

Two-way ANOVA on ranked saw palmetto measures showed a significant main effect for burn age (P=0.020; Table 6). Mean saw palmetto density was greater between 0-30cm (0.16±0.73) than 31-70cm (0.09±0.48) or 71-100cm (0.01±0.11) (Figure 7). Post hoc analyses using Tukey's HSD multiple comparisons test indicated that there were no significant differences between height classes. Saw palmetto density was greater in Age-0 (0.24±0.87) than Age-1 (0.09±0.39) and Ages-2 and 3 (0±0 for both). Post-hoc tests showed that Age-0 had a significantly greater density than Age-3 (P=0.025).

Table 6. Analysis of variance results for saw palmetto plant densities among height classes and burn age categories

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
YEAR	4169.037	3	1389.679	3.365	0.020
HEIGHT	741.358	2	370.679	0.898	0.409
YEAR*HEIGHT	2371.781	6	395.297	0.957	0.456
Error	76808.833	186	412.951		

.

Tree density

Two-way ANOVA on ranked tree measures showed significant main effects for height (P<0.001) and burn age (P<0.001; Table 7). Mean tree density was greater between 31-70cm (0.45±2.28) than 0-30cm (0.24±0.65) or 71-100cm (0.01±0.04) (Figure 8). Post hoc analyses using Tukey's HSD multiple comparisons test indicated that the 31-70cm height class was significantly greater than both 0-30cm (P=0.048) and 71-100cm (P<0.001). Tree density was greater in Age-3 (0.60±2.51) than Age-2 (0.32±0.84), Age-1 (0.03±0.11) or Age-0 (0.01±0.05). Post hoc tests also showed that Age-3 had a significantly greater density than Age-0 (P=0.001), Age-1 (P=0.018) and Age-2 (P=0.044).

Table 7. Analysis of variance results for tree densities among height classes and burn age categories

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
HEIGHT	21096.932	2	10548.466	9.689	0.000
YEAR	20912.225	3	6970.742	6.403	0.000
HEIGHT*YEAR	6744.867	6	1124.145	1.033	0.406
Error	202495.840	186	1088.687		

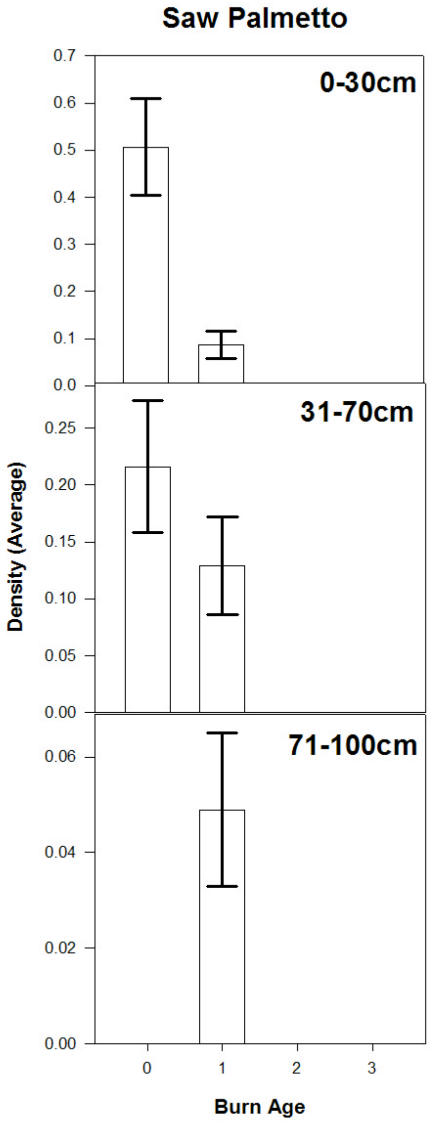

Figure 7. Average saw palmetto density per burn age category for each height class.

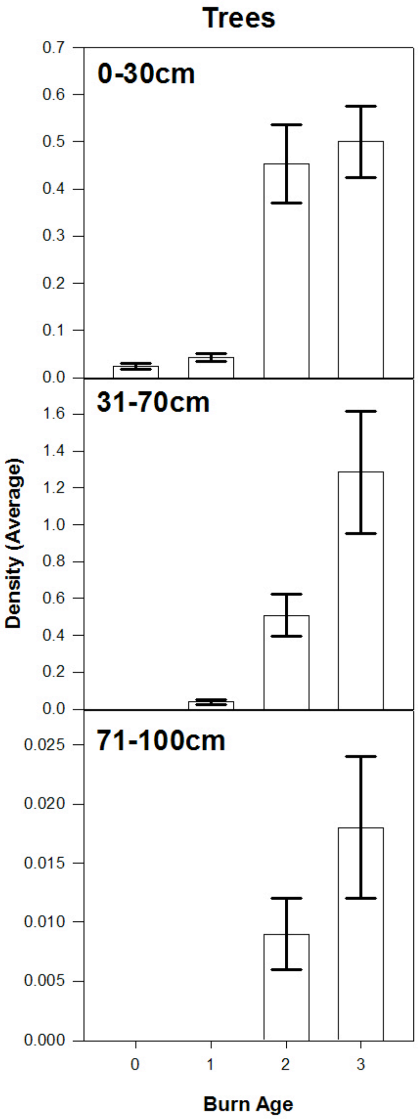

Figure 8. Average tree density per burn age category for each height class.

Bird Community-level Patterns

I detected 45 species and 627 individuals among the four burn age categories. One-way analysis of variance revealed that neither species richness nor overall abundance differed among burn years (Table 9).

Shannon's Diversity Index (H) ranged from 2.48± 0.37(Age-0) to 2.61±0.22(Age-1). Equitability (E_H) ranged from 0.57±0.08 (Age-3) to 0.67±0.01(Age-1) (Table 8). One-way ANOVA's of Shannon's diversity and equitability reveal no significant differences among any of the indices between burn ages (Table 9).

Table 8. Mean and standard deviation of four indices for bird community structure among the four burn age categories.

	Age-0	Age-1	Age-2	Age-3
Individuals	61 ± 3.61	51±13.85	59.5±4.95	57.33±6.11
Species Richness	18.33±5.86	18±4.36	19.50±3.45	18.33±4.73
H	2.48±0.37	2.61±0.22	2.58±0.31	2.51±0.39
E_h	0.61±0.09	0.67±0.01	0.63±0.06	0.57±0.08

Table 9. Results of ANOVA's on the values of Shannon's Diversity, Shannon's Equitability, Species richness, and abundance among burn years.

	SS	df	MS	F- ratio	P
Shannon's Equitability Eh	0.004	3	0.001	0.562	0.657
Error	0.016	7	0.002		
Shannon's Diversity H	0.027	3	0.09	0.082	0.967
Error	0.769	8	0.11		
Abundance	168.83	3	56.28	0.77	0.55
Error	509.17	7	72.74		
Species Richness	2.89	3	0.97	0.04	0.99
Error	163.83	7	23.41		

I analyzed the interaction between cover-type and burn age category with respect to bird abundance and species richness at each bird point-count station. The results of the two-way ANOVA for bird abundance were not significant (P=0.539) (Table 10). This indicates that cover type and time since burn are not strong predictors of bird abundance among management compartments surveyed.

Table 10. Analysis of variance for bird abundance in relation to time since burn and cover type

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
COVER	0.000	1	0.000	0.000	0.995
YEAR	25.414	3	8.471	1.156	0.334
COVER*YEAR	16.028	3	5.343	0.729	0.539
Error	424.942	58	7.327		

The results of the two-way ANOVA for bird species richness were not significant (P=0.988). (Table 11) This indicates that cover type and time since burn are not strong predictors of bird species richness among management compartments surveyed.

Table 11. Analysis of variance for bird species richness in relation to time since burn and cover type

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
COVER	1.544	1	1.544	0.429	0.515
YEAR	11.249	3	3.750	1.043	0.380
COVER*YEAR	10.304	3	3.435	0.956	0.420
Error	208.484	58	3.595		

Of the 45 species detected, 15 (33.33%) are considered migrants that use this habitat for breeding and 29 (66.67%) are residents of the longleaf pine savanna ecosystem in our study site. Migrants made up 22.33% and residents made up 77.67% of total detections. Of the species detected, 48.89% were ground foragers, 20% were foliage gleaners, 11.11% were bark gleaners, 8.89% were flycatchers, 6.67% were aerial food catchers, and 4.44% used soaring to capture food. Of the species detected, 35% were ground nesters, 32% were shrub nesters, 25% were cavity nesters, 23% were tree nesters, and <1% nested in burrows or banks.

Due to the fact that the greatest proportion of bird species were ground foragers and we surveyed birds during the breeding season to understand their habitat requirements, I tested the correlation between abundance and nest microhabitat choice among burn ages.

The results of the two-way ANOVA for bird abundance and nest microhabitat preference indicated that there were no significant differences. (Table 12)

Table 12. Analysis of Variance of nest microhabitat preferences among burn ages

Source	Sum-of-Squares	df	Mean-Square	F-ratio	P
NEST	236.792	4	59.198	1.720	0.148
YEAR	9.239	3	3.080	0.089	0.966
NEST*YEAR	226.060	12	18.838	0.547	0.881
Error	5505.636	160	34.410		

Ten species were detected in ≥20% of point-count stations, and these accounted for 65.55% of all observations. Eastern meadowlark (*Sturnella magna*) had the highest point presence (72.22%)(Table 13).

Table 13. [a] Percentage of points in which each species was detected

Common Name	Scientific Name	% [a]
Carolina wren	*Thryothorus ludovicianus*	27.78
Common nighthawk	*Chordeiles minor*	20.83
Common yellowthroat	*Geothlypis trichas*	47.22
Eastern bluebird	*Sialia sialis*	29.17
Eastern kingbird	*Tyrannus tyrannus*	27.78
Eastern meadowlark	*Sturnella magna*	72.22
Eastern towhee	*Pipilo erythrophthalmus*	43.56
Red-bellied woodpecker	*Melanerpes carolinus*	44.44
Red-shouldered hawk	*Buteo lineatus*	22.22
Tufted titmouse	*Parus bicolor*	27.78

Data from the bird point-counts were analyzed with Program DISTANCE with a half-normal key function and no additional adjustments. Of the ten bird species previously identified with a point presences of >20%, only those with >10 observations in each burn age category were analyzed in Program DISTANCE to ensure reliable estimates. For each species, the point estimate per hectare and detection probability was estimated from the model and summarized in Table 14.

Table 14. Summary of point estimate and detection probability estimates for selected bird species

Species	Point Estimate (per ha.)	SE	Detection Probability	No. of Obs.	Year
Carolina wren (*Thryothorus ludovicianus*)	0.36	0.16	98.8	10	1
Common nighthawk (*Chordeiles minor*)	0.55	0.20	92.4	19	0
Common yellowthroat (*Geothylpis trichas*)	0.96	0.30	94.6	17	1
	0.73	0.26	96.3	12	3
	0.56	0.21	97.8	14	0
Eastern bluebird (*Sialia sialis*)	0.25	0.13	99.1	10	0
Eastern kingbird (*Tyrannus tyrannus*)	1.05	0.37	92.2	11	3
Eastern meadowlark (*Sturnella magna*)	1.25	0.41	94.3	12	1
	0.33	0.08	98.5	25	2
	2.99	0.53	97.6	47	0
	1.88	1.62	99.8	37	3
Eastern towhee (*Pipilo erythrophthalmus*)	0.31	0.15	98.9	10	0
	0.59	0.22	99.5	13	1
Red-bellied woodpecker (*Melanerpes carolinus*)	0.30	0.14	97.9	17	3
	0.23	0.12	99.0	13	0
	0.04	0.02	99.9	12	2
Tufted titmouse (*Baeolophus bicolor*)	0.18	0.10	99.6	10	0

Discussion

My results suggest that compartments burned between 0-3 years previously do not differ in their ability to support a wide variety of grassland bird species. This may be due to the fact that prescribed burns at MSCNWR are done on a relatively small scale, generally one management compartment at a time, and frequently large portions of which are left unburned.

This creates a landscape mosaic and an edge around most management compartments comprised of later successional stages of longleaf pine savanna, open woodland, or forest habitat. This diverse habitat is a result of frequent, low-severity burns that increase vegetation structural complexity and encourage species richness, just as Connell (1978) predicts.

Given this, it is not surprising that a wide variety of avifauna is supported on even the newly burned management compartments. Overall, the current practice of burning units of 100-200 hectares on a 2-3 year rotation appears to be consistent with maintaining a diverse grassland bird community at MSCNWR.

Vegetation Response to Fire

Based upon the results of this study, it is clear that longleaf pine savannas at MSCNWR are dominated by graminoids (grasses and sedges) for the first three years post-fire. Regrowth is rapid, with highest graminoid density being found within 2-3 years post-burn. Still, it should be noted that vegetation surveys were carried out within 4 months on management compartments classified as Age-0 and revealed relatively high graminoid density at the 0-30cm level (160.44±43.51). While this may not provide sufficient ground cover for all ground nesting species, the correlation between high species richness on Age-0 management compartments may indicate suitable breeding habitat for a variety of bird species. Saw palmetto was found to have the greatest density at 0-30cm height-class and also in the Age-0 burn units. This is odd because saw palmettos should persist in this environment for a number of years. However, I studied patterns in compartments of differing burn ages rather than following saw palmetto dynamics in a single compartment over time, and so it is possible that differences in soil quality or water availability may have influenced this pattern. Greater study of this pattern is needed to understand the underlying factors contributing to these results. Trees followed an expected pattern of increasing in density throughout the burn ages, with the greatest density being found in Age-3. Another notable result was the high density of carnivorous plants in Age-1 burn units. This is evidence that burning benefits carnivorous plants and thus contributes to maintaining habitat diversity.

In comparison to other temperate ecosystems in North America, longleaf pine savannas have among the shorter fire-return intervals. In contrast, northeastern deciduous forests range in fire-return intervals of 5-7 years; sagebrush communities are on the order of 10-20; and tallgrass prairies 3-10 years. The shorter fire-return interval experienced by longleaf pine savannas would be expected to select for plant characteristics that permit rapid vegetation re-growth, outpacing that of other ecosystems. If this is the case, it is reasonable to expect less obvious community shifts among avifauna between 0-3 years post-fire than one would see with a longer fire return interval. Because longleaf pine savannas quickly return to a pre-burn state of relatively complete herbaceous groundcover, relatively uniform vegetation structural characteristics among early succession stages should be anticipated.

Bird response to fire

Following a growing season burn in Age-0 management compartment, the species found in greatest abundance included common yellowthroat (14), common nighthawk (*Chordeiles minor*) (19), red-bellied woodpecker (13), and eastern meadowlark (47). According to the literature cited in the Birds of North America (Poole 2005), the common nighthawk and eastern meadowlark are commonly associated with open grassland habitat. Common yellowthroats are more often associated with scrub habitat and are scrub nesters, and red-bellied woodpeckers with forest and are cavity nesters. Breeding red-bellied woodpeckers may have not have been negatively affected by fire because it either created cavities for nesting in dead trees or did not significantly alter the canopy layer. It also may indicate that groundcover type did not influence habitat choice for this species. Still, these species have managed to persist even when breeding habitat may have been significantly altered.

These results are consistent with a study by Rutledge and Conner (2002) in which the effects of groundcover restoration on breeding birds in a longleaf pine habitat were investigated. They found that species richness did not differ significantly among unburned sites and those that had been burned on a 2-year fire return interval for six years prior to the study, and on a one-year fire-return interval before that. However, relative abundance of 6 of 20 species did differ, and thus departs from the results of this study. They found that ground-nesting species, Bachman's sparrows and northern bobwhites, were more common in undisturbed sites. In my own study, northern bobwhites were only detected on sites 0-, 2-, and 3-years since last burn, and Bachman's sparrows were only detected in sites 0- and 1-year since last burn. Again, the results of my study show a lack of differentiation among habitats that differ in time since last burn. The study by Rutledge and Conner (2002) concludes that breeding bird communities do not differ significantly between native and disturbed groundcover types. They suggest that this may indicate either that avian communities remained similar despite the loss of native groundcover due to succession in undisturbed sites, or avian communities differed during early successional stages but became more similar as succession occurred. Due to a lack of historical data on the original avian communities and understory composition, it is difficult to discern the underlying causes of the similarity (Rutledge and Conner 2002).

Birds may also be responding to other characteristics of management compartments at MSCNWR. Edge effect and patch size may influence habitat preference of breeding grassland birds. Previous studies have indicated that edges may have a negative effect on grassland birds because they harbor brood parasites and predators (Winter et al. 2006, Winter et al. 2000). Whether or not the presence of a forested edge on the majority of management compartments at MSCNWR has had a positive or negative impact on grassland bird habitat preference is unknown and should be investigated further. In addition, it is necessary to understand at what habitat level grassland birds respond to management strategies. According to Winter et al. (2006), the effect of patch size on grassland bird habitat preference has not yet been identified. In addition, the authors imply that such an interaction is not likely to be consistent among all grassland birds because of regional and annual changes in bird distribution as well as a lag in response to environmental changes (Winter et al. 2006, Wiens et al. 1987).

Community-level response

In this study, bird species richness and abundance did not differ significantly among burn years. The age of the burn compartments was not a strong predictor of grassland bird species richness or abundance. There were some significant differences between vegetation height classes and among burn years for certain vegetation types. However, because there was no significant correlation between cover type and bird species richness or abundance among burn years, this suggests that birds are either not using these cues or they are not pronounced enough to affect their choice of breeding habitat at MSCNWR. This could be due in part to how prescribed burns are carried out at MSCNWR. Prescribed burns are generally between 100-200 hectares, with smaller fires occurring on smaller management units. In my study, the mean fire size was 153 ha. During a prescribed burn, backfires are set off of a fireline, usually a road, that are not as hot as the prescribed burns that are set on the interior of the management compartments (personal communication with S. Hereford). An unintended result of this protocol was that some management compartments are left with a forest edge surrounding the unit which appears to provide sufficient breeding habitat for birds that are grassland specialists as well as those that prefer shrub, scrub, or forested areas.

Although no significant difference in abundance or species richness was found among burn units differing in time since burn, this may be further proof that the current fire management meets the needs of a wade variety of grassland birds. The lack of significant difference in bird abundance among management compartments surveyed is therefore not surprising and has been seen by others. In a study by White et al. (1999), 18 sites differing in time since burn (1-, 2-, and 3-year) were compared to six sites that had not been burned in >20 years. In a comparison of the fire-excluded sites versus burned sites, 16 of 46 species showed no difference in density, 24 species declined, and 6 increased due to fire exclusion. Many species showed were significantly more abundant on the recently burned site versus the unburned, including pine warbler, prairie warbler, northern flicker, and brown-headed nuthatch. Species richness and evenness did not differ among recently burned vs. unburned sites (White et al. 1999). These results are similar to my own in that no significant difference in bird density was observed, however they depart from my results in that a noticeable decline in population density occurred in many species due to fire exclusion. However this is likely due to the fact that fire had been excluded for >20 years, in comparison to my study in which the longest fire-exclusion period was 3 years. However, both my study and the one by White et al. (1999), demonstrate how an intermediate disturbance may yield the highest diversity in longleaf pine savanna habitat. The decrease in species richness seen in their study is also consistent with the intermediate disturbance hypothesis (Connell 1978) that predicts a drop-off in species richness with the lengthening of time since burn. A study by Engstrom et al. (1984) compared bird community changes over 15 years of fire exclusion in northeast Florida loblolly (*Pinus taeda*) pine stand. Results indicated that after 15 years of exclusion, only 11 of the 43 breeding bird species originally surveyed were detected. These results also indicate that a significant reduction in disturbance frequency (>5 year interval) may contribute to decreases in species richness of grassland birds.

Species abundance estimates

The results of data analysis in Program DISTANCE indicate that bird species with greater than 20% detection rate at point count stations had detection probabilities of >92%. It is of interest that birds both typically associated with open grassland habitat, such as eastern meadowlarks, and those associated with a closed canopy forest, such as red-bellied woodpeckers, were both found in high abundance among burn treatments. Again, this may be due to the availability of edge surrounding longleaf pine savanna habitat that provides sufficient complex, structural vegetation for these birds.

Program DISTANCE estimated relative density for each species. Based upon reported territory sizes in the Birds of North America (BNA) database, one can calculate the expected density of birds for each burn age. This allows for a comparison between the estimated density produced by the model and expected densities based on observations.

Carolina wrens (*Thryothorus ludovicianus*) in Age-1 were estimated at a density of 0.36 per ha which would translate to 20 birds in this burn age category. This estimate was based on 10 observations, so the program has over-estimated the observed density. However, according to published estimates of breeding bird territories in the Birds of North America (BNA) database, Carolina wrens have a mean territories size of 4.1 ha (Haggerty and Morton 1995). Given this and based upon the total observation area for Age-1 (56.52 ha), this would yield an estimate of 14 birds or 0.25 per ha. This is smaller than the estimated density but not too different than the estimated density.

Common nighthawks had >10 observations in Age-0 and generated an estimated density of 0.55 per ha which would translate to 31 birds. This is greater than the 19 observations made for this bird. According to the BNA database, common nighthawks may have a territory size of 10.4 ha (Brigham et al. 2011). This would results in an estimate of 5.43 birds for this burn age, a much smaller density than observed or predicted. One reason for this inconsistency may be due to the fact that common nighthawk had the second largest variance in detections among burn ages of all bird species detected. This could have affected the ability of the model to make a strong prediction for density and relative abundance.

Common yellowthroats with >10 observations were observed in Age-0, -1, and -3. Their estimated densities ranged from 0.56- 0.96 per ha which would equate to 31-54 birds. These estimates are more than twice the observed densities for each burn age. According to BNA, common yellowthroat territories range from 0.2-2.9 ha (Guzy and Richison 1999). Taking the mean territory size (1.55 ha) yields an estimated density of 0.65 birds per ha. This falls just below the average generated density estimates from the model and estimates 36.46 birds in this age category. These estimates are generally in line with observed densities when one considers that the entire range of 0.25-0.96 would encapsulate the estimated density of 0.65 found in the literature. Age-0, therefore, likely had the best estimation of the true density of common yellowthroats.

Eastern bluebirds were found in their greatest density in Age-0 with an estimated density of 0.25 per ha which equates to 14.13 birds. Eastern bluebirds are thought to have territories of an average size of 2.1 ha (Gowaty and Plissner 1998) which would yield 26.91 birds in this age category which is a larger than the predicted density. Variance for the observed density among burn years was relatively high and therefore may have influenced the quality of the estimates the model was able to generate.

Eastern meadowlarks had estimated densities that ranged from 0.33-2.99 birds per ha that equates to 12.43-168.99 birds. According to Jaster et al. (2012), eastern meadowlarks typically have territories that average 2.8 ha. Given this, expected densities would range from 13-20 birds. This is a more conservative estimate than those generated by the model, although the estimate for Age-2 does fall within this range. It is possible that Age-2 burn units provided particularly favorable habitat for this species and thus explain the high abundance on these plots.

Eastern towhee's generated estimated densities of 0.31(Age-0) and 0.59(Age-1) per ha which equates to 17 and 33 birds per age class respectively. Based on the observed territories of eastern towhee's which average 1.4 ha (range: 1.2-1.6) (Greenlaw 1996), expected densities would be 40.37 birds per burn age. While this is larger than the estimates derived for Age-0, it is close to Age-1 and thus may be a viable density estimate.

Red-bellied woodpeckers were found in their greatest densities on Age-0, Age-2, and Age-3 plots and generated estimated densities of 0.3, 0.04, and 0.23 birds per ha respectively. This would yield a total of 16.96 (Age-0), 1.51 (Age-2), and 13 (Age-3) birds for each burn age. Their observed mean territory size is 8.80ha (range 1.6-16 ha) (Shackelford et al. 2000); given this, expected densities would be 6.4 birds per burn age. This is smaller than estimates generated by the model, and yet the wide range of territory sizes may in part cause this under-estimation.

Tufted titmouse was found in greatest density on Age-0 burn units and generated density estimates of 0.18 birds per ha which would equate to 10.17 birds. Breeding territories range from 3.2-5.0 ha with a mean of 4.2 ha (Grubb and Pravasudov 1994). Given this, expected densities for this burn age would be 13.46 birds. This is close to the estimated density produced by the model and only slightly larger than observed densities collected from the bird surveys.

In sum, estimated of density derived from Program DISTANCE were generally consistent with densities extrapolated from point counts and from previously published data for several bird species, namely eastern meadowlark, common yellowthroat, Carolina wren, eastern towhee, red-headed woodpecker and tufted titmouse for select burn ages. Relative density estimates diverged widely for common nighthawks and eastern bluebirds, and may be due to small sample sizes or large variances in densities across burn age categories.

Conclusions

Grassland breeding birds did not differentiate among compartments that differed in time since last burn at the Mississippi Sandhill Crane National Wildlife Refuge. Their highest total abundance was found in Age-0 and lowest in Age-2. Increased density of certain bird species such as, indigo bunting (Conner &Rutledge 2002), common yellowthroat, eastern wood-pewee, northern flicker and brown-headed nuthatch (White et al. 1999) on recently burned land provides support for the idea that prescribed burn regimes can help to create diverse habitats that support a wide variety of grassland birds. As such, the use of prescribed burns on a 2-3 year cycle should be continued to meet the needs of the greatest number of grassland birds in the longleaf pine savanna ecosystem. My study indicated that recently burned sites (Age-0) had a high proportion of species nesting and foraging within just a few months of a burn. This observation may be helpful in designing prescribed burn management protocols that seek to optimize grassland breeding bird habitat.

My study indicated that graminoids were present in the greatest densities of all vegetation types studied. This reinforces the importance of restoration and management of longleaf pine savanna groundcover. Great density of graminoids and low density of shrubs within the first two growing seasons since burn (Age-0 and -1) also show that vegetation recovery post-fire is rapid and shrub encroachment is deterred by fire.

Recommendations

MSCNWR is divided up into three major units: Fontainebleau, Gautier, and Ocean Springs. According to the MSCNWR Comprehensive Conservation Plan (USFWS 2007), the savannas of the Gautier unit experience a fire return interval of about 2 years, and the remaining units experience a 2-3 year fire return interval. This regime generally matches with the current goals and practices of the refuge. An increase in the area managed by prescribed burns would help reach their priority goal to restore longleaf pine ecosystems within the refuge. Management for birds of conservation concern should include a 2-3 year fire-return interval. This is an appropriate regime for Bachman's sparrow, Brown-headed nuthatch and northern bobwhite populations that utilize refuge habitat for breeding and have been seen to decrease in numbers after 3 or more years post-burn (Engstrom 1993).

Based upon the findings of this and previous studies, I recommend that management units at MSCNWR continue to be burned on a 2-3 year rotation, to halt the invasion of woody species and to encourage the restoration of longleaf pine savanna habitat. Due to the fact that there was no significant difference in bird species richness or abundance among management units differing in time since burn within 0-3 years, these units should act as viable breeding habitat for a wide variety of avifauna.

It is my recommendation that bird point counts and corresponding vegetation surveys be done annually at MSCNWR. Obtaining distance estimates with smaller confidence intervals will increase the likelihood of accurate interpretation of species and population-level estimates. The refuge would be advised to use three-year running averages, a time-scale appropriate for this habitat due to the fire frequency, to track changes in density. This would ensure adequate sample sizes for high quality distance analyses on a reasonable time scale. It is likely misleading to base any major management recommendations small-scale, short-term studies (Winter et al. 2006). Surveying a greater number and wider variety management compartments that vary in age since last prescribed burn is key to understanding how longleaf pine successional stages may influence breeding bird habitat selection. Studying management compartments that have not received recent prescribed burn treatment (>5 years), would likely yield different results in species richness and abundance than those found in this study.

There would likely be a shift to a woody species dominated landscape and a decrease in overall species richness. A study of this focus would allow for comparisons across successional stages and greatly inform the prescribed burn regime at MSCNWR.

Literature Cited:

1. Allen, J. C., Krieger, S. M., Walters, J. R., & Collazo, J. A. 2006. Associations of breeding birds with fire-influenced and riparian-upland gradients in a longleaf pine ecosystem. The Auk, 123(4): 1110-1128.
2. Artman, VL, Hutchinson, TF, and JD Brawn. 2005. Fire ecology and bird populations in eastern deciduous forests. Studies in Avian Biology 30:127-138.
3. Artman, VL, Sutherland, E.K., and J.F. Downhower. 2001. Prescribed burning to restore mixed-oak communities in southern Ohio: effects on breeding bird populations. Conservation Biology 14:1423-1434.
4. Askins, R.A, and Askins, R. 2002. Restoring North America's birds: lessons from landscape ecology. Yale University Press.
5. Askins, R. A., Chavez-Ramirez, F., Dale, B. C., Haas, C.A., Herkert, J. R., and Peter D. Vickery. 2007. Conservation of grassland birds in North America: Understanding Ecological Processes in different regions: “Report of the AOU Committee on Conservation. Ornithological Monographs 64: iii-viii, 1-46.
6. Barbour, M.G., Burk, J.H., Pitts, Wanna D. 1987 Terrestrial Plant Ecology. The Benjamin/Cummings Publishing Inc., Massachusetts. 182-208.
7. Beckage, Brian, Platt, W.J. and Louis J. Gross. 2009. Vegetation, Fire, and Feedbacks: a disturbance-mediated model of savannas. The American Naturalist 174(6): 805-818.
8. Boerner, Ralph E.J. 1982. Fire and Nutrient Cycling in temperate ecosystems. BioScience 32(3): 187-192.
9. Brawn, J. D., Robinson, S.K, and F. R. Thompson III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32: 251-276.
10. Brigham, R. M., Janet Ng, R. G. Poulin and S. D. Grindal. 2011. Common Nighthawk (*Chordeiles minor*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/213doi:10.2173/bna.213
11. Buckland, S.T., Anderson, D.R., Burnham, K.P., and Laake, J.L. 1993. Distance sampling: estimating abundance of biological populations. Chapman & Hall.

12. Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. Science 199(4035): 1302-1320.
13. Dickson, J. G. 2000. Fire and bird communities in the South. In The Role of Fire in Nongame Wildlife Management and Community Restoration: Traditional Uses and New Directions Proceedings of a Special Workshop (p. 52).
14. Engstrom, RT, Crawford, RL, and WW Baker. 1984. Breeding bird populations in relation to changing forest structure following fire exclusion: a 15-year study. Wilson Bulletin 96: 437-450.
15. Engstrom, RT, Vickery, PD, Perkins, DW, and WG Shriver. 2005. Effects of fire regime on birds in southeastern pine savannas and native prairies. Studies in Avian Biology 30:147-160.
16. Engstrom, R. Todd. 1993. Characteristic mammals and birds of longleaf pine forests. Proceedings of the Tall Timbers Fire Ecology Conference. Vol. 18:127-138.
17. Frost. C. C. 1998. Presettlement fire frequency regimes of the United States: a first approximation. Proceedings of the Tall Timbers Fire Ecology Conference 20:70-81.
18. Frost, C.C. 2006. History and future of the longleaf pine ecosystem. Pages 9-42 in S. Jose, E.J. Jokela, and D.L. Miller, editors. The longleaf pine ecosystem: ecology, silviculture, and restoration. Springer, New York.
19. Garren, Kenneth H. 1943. Effects of fire on vegetation of the southeastern United States. Botanical Review 9(9): 617-654.
20. Gowaty, P.A. and Jonatha H. Plissner. 1998. Eastern Bluebird (*Sialia sialis*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell lab of Ornithology; retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/381doi:10.2173/bna.381.
21. Greenlaw, Jon S. 1996. Eastern Towhee (*Pipilo erythrophthalmus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/262doi:10.2173/bna.262

22. Grubb, Jr, T. C. and V. V. Pravasudov. 1994. Tufted Titmouse (*Baeolophus bicolor*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/086doi:10.2173/bna.86
23. Guzy, Michael J. and Gary Ritchison. 1999. Common Yellowthroat (*Geothlypis trichas*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/448doi:10.2173/bna.448
24. Haggerty, Thomas M. and Eugene S. Morton. 1995. Carolina Wren (*Thryothorus ludovicianus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/188doi:10.2173/bna.188
25. Hamel, P.B., Smith, W.P., Twedt, D.J., Woehr, J.R., Morris, E., Hamilton, R.B., and R.J. Cooper. 1996. A land manager's guide to point counts of birds in the southeast. . Gen. Tech. Rep. SO-120. New Orleans, LA: U.S. Dept. Of Agriculture, Forest Service, Southern Research Station. 39 pp.
26. Jackson, JA. 1988. The southeastern pine forest and its birds: past, present and future. Bird Conserv. 3: 119-159.
27. Jaster, Levi A., William E. Jensen and Wesley E. Lanyon. 2012. Eastern Meadowlark (*Sturnella magna*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/160doi:10.2173/bna.160
28. Jose, S., Jokela, E. J., & Miller, D. L. (2006). The longleaf pine ecosystem (pp. 3-8). Springer New York.
29. Knick,ST, Holmes, AL, Miller, RF. 2005. The role of fire in structuring sagebrush habitats and bird communities. Studies in Avian Biology 30: 63-75.
30. Knopf, F.L. 1994. Avian assemblages on altered grasslands. Studies in Avian Biology 15: 247-257.
31. Komarek, E.V. 1974. Effects of fire in temperate forests and related ecosystems: Southeastern United States. In Fire and Ecosystems, eds. T.T. Kozlowski and C. Ahlgren, pp. 251-277. New York: Academic Press.

32. Loudermilk, E.L., Cropper, W.P. Jr., Mitchell, R.J., and H. Lee. 2011. Longleaf pine (*Pinus palustris*) and hardwood dynamics in a fire-maintained ecosystem: A simulation approach.
33. Mattoon, W.R. 1922. Longleaf pine. USDA Bulletin No. 1061. Washington, DC: US Government Printing Office.
34. Medin, DE, Welch, BL, and WP Clary. 2000. Bird habitat relationships along a Great Basin elevational gradient. USDA Forest Service Research Paper RMRS-RP-23. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
35. Miller, RF, Svejcar, T, and JA Rose. 2000. Impacts of western juniper on plant community composition and structure. Journal of Range Management 53:574-585.
36. Noss, R.F., E.T. LaRoe, and J.M. Scott. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Report No. 0611-R-01 (MF). National Biological Service, Washington, D.C., USA.
37. Outcalt, K.W., Sheffield, R.M. 1996. The longleaf pine forest: trends and current conditions. USDA Forest Service Southern Research Station. Resource Bulletin SRS-9.
38. Poole, A. (Editor). 2005. The Birds of North America Online: http://bna.birds.cornell.edu/BNA/. Cornell Laboratory of Ornithology, Ithaca, NY.
39. Peterjohn, B. G., & Sauer, J. R. 1999. Population status of North American grassland birds from the North American breeding bird survey, 1966-1996. Studies in Avian Biology 19: 27-44.
40. Peterson, K.L., and L.B. Best.1987.Effects of prescribed burning on nongame birds in a sagebrush community. Wildlife Society Bulletin 15:317-329.
41. Provencher, L., Gobris, N.M., and Brennan L.A. 2002. Effects of hardwood reduction on winter birds in northwest Florida longleaf pine sandhill forests. The Auk 119(1): 71-87.
42. Reinking, DL. 2005. Fire regimes and avian responses in the central tallgrass prairie. Studies in Avian Biology 30:116-126.
43. Rutledge, B. T. and L. Mike Conner. 2002. Potential effects of groundcover restoration on breeding bird communities in longleaf pine stands. Wildlife Society Bulletin 30(2): 354-360.
44. Saab, V. A., and H.D.W. Powell. 2005. Fire and avian ecology in North America: process influencing pattern. Studies in Avian Biology 30:1-13.

45. Shackelford, Clifford E., Raymond E. Brown and Richard N. Conner. 2000. Red-bellied Woodpecker (*Melanerpes carolinus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/500doi: 10.2173/bna.500
46. Shepherd, B.L., Miller, D.L., and Mack Thetford. 2012. Fire season effects on flowering characteristics and germination of longleaf pine (*Pinus palustris*) savanna grasses. Restoration Ecology 20(2):268-276.
47. U.S. Fish and Wildlife Service. 2007. Mississippi Sandhill Crane National Wildlife Refuge Comprehensive Conservation Plan. U.S. Department of the Interior Fish and Wildlife Service Southeast Region, Atlanta, GA. U.S.
48. Van Lear, D.H. and R. F. Harlow. 2002. Fire in the Eastern United States: Influence on wildlife habitat. Pp 2-10 in Ford, W. Mark; Russell, Kevin R.; Moorman, Christopher E., eds. Proceedings: the role of fire for nongame wildlife management and community restoration: traditional uses and new directions; 2000. Gen. Tech. Rep. NE-288. U.S. Department of Agriculture, Forest Service, Northeastern Research Station. Newtown Square, PA: 145 p.
49. Van Lear, D.H., Carroll, W.D., Kapeluck, P.R., and R. Johnson. 2005. History and restoration of the longleaf pine-grassland ecosystem: Implications for species at risk. Forest Ecology and Management 211:150-165.
50. Veech, Joseph A. 2006. A comparison of landscapes occupied by increasing and decreasing populations of grassland birds. Conservation Biology 20(5):1422-1432.
51. Vickery, P.D. and James R. Herkert. 2001. Recent Advances in Grassland Bird Research: Where do we go from here? The Auk 118(1): 11-15.
52. Vickery, P.D., P. L. Tubaro, J.M.C. $_{\text{da}}$Silva, B.G. Peterjohn, J.R. Herkert, and R. B. Cavalcanti. 1999. Conservation of grassland birds in the Western Hemisphere. Pages 2-26 in Ecology and Conservation of Grassland Birds in the Western Hemisphere (P.D. Vickery and J.R. Herkert, Eds.). Studies in Avian Biology, no. 19.
53. Wiens, J.A., Rotenberry, J.T., and B.V. Horne. 1987. Habitat occupancy patterns of North American shrubsteppe birds: The effect of spatial scale. Oikos 48:132-147.

54. White, D.H., Chapman, B.R., Brunjes, I.V., Raftovich, R.V. Jr., and J.T. Seginak. 1999. Abundance and reproduction of songbirds in burned and unburned pine forests of the Georgia Piedmont. Journal of Field Ornithology 70:414-424.
55. Winter, M., Johnson, D.H., and J Faaborg. 2000. Evidence for edge effects on multiple levels: artificial nests, natural nests, and distribution of nest predators in Missouri tallgrass prairie fragments. Condor 102:256-266.
56. Winter, M., Johnson, D.H., Shaffer, J.A., Donovan, T.M., and W.D. Svedarsky. 2006. Patch size and landscape effects on density and nesting success of grassland birds. The Journal of Wildlife Management 70(1): 158-172.
57. Zimmerman, JL. 1992. Density-independent factors affecting the avian diversity of the tallgrass prairie community. Wilson Bulletin 104: 85-94.

Appendix

Table 15. Species abundances and corresponding classifications

Species (Alpha Code)	AGE-0	AGE-1	AGE-2	AGE-3	Habitat Preference	Nest Microhabitat	Migratory Status	Foraging Behavior
AMCR	0	0	0	1	Open Woodland	Tree	Resident	Ground
BACS	1	1	0	0	Open Woodland	Ground	Resident	Ground
BHCO	2	8	0	3	Grassland	Tree	Resident	Ground
BHNU	3	6	3	5	Forest	Cavity	Resident	Bark
BLGR	0	2	0	2	Open Woodland	Shrub	Migrant	Ground
BLJA	1	5	0	1	Forest	Tree	Resident	Ground
BRTH	2	2	0	2	Scrub	Shrub	Resident	Ground
CACH	1	1	2	1	Forest	Cavity	Resident	Foliage Gleaner
CARW	3	10	5	7	Open Woodland	Cavity	Resident	Ground
CHSP	1	9	0	0	Open Woodland	Shrub	Resident	Ground
COGD	0	0	0	1	Scrub	Ground	Migrant	Ground
CONI	19	0	7	5	Grassland	Ground	Migrant	Aerial
COYE	14	17	7	12	Scrub	Shrub	Resident	Foliage Gleaner
CWWI	1	0	0	0	Open Woodland	Ground	Migrant	Aerial
DOWO	2	1	2	1	Forest	Cavity	Resident	Bark
EABL	10	4	4	7	Grassland	Cavity	Resident	Ground
EAKI	5	6	7	11	Grassland	Tree	Migrant	Flycatching
EAME	47	12	25	37	Grassland	Ground	Resident	Ground
EATO	10	13	4	9	Scrub	Ground	Resident	Ground
EAWP	3	2	0	0	Forest	Tree	Migrant	Flycatching
GCFL	2	2	5	7	Open Woodland	Cavity	Migrant	Flycatching
INBU	5	1	0	0	Open Woodland	Shrub	Migrant	Foliage Gleaner
MODO	1	0	3	2	Open Woodland	Tree	Resident	Ground
NOBO	1	0	2	3	Grassland	Ground	Resident	Ground
NOCA	4	3	5	3	Open Woodland	Shrub	Resident	Ground
NOFL	0	0	0	1	Open Woodland	Cavity	Resident	Ground
NOMO	0	1	3	1	Shrub	Shrub	Resident	Ground
NRWS	0	3	2	1	Banks/Gorges	Burrows	Migrant	Aerial
OROR	0	5	2	1	Open Woodland	Tree	Migrant	Foliage Gleaner
PIWA	3	2	0	4	Forest	Tree	Resident	Bark
PIWO	1	2	1	2	Forest	Cavity	Resident	Bark
PRAW	0	2	0	0	Open Woodland	Shrub	Migrant	Foliage Gleaner
RBWO	13	8	12	17	Forest	Cavity	Resident	Bark
RHWO	2	0	1	1	Open Woodland	Cavity	Resident	Flycatching
RSHA	1	8	4	7	Forest	Tree	Resident	Soaring
RTHA	0	1	1	3	Open Woodland	Tree	Resident	Soaring
RWBL	0	0	1	0	Marsh	Shrub	Resident	Ground
SACR	2	0	0	0	Marsh	Ground	Resident	Ground

Table 15.
cont.

SEWR	6	3	0	7	Marsh	Ground	Migrant	Ground
SUTA	1	1	1	0	Open Woodland	Shrub	Migrant	Foliage Gleaner
TUTI	10	5	7	2	Forest	Tree	Resident	Foliage Gleaner
WITU	1	0	0	0	Open Woodland	Ground	Resident	Ground
WOTH	1	0	1	0	Forest	Tree	Migrant	Ground
YBCH	2	6	2	5	Scrub	Shrub	Migrant	Foliage Gleaner
YBCU	2	1	0	0	Open Woodland	Shrub	Migrant	Foliage Gleaner

Table 16. Vegetation average density and standard deviation for each burn age category.

		0-30cm		31-70cm		71-100cm	
		Average	SD	Average	SD	Average	SD
Graminoid	0	160.44	43.51	24.80	18.01	2.09	4.28
	1	227.28	65.96	46.27	16.58	4.58	2.91
	2	229.94	63.92	26.42	18.03	1.84	2.50
	3	252.98	135.07	30.33	20.33	0.49	0.73
Shrub							
	0	11.33	11.06	1.23	1.37	0.04	0.13
	1	8.56	8.26	3.33	4.39	0.37	0.98
	2	25.61	20.56	7.38	5.45	0.31	0.59
	3	15.39	8.90	7.27	5.72	0.59	1.56
Saw Palmetto							
	0	0.51	1.31	0.22	0.74	0.00	0.00
	1	0.09	0.37	0.13	0.55	0.05	0.21
	2	0.00	0.00	0.00	0.00	0.00	0.00
	3	0.00	0.00	0.00	0.00	0.00	0.00
Tree							
	0	0.02	0.08	0.00	0.00	0.00	0.00
	1	0.04	0.11	0.04	0.16	0.00	0.00
	2	0.46	0.86	051	1.17	0.01	0.03
	3	0.50	0.96	1.29	4.22	0.02	0.08
Carnivorous							
	0	1.35	2.32	0.07	0.17	0.00	0.00
	1	4.12	4.42	0.55	0.84	0.00	0.00
	2	2.30	3.22	0.31	0.28	0.00	0.00
	3	0.99	2.34	0.15	0.39	0.00	0.00
Forb							
	0	1.17	1.60	0.32	0.50	0.01	0.03
	1	0.45	1.11	0.14	0.42	0.00	0.00
	2	1.27	2.15	1.34	2.65	0.25	0.83
	3	0.21	0.34	0.39	1.25	0.26	0.95

Vita

The author was born in Los Angeles, California. She obtained her Bachelor's degree from Tulane University in 2008. In August of 2011, Elizabeth joined the University of New Orleans biology graduate program to pursue a Master of Science in Biological Sciences under the advisement of Dr. Jerome J. Howard which she completed in 2013.