

WILDLIFE AND HABITAT RESPONSES TO PRESCRIBED BURNING, ROLLER
CHOPPING, AND GRAZING OF FLORIDA RANGELAND AND PASTURE

By

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To my Mum and Dad

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Abstract of Dissertation Presented to the Graduate School
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In Florida, prescribed burning and roller chopping are management practices widely promoted under assistance-based programs to improve rangeland livestock forage and wildlife habitat conditions. In addition, many assistance-based programs in Florida also promote management activities that intend to improve monoculture and mixed pasture habitats for the benefit of wildlife. However, there is a lack of information concerning the impacts of these treatments and little is known of the role grazing lands play in providing habitat for wildlife, impeding management and conservation efforts. This study was designed to fill knowledge gaps regarding the response of native rangeland (pine flatwoods) vegetative, avian, and invertebrate communities to prescribed burning and roller chopping. It also examined vegetation and bird community responses to grazing of monoculture and mixed pastures to provide information of use in tailoring management programs to sites where avian conservation is a priority. Roller chopping, particularly during the growing season, was effective at reducing shrub cover, height, and density in pine flatwoods. However, burning and roller chopping practices frequently resulted in decreases in herbaceous vegetation. Burning and roller chopping treatments led to reductions in arthropod familial richness.

Total arthropod abundance was lower on all but growing season roller chop sites. Depending on treatment and season of application, reductions in familial richness and abundance were observed in a variety of individual arthropod orders. Dormant season burning resulted in decreases in non-breeding, overwintering avian species richness and abundance. However, growing season burning resulted in increases in species richness and abundance for this avian group. Growing season roller chopping resulted in increases in abundance of permanent resident and breeding avian species. On monoculture pasture, an increase in grazing intensity led to declines in total avian species richness and short-distance migrant, neo-tropical migrant, and permanent resident avian species richness and abundance. Declines in total avian species richness and neo-tropical migrant avian species richness and abundance were observed on mixed pastures subject to increasing grazing intensity. However, short-distance migrant and urban avian species richness and grassland avian abundance increased on this pasture type in the presence of grazing.

CHAPTER 1 INTRODUCTION

Florida is an ecologically diverse state with a climate ranging from temperate to subtropical and is home to a wide diversity of flora and fauna, including 425 bird, 184 reptile and amphibian, 75 mammal, 126 fish, and 3,500 vascular plant species (Ewell 1990, Cox et al. 1994). The state ranks fourth in the contiguous United States in terms of overall species richness (Noss and Peters 1995), having more species than any other state east of the Mississippi River (Ewell 1990).

Unfortunately, rapid population growth poses a pervasive threat to the future of Florida's ecosystems and wildlife. In 2000, the state was ranked fourth in the United States in population size, being home to 15,982,378 people (United States Census Bureau [USCB] 2000), and its population density was approximately double that of the most populous state, California (USCB 2000). In 2004, Florida's human population had exceeded 17,000,000 (USCB 2000), an increase of over 15,600,000 from 1936 (Florida Biodiversity Task Force 1993). By 2030, it is projected the state's population will have undergone a further 80% increase totaling approximately 28.7 million (USCB 2000). One consequence of past and predicted increases in population size has been residential, commercial, and industrial development. This development has resulted in the loss, degradation, and fragmentation of wildlife habitats, causing the disruption of ecological connectivity and disturbance to local and landscape-level ecological functions (e.g., wildlife movements, fire regimes, hydrology, and sediment movement; Florida Fish and Wildlife Conservation Commission [FWC] 2005). The actions of Florida's residents and landowners (e.g., road construction, alterations in fire regimes, changes in grazing and other land management practices, introduction of exotic plants and animals, and

degradation of water resources) place additional pressures on the state's ecosystems (e.g., Melaluca [*Melaleuca brevifolia* Turcz], Japanese climbing fern [*Lygodium japonicum* (Thunb.) Sw.], Brazillian pepper [*Schinus terebinthifolius* Raddi]; Myers and Ewel 1990, FWC 2005, Gordon et al. 2005).

One hundred and eighteen animal species, many of which are associated with habitats undergoing considerable declines in quantity and condition because of the stresses described above, are state listed as endangered, threatened, or of special concern by FWC (2005). Fifty-seven of these animals are also federally listed as endangered or threatened (United States Fish and Wildlife Service [USFWS] 2006). In an assessment of ecosystem risks in the United States based on development pressures, endangered habitats, and imperiled species, Florida was ranked first in the "extreme risk" category out of 10 states (Noss and Peters 1995).

Like other terrestrial habitats in the state, Florida's rangelands and pastures are under threat of destruction, degradation, and fragmentation with several of high conservation priority because of declines in quantity and quality (FWC 2005, Gordon et al. 2005). These declines can partly be attributed to suburban, urban, and industrial development (FWC 2005, Gordon et al. 2005). In recent years, the demand for suitable development lands has risen dramatically, with the highest burden being placed on coastal and upland habitats (FWC 2005), sites usually considered most desirable or appropriate for construction. In addition to development, conversion to intensive agriculture (e.g., monoculture pasture, sugar cane plantings, citrus groves, pine plantations, and row crops), changes in land management techniques (e.g., alteration of natural fire regimes and increases in grazing pressure), and establishment of exotic

plants (Myers and Ewel 1990, FWC 2005, Gordon et al. 2005) pose additional threats to the persistence and integrity of remaining rangeland and pasture areas.

Although rangeland and pasture problems are complex, because these areas potentially occupy more than 8 million hectares of the state, their maintenance and improvement is key to the persistence of more than 140 associated wildlife species considered to be of greatest conservation need (Millsap et al. 1990, FWC 2005, Gordon et al. 2005).

The majority of Florida's land area is characterized as nonfederal rural lands (United States Department of Agriculture Natural Resource Conservation Service [USDA-NRCS] 1997), with a large proportion privately owned (FWC 2005). The Florida Fish and Wildlife Conservation Commission believes the key to conserving many of Florida's native species involves maintaining or enhancing habitats that currently exist on private lands. They hope this can be partly achieved through the implementation of non-regulatory, assistance-based activities. In conjunction with other state and federal agencies such as the USDA, FWC is utilizing assistance-based programs to encourage private landowners to employ land management practices of potential benefit to wildlife and their habitats. Many assistance-based programs provide technical and financial assistance for the implementation of such management activities. These include USDA Farm Bill programs such as the Environmental Quality Incentives Program and Wildlife Habitat Incentives Program.

In Florida, assistance-based programs are being used to facilitate the maintenance and improvement of a variety of native rangeland habitats. However, few quantitative data exist detailing plant, invertebrate, and vertebrate population and

community response to the management practices promoted by such programs (Berkland et al. 2005, FWC 2005, Gray et al. 2005). In Florida, a lack of knowledge as to how management activities endorsed by assistance-based programs impact rangeland vegetation, livestock forage, wildlife, and habitat may hinder implementation. If landowners are to successfully utilize these management practices and their promotion under assistance-based programs is to be fully achieved, we need a clearer understanding of how they affect rangeland vegetative communities and wildlife. In this dissertation, I quantified the effects a number of land management practices have on central and south Florida's native rangeland habitats and associated wildlife. In particular, I examined the response of native rangeland (pine flatwoods) vegetative, avian, and invertebrate communities to prescribed burning and roller chopping, 2 management activities promoted through assistance-based programs across the state.

In addition, I investigated wildlife-habitat associations on monoculture and mixed (also known as improved and semi-improved) pasture. In order to meet livestock production demands, large areas of native rangeland have been converted to monoculture or mixed pasture. Many assistance-based programs promote management activities that intend to improve these altered rangeland habitats to the benefit of wildlife (e.g., brush management, prescribed burning, prescribed grazing, brush control, and fencerow maintenance). However, we know little of the role these grazing lands play in providing habitat for wildlife in Florida, hindering management and conservation efforts. If we are to apply management practices of benefit to Florida's native species to these lands and subsidize the most appropriate management activities under assistance-based programs, we need to identify features of these non-native

rangeland habitats important to wildlife. I examined vegetation and avian community response and avian-habitat associations on monoculture and mixed rangelands subject to various livestock grazing densities in an attempt to identify habitat characteristics important to avian conservation and of use in tailoring management programs.

CHAPTER 2 WILDLIFE HABITAT EFFECTS OF PRESCRIBED BURNING AND ROLLER CHOPPING IN PINE FLATWOODS

Introduction

Pine flatwoods occur throughout the southeastern coastal plain of the United States and formerly covered approximately 50% of the land area of Florida (Abrahamson and Hartnett 1990). Unfortunately, large areas of this pine savanna habitat currently exist in a highly degraded state (Means 1996). Historically, flatwoods habitats were maintained by frequent, low intensity, lightning-ignited fires during the May–July thunderstorm season (Komarek 1968, Abrahamson and Hartnett 1990, Pyne et al. 1996). However, over the past 50 years on much of Florida’s pine flatwoods, fire suppression, reductions in fire frequency, or a shift in fire season, commonly a result of human intervention, have resulted in excessive shrub growth and proliferation of species such as saw palmetto (*Serenoa repens* [Bartram] Small), gallberry (*Illex glabra* [L.] A. Gray), wax myrtle (*Morella cerifera* [L.] Small), and fetterbush (*Lyonia lucida* [Lam] K. Koch). These increases in shrub dominance have resulted in the loss of many grass and forb species and declines in the otherwise species rich herbaceous ground layer of many pine flatwoods habitats (Wade et al. 1980, Huffman and Blanchard 1991, Glitzenstein et al. 1995). Such changes threaten the integrity of remaining pine flatwoods and their suitability as habitat for many wildlife species of conservation concern (Florida Fish and Wildlife Conservation Commission [FWC] 2005). These changes have also resulted in declines in forage quantity and quality, potentially restricting the use of these areas for livestock production (Moore 1974).

Today, most landowners and managers of pine flatwoods are faced with the challenge of habitat maintenance and improvement. In many instances, this is being

achieved through the reintroduction of prescribed burning and application of mechanical treatments (e.g., roller chopping). Many landowners and managers in Florida are cattle ranchers, whose goal is to reduce shrubs and increase the growth and production of more palatable grasses and forbs as livestock forage (Yarlett 1965, Moore 1974, Kalmbacher and Martin 1984, Tanner et al. 1988). Concomitantly, the majority of wildlife species that occupy pine flatwoods habitats benefit from increases in grass and forb cover, as they provide diverse food, cover, and other habitat resources (Huber and Steuter 1984, Madden et al. 1999, FWC 2005). However, limited numbers of understory trees and shrubs are also an integral component of wildlife habitat and their maintenance at low levels is typically desirable.

The United States Department of Agriculture, in collaboration with FWC, is currently using the Environmental Quality Incentives Program (EQIP) and Wildlife Habitat Incentives Program, among others, to encourage private landowners to maintain and restore wildlife habitat. These programs are providing financial and technical assistance for landowners to implement management practices that reduce shrub and understory hardwood cover in pine flatwoods, with the intention of increasing herbaceous plant growth. Practices promoted under assistance programs include prescribed burning and roller chopping during dormant (November–March) and growing (April–October) seasons.

Prescribed burning and roller chopping can reduce shrubby vegetation and promote growth of herbaceous groundcover species in southeastern rangeland habitats (Wade et al. 1980, Kalmbacher and Martin 1984, Tanner et al. 1988, Robbins and Myers 1992, Glitzenstein et al. 1995, Watts and Tanner 2003, Watts et al. 2006),

potentially improving their quality for wildlife. However, the seasonal effects and benefits of these treatments are unclear. Studies detailing and comparing the habitat effects of these practices, particularly dormant and growing season roller chopping, are lacking. Research conducted has typically been extremely localized, being confined to a single study area. If we are to make general recommendations on the use of these practices to individuals managing pine flatwoods habitat across the state, there is a need for detailed research that evaluates vegetation response to prescribed burning and roller chopping practices over a larger area. In addition, we know little regarding how livestock management, a dominant land-use in many pine flatwoods, alters the effects of prescribed burning and roller chopping on vegetation and wildlife habitat. A lack of understanding of how season of prescribed burning and roller chopping impacts pine flatwoods vegetation, livestock forage, and wildlife habitat in grazed and nongrazed areas could hinder practice implementation and may limit landowner participation in assistance-based management programs. Therefore, detailed research that considers vegetative community and wildlife habitat response to prescribed burning and roller chopping practices in pine flatwoods in varying seasons is required.

The objectives of my study were to 1) fill recognized gaps in our understanding of how prescribed burning, roller chopping, and combinations of the 2 practices, effect native pine flatwoods vegetation structure and composition and wildlife habitat, and 2) explore whether grazing interacts with prescribed burning and roller chopping to effect vegetation and habitat.

Methods

Study Sites

I conducted my study in privately- and publicly-owned pine flatwoods across a 6-county area (Desoto, Highlands, Lee, Manatee, Osceola, and Sarasota) of central and south Florida. In these flatwoods, I established 50-paired treatment and control sites with varying management (i.e., prescribed burning and roller chopping) histories and grazing regimes. When grazed, both the treatment and paired control study sites were subject to similar grazing pressures at similar times. During the study, local landowners and managers prescribed burned and roller chopped these pine flatwoods sites using varying, individual protocols.

Florida's pine flatwoods are characterized as having a pure or combined overstory stand of scattered slash (*Pinus elliotti* Engelm.) and longleaf (*P. palustris* Mill.) pine. The understory and shrub layer includes saw palmetto, gallberry, wax myrtle, staggerbush (*Lyonia fruticosa* [Michx.] G.S. Torr.), dwarf huckleberry (*Gaylussacia dumosa* [Andrews] Torr. & A. Gray), dwarf live oak (*Quercus mimima* [Sarg.] Small), and tarflower (*Befaria racemosa* Vent.). When the shrub and understory layer is relatively open, an often diverse herbaceous layer exists. This layer contains a wide variety of grasses (e.g., *Agrostis* spp., *Andropogon* spp., *Aristida* spp., *Eragrostis* spp., *Panicum* spp., and *Paspalum* spp.). Common forbs include legumes (e.g., *Cassia* spp., *Crotalaria* spp., *Galactia* spp., *Tephrosia* spp.), milkweeds (*Asclepias* spp.), milkworts (*Polygala* spp.), and a wide variety of composites (e.g., *Aster* spp., *Chrysopsis* spp., *Eupatorium* spp., *Liatris* spp., and *Solidago* spp.; Abrahamson and Hartnett 1990, United States Fish and Wildlife Service 1999).

Treatment Types

Treatment types included dormant season (November–March) burn, growing season (April–October) burn, dormant season roller chop, growing season roller chop, and a roller chop/burn combination. The roller chop/burn combination (hereafter referred to as roller chop/burn) involved roller chopping in the dormant season followed by burning within 6 months. I established a total of 11 dormant season burn, 9 growing season burn, 9 dormant season roller chop, 12 growing season roller chop, and 9 roller chop/burn sites, each paired with an adjacent untreated control.

Habitat Sampling

I assessed the effects of management treatments (i.e., prescribed burning, roller chopping, and combinations of the two) on pine flatwoods habitat characteristics using a paired-sample approach, where plant community composition and structure and ground layer variables were compared between sampling points randomly located within paired treated (e.g., burned) and untreated sites. Sampling points in untreated sites were randomly located adjacent to treated sites, and were of similar current and past management (e.g., grazing intensity), surrounding land-use, plant community (e.g., overstory cover), and soil conditions, and were located in the same pasture or management unit. Within each site, 1 randomly selected treatment or control sampling point was established. Sampling points that occurred within 50 m of the edge of a site were rejected and randomly relocated to minimize edge effects. Sites, within which treatment and control sampling points were located, ranged from 2–20 ha in size.

I assessed habitat characteristics at each sampling point once in winter (January–March), spring (April–May), and summer (June–September), during each of 2 years (2007–2008) following treatment. At each point, plant community composition and

structure, litter and soil variables, and vertical obstruction were examined in several strata (i.e., ground, herbaceous, shrub, understory, and overstory levels), using a 0.03 ha nested circular plot design similar to that described by Dueser and Shugart (1978) and Higgins et al. (2005).

Ground layer. I determined litter cover (%; ocular estimate) within 4 1-m² subsample plots, 1 randomly located in each quadrant of the 0.03-ha circular plot, along with soil density (g/cm³), moisture (%), and pH. Litter cover was recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Hays et al. 1981, Higgins et al. 2005). Soil density was measured as the dry weight density (g/cm³) of a 5-cm diameter 10-cm deep soil core sample after oven drying at 45°C for 48 hours (Donhaue et al. 1971). I used a Kelway soil tester to measure soil pH and moisture (Rodewald and Yahner 2001).

Herbaceous layer. I measured species richness (no. of species), cover (%; ocular estimate), and maximum height (cm) of forbs and graminoids within the 1-m² subsample plots. Forb and graminoid cover were recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Hays et al. 1981, Bullock 1996, Krebs 1999, Higgins et al. 2005).

Vertical obstruction. Vertical obstruction (%) from 0–2 m above ground was measured using a cover pole (Griffin and Youtie 1988) centered on the 0.03-ha circular plot at a distance of 5 and 10 m.

Shrub layer. I counted and measured the height of all shrubs (woody vegetation <2.0 m in height) in 2 perpendicular 20-m² quadrats centered on the 0.03-ha plot to estimate species richness (no. of species), density (no./m²), and maximum height (cm)

for individual species and all combined (Hays et al. 1981, Krebs 1999, Higgins et al. 2005). Shrub cover (%) was assessed along 2 perpendicular 20-m transects centered on the 0.03-ha circular plot using the line intercept method (Hays et al. 1981, Higgins et al. 2005).

Understory. I counted, identified the species, and measured the diameter at breast height (dbh) of all understory (woody vegetation <7.5 cm dbh, ≥ 2.0 m in height) plants within the 0.03-ha circular plots to estimate species richness, density (no./ha), and basal area (cm^2/ha) for individual species and all combined (Krebs 1999). Understory canopy cover (%) was estimated from 41 evenly spaced, vertical ocular tube sightings taken at a height of 0.75 m along 2 perpendicular 20-m transects centered on the 0.03-ha circular plot (James and Shugart 1971).

Overstory. All overstory (woody vegetation ≥ 7.5 cm dbh) plants were also counted, species identified, and dbh measured within the 0.03-ha circular plots to estimate species richness, density (no./ha), and basal area (cm^2/ha) for individual species and all combined (Krebs 1999). I estimated overstory canopy cover (%) from 41 evenly spaced, vertical ocular tube sightings taken at a height of 1.5 m along 2 perpendicular 20-m transects centered on the 0.03-ha circular plot (James and Shugart 1971).

I recorded the presence of livestock on study sites for incorporation into analyses.

Analyses

I used repeated measures mixed model regressions, followed by Fisher's Protected LSD tests, to examine differences in habitat characteristics (e.g., understory cover, forb richness, graminoid cover) between untreated (control) and treated sites, both within (e.g., dormant season burn) and among (i.e., dormant season burn, growing

season burn, dormant season roller chop, growing season roller chop, and roller chop/burn) treatment types. Repeated measures were time since treatment (time) and season. Study site pair was included as a blocking factor and presence of grazing as an additional influential independent variable. In my results and discussion, I focused on treatment rather than repeated measures or grazing effects. Two-, 3-, and 4-way treatment effect interactions were noted in the results, if they occurred. Due to the timing of data collection, it was not possible to test for 3-way interactions for growing season burning and roller chopping treatments. As 3- and 4-way treatment interaction effects are difficult to reliably interpret, they were not discussed further (Zar 1999). In the case of two-way treatment interactions, when differences in linear combinations of groups or biologically meaningless comparisons (e.g., shrub cover in growing burn sites in year 1 versus shrub cover in control sites in year 2) arose, I stated that post hoc comparisons revealed no differences based on treatment and the interacting factor.

I rank transformed all data sets prior to analyses due to violations of normality and homogeneity of variance assumptions (Conover 1998, Zar 1999, SYSTAT 2007). Statistical significance was concluded at $P \leq 0.1$ for all tests. I used this value rather than the more common $P \leq 0.05$, to minimize the probability of making a Type II error (Mapstone 1995, Zar 1999). All statistical tests were performed using SYSTAT (2007) statistical software.

Results

Dormant Season Burn

Ground layer. Variance of litter cover was affected by dormant season burning (Table 2-1). Mean litter cover and mean and variance of litter depth were affected by a dormant season burning * time interaction (Table 2-2). Dormant season burning alone

and in all combinations with time, season, and grazing had no effect on maximum, mean, and variance of soil pH, maximum, mean, and variance of soil moisture, and mean and variance of soil density ($P \geq 0.125$).

Herbaceous layer. A dormant season burning * time interaction affected variance of forb height and graminoid cover and mean and variance of graminoid height. Mean graminoid cover was also affected by a dormant season burning * time interaction. However, no differences in this habitat characteristic based on burning and time were observed from post hoc comparisons (Table 2-2). Mean graminoid cover and variance of graminoid height were affected by a dormant season burning * grazing interaction. A dormant season burning * grazing interaction also affected mean forb species richness and mean and variance of forb height. However, post hoc comparisons revealed no differences in these habitat characteristics because of burning and grazing (Table 2-3). Mean graminoid height was affected by a dormant season burning * season interaction (Table 2-4). Variance of forb species richness and mean forb cover and height were affected by a dormant season burning * time * season interaction ($P \leq 0.091$), and variance of graminoid species richness a burning * time * season * grazing interaction ($P = 0.018$). Variance of forb cover and maximum and mean graminoid richness were unaffected by dormant season burning alone and in all combinations with time, season, and grazing ($P \geq 0.234$).

Vertical obstruction. Vertical obstruction from 10 m was affected by dormant season burning (Table 2-1). Vertical obstruction at 5 m was affected by a dormant season burning * time interaction (Table 2-2).

Shrub layer. Shrub cover was affected by a dormant season burning * time interaction (Table 2-2) and dormant season burning * grazing interaction (Table 2-3). However, post hoc comparisons found no differences in shrub cover as a result of burning and grazing. Shrub height was affected by a dormant season burning * time * season interaction ($P = 0.021$). Dormant season burning alone and in all combinations with time, season, and grazing had no effect on shrub density ($P \geq 0.810$).

Understory. A dormant season burning * time interaction affected understory species richness (Table 2-2). Understory species richness and understory density were affected by a dormant season burning * grazing interaction (Table 2-3). Understory cover was unaffected by dormant season burning alone and in all combinations with time, season, and grazing ($P \geq 0.105$).

Overstory. Overstory canopy cover was affected by dormant season burning (Table 2-1). Dormant season burning alone and in all combinations with time, season, and grazing had no effect on overstory richness, density, and basal area ($P \geq 0.357$).

Growing Season Burn

Ground layer. Mean litter cover and soil pH and mean and variance of litter depth were affected by growing season burning (Table 2-1). A growing season burning * grazing interaction affected variance of soil pH (Table 2-3). Variance of soil moisture was affected by a growing season burning * time, mean soil density by a burning * grazing, and variance of soil pH by a burning * season interaction. However, post hoc comparisons found no differences in any of these habitat characteristics as a result of burning and, as appropriate, time, grazing, or season interactions (Tables 2-2, 2-3, and 2-4, respectively). Variance of litter cover was affected by a growing season burning * season * grazing interaction ($P = 0.072$). Maximum and mean soil moisture and

variance of soil density were unaffected by growing season burning alone and in all combinations with time, season, and grazing ($P \geq 0.647$).

Herbaceous layer. Growing season burning affected mean graminoid species richness and forb cover and variance of forb height (Table 2-1). Mean graminoid cover and height were affected by a growing season burning * time interaction. A growing season burning * time interaction also affected variance of forb cover. However, post hoc comparisons found no differences in this habitat characteristic based on burning and time (Table 2-2).

Mean forb height and variance of graminoid height were affected by a growing season burning * grazing interaction. However, again no differences in these habitat characteristics based on burning and grazing were observed from post hoc comparisons (Table 2-3). Variance of graminoid cover was affected by a growing season burning * time * season * grazing interaction ($P = 0.066$). Growing season burning alone and in all combinations with time, season, and grazing had no effect on maximum, mean, and variance of forb richness, and variance of graminoid richness ($P \geq 0.280$).

Vertical obstruction. Vertical obstruction from 10 m was affected by growing season burning (Table 2-1). Visual obstruction from 5 m was affected by a growing season burning * time interaction (Table 2-2).

Shrub layer. Shrub height and cover were affected by a growing season burning * time interaction (Table 2-2). Shrub species richness was affected by a growing season burning * grazing interaction, but differences in this habitat characteristic based on burning and grazing were not observed from post hoc comparisons (Table 2-3).

Shrub density was unaffected by growing season burning alone and in all combinations with time, season, and grazing ($P \geq 0.305$).

Understory. All understory habitat characteristics were unaffected by growing season burning alone and in all combinations with time, season, and grazing ($P \geq 0.220$).

Overstory. All overstory habitat characteristics were unaffected by growing season burning alone and in all combinations with time, season, and grazing ($P \geq 0.500$).

Dormant Season Roller Chop

Ground layer. Mean and variance of soil moisture were affected by dormant season roller chopping (Table 2-1). A dormant season roller chopping * time interaction affected mean and variance of litter depth (Table 2-2). Variance of litter depth and mean soil pH were affected by a dormant season roller chopping * grazing interaction. However, post hoc comparisons revealed no differences in shrub density based on roller chopping and grazing (Table 2-3). Dormant season roller chopping alone and in all combinations with time, season, and grazing had no effect on mean and variance of litter cover, variance of soil pH, and maximum, mean, and variance of soil density ($P \geq 0.200$).

Herbaceous layer. Mean forb species richness, cover, and height and mean graminoid cover were affected by a dormant season roller chopping * time interaction. However, for none of these habitat characteristics did post hoc comparisons reveal differences based on roller chopping and time (Table 2-2). A dormant season roller chopping * grazing interaction affected variance of graminoid cover, but post hoc comparisons revealed no differences in this habitat characteristic as a result of roller

chopping and grazing (Table 2-3). A dormant season roller chopping * season interaction affected mean graminoid height (Table 2-4). Variance of forb cover and graminoid species richness were affected by a dormant season roller chopping * season * grazing interaction ($P \leq 0.084$) and variance of forb height by a dormant season roller chopping * time * season * grazing interaction ($P = 0.080$). Variance of forb richness, maximum and mean graminoid richness, and variance of graminoid height were unaffected by dormant season roller chopping alone and in all combinations with time, season, and grazing ($P \geq 0.203$).

Vertical obstruction. A dormant season roller chopping * time interaction affected vertical obstruction from 5 m and 10 m (Table 2-2). Visual obstruction from 5 m was also affected by a dormant season roller chopping * grazing interaction. However, post hoc comparisons revealed no differences in this habitat characteristic based on roller chopping and grazing (Table 2-3).

Shrub layer. Shrub height was affected by dormant season roller chopping alone (Table 2-1). A dormant season roller chopping * time interaction affected shrub density and cover. However, post hoc comparisons revealed no differences in shrub density based on roller chopping and time (Table 2-2). Shrub species richness was affected by a dormant season roller chopping * grazing interaction, but no differences in this habitat variable based on roller chopping and grazing were observed from post hoc comparisons (Table 2-3).

Understory. All understory habitat characteristics were unaffected by dormant season roller chopping alone and in all combination with time, season, and grazing ($P \geq 0.316$).

Overstory. All overstory habitat characteristics were unaffected by dormant season roller chopping alone and in all combinations with time, season, and grazing ($P \geq 0.673$).

Growing Season Roller Chop

Ground layer. Mean and variance of litter depth were affected by growing season roller chopping (Table 2-1). A growing season roller chopping * time interaction affected variance of soil pH and mean soil density. Post hoc comparisons found no differences in mean soil density based on roller chopping and time (Table 2-2). A growing season roller chopping * grazing interaction affected mean litter cover (Table 2-3). Mean litter cover was affected by a combination of growing season roller chopping and season, but no differences in this habitat attribute based on roller chopping and season were observed from post hoc comparisons (Table 2-4). Variance of soil density was affected by a growing season roller chopping * time * season * grazing interaction ($P = 0.023$). Growing season roller chopping alone and in all combinations with time, season, and grazing had no effect on variance of litter cover, maximum and mean soil pH, and maximum, mean, and variance of soil moisture ($P \geq 0.448$).

Herbaceous layer. Mean forb height and graminoid species richness, cover, and height were affected by a growing season roller chopping * time interaction (Table 2-1). A growing season roller chopping * grazing interaction affected variance of graminoid height, but post hoc comparisons revealed no differences in this habitat characteristic as a result of roller chopping and grazing (Table 2-2). Mean graminoid height was affected by a growing season burning * season interaction (Table 2-3). Variance of graminoid species richness was affected by a growing season roller chopping * time * season * grazing interaction ($P = 0.050$). Maximum, mean, and variance of forb

richness, maximum, mean and variance of forb cover, and variance of forb height, graminoid cover, and graminoid height were unaffected by growing season roller chopping alone and in all combinations with time, grazing, and season ($P \geq 0.107$).

Vertical obstruction. Vertical obstruction from 5 m and 10 m were affected by a growing season roller chopping * time interaction (Table 2-2).

Shrub layer. Growing season roller chopping affected shrub species richness (Table 2-1). Shrub density and cover were affected by a growing season roller chopping * grazing interaction. A growing season roller chopping * grazing interaction also affected shrub height. However, no differences in this habitat characteristic based on roller chopping and grazing were observed from post hoc comparisons (Table 2-2).

Understory. All understory habitat characteristics were unaffected by growing season roller chopping alone and in all combinations with time, season, and grazing ($P \geq 0.735$).

Overstory. All overstory habitat characteristics were unaffected by growing season roller chopping alone and in all combinations with time, season, and grazing ($P \geq 0.754$).

Roller Chop/Burn

Ground layer. Variance of litter depth was affected by roller chopping/burning (Table 2-1). Mean and variance of litter cover and mean litter depth were affected by a roller chopping/burning * time interaction (Table 2-2). A roller chopping/burning * grazing interaction affected variance of litter cover, mean litter depth, and soil density, but post hoc comparisons revealed no differences in these habitat characteristics based on roller chopping/burning and grazing (Table 2-3). Roller chopping/burning alone and

in all combinations with time, season, and grazing had no effect on maximum, mean, and variance of soil pH and maximum and mean soil moisture ($P \geq 0.621$).

Herbaceous layer. A roller chopping/burning * time interaction affected mean and variance of forb height. No differences in graminoid species richness based on roller chopping/burning and time were observed from post hoc comparisons (Table 2-2). A roller chopping/burning * grazing interaction affected mean forb species richness, cover, and height and mean graminoid cover and height. However, post hoc comparisons revealed no differences in any of these habitat characteristics because of roller chopping/burning and grazing (Table 2-3). Mean forb height and mean graminoid cover and height were affected by a roller chopping/burning * season interaction. Post hoc comparisons found no differences in mean forb height based on roller chopping and season (Table 2-4). Variance of graminoid height was effected by a roller chopping/burning * time * season interaction ($P = 0.024$) and variance of graminoid cover by a roller chopping/burning * time * season * grazing interaction ($P = 0.046$). Maximum and variance of forb richness, variance of forb cover, and maximum and variance of graminoid richness were unaffected by roller chopping/burning alone and in all combinations with time, season, and grazing ($P \geq 0.176$).

Vertical obstruction. Visual obstruction from 10 m was affected by roller chopping/burning (Table 2-1). A roller chopping/burning * time interaction affected visual obstruction from 5 m (Table 2-2).

Shrubs. Shrub cover was affected by roller chopping/burning (Table 2-1). A roller chopping/burning * time interaction affected shrub species richness. However, post hoc comparisons revealed no differences in this habitat characteristic based on roller

chopping/burning and time (Table 2-2). Shrub height was affected by a roller chopping/burning * grazing interaction (Table 2-3). Shrub density was unaffected by roller chopping/burning alone and in all combinations with time, season, and grazing ($P = 0.330$).

Understory. Roller chopping/burning affected understory species richness and density (Table 2-1). Understory basal area and cover were unaffected by roller chopping/burning alone and in all combinations with time, season, and grazing ($P \geq 0.150$).

Overstory. All overstory habitat characteristics were unaffected by roller chopping/burning alone and in all combinations with time, season, and grazing ($P \geq 0.363$).

Treatment Type Comparisons

Ground layer. Treatment type affected variance of litter depth (Table 2-5). A treatment type * time interaction affected mean litter cover and litter depth (Table 2-6). A treatment type * grazing interaction affected mean litter cover (Table 2-7). Mean soil moisture was affected by a treatment type * season * grazing interaction ($P = 0.087$) and variance of litter cover by a treatment type * time * season * grazing interaction ($P = 0.020$). Mean and variance of soil pH, variance of soil moisture, and mean and variance of soil density were unaffected by treatment type alone and in all combinations with time, season, and grazing ($P \geq 0.119$).

Herbaceous layer. Mean forb species richness and cover, variance of forb height, and mean graminoid species richness and height were affected by a treatment type * time interaction (Table 2-6). A treatment type * grazing interaction affected mean forb richness and cover, variance of forb height, and variance of graminoid richness and

height. However, post hoc comparisons revealed no differences in variance of graminoid height based on treatment type and grazing (Table 2-7). Mean graminoid height was affected by a treatment type * season interaction (Table 2-8). Mean forb height ($P = 0.030$) was affected by a combination of treatment type * time * season and mean graminoid cover by a treatment type * grazing * season interaction ($P = 0.062$). Treatment type alone and in all combinations with time, season, and grazing had no effect on variance of forb richness and cover and variance of graminoid cover ($P \geq 0.127$).

Vertical obstruction. Vertical obstruction from 5 m and 10 m were affected by a treatment type * time (Table 2-6) and a treatment type * grazing (Table 2-7).

Shrubs. A treatment type * time interaction affected shrub height and cover (Table 2-6). Shrub species richness, density, and height were affected by a treatment type * grazing interaction (Table 2-7).

Understory. Understory species richness, density, and cover were affected by treatment type (Table 2-5). Understory basal area was unaffected by treatment type alone and in all combinations with time, season, and grazing ($P \geq 0.169$).

Overstory. Overstory species richness, density, basal area, and cover were affected by a treatment type * grazing interaction (Table 2-7).

Discussion

I found that improvements to wildlife habitat in flatwoods, through reductions in shrub cover and potential increases in herbaceous growth, can be achieved through the application of prescribed burning and roller chopping. However, these treatments vary in the length of time they exhibit an effect on vegetation structure and composition.

The reduction in litter cover and depth that occurred on dormant and growing season burn sites into the second year post-treatment is likely beneficial for plant development. The combustion of litter during a fire can result in nutrients from ash, particularly nitrogen and phosphorus, important in the growth of many plant species, being returned to the soil (Ahlgren 1960, Hendricks et al. 2002, Christensen 2004). In addition, the presence of litter creates a physical barrier for seedling and sprout emergence and to seeds reaching the soil. As a result, litter removal can result in increased plant growth (Ahlgren 1960, Facelli and Pickett 1991, Hendricks et al. 2002, Christensen 2004). While dormant and growing season roller chopping also resulted in reductions in litter cover and depth, considerable litter remained on the soil surface. This litter may have resulted in some suppression of herbaceous regrowth (Ahlgren 1960, Facelli and Pickett 2001, Hendricks et al. 2002, Christensen 2004).

Dormant and growing season burning caused reductions in many herbaceous characteristics related to forb and graminoid richness, height, and cover. On dormant season burn sites, affected herbaceous habitat characteristics typically returned to, but not above, pre-burn levels later in the study. However, effects on herbaceous characteristics were often more prolonged on growing season burn sites. Increases in herbaceous species biomass or abundance, which might be reflected in increases in cover and height, have been observed on dormant burn compared to control sites (Fitzgerald 1990, Robbins and Myers 1992), often when there has been a reduction in litter cover and depth (Ahlgren 1960, Facelli and Pickett 1991, Hendricks et al. 2002, Christensen 2004). Growing season burning is frequently promoted as a means to restore grass and forb cover in areas invaded by hardwoods and shrubs (Glitzenstein et

al. 1995). Many studies examining the response of herbaceous pine flatwoods vegetation to growing season burning have focused on flowering and seed production. Wiregrass (*Aristida stricta* Michx.), a common flatwoods species, flowers profusely and will only produce viable seeds following growing season burning (Outcalt 1994). Cutthroat grass (*Panicum abscissum* Swallen) has also been found to flower abundantly when burned between mid-April and mid-August, but rarely flowers if burned at other times (Myers and Boettcher 1987). Little bluestem (*Schizachyrium rhizomatum* [Swallen] Gould) and other bluestem grasses (*Andropogon* spp.) have exhibited similar patterns (Platt et al. 1988). However, few studies have examined changes in herbaceous cover following fires in different seasons, and most information for Florida is anecdotal. Studies of tallgrass prairie suggest that growing season fires may increase grass production but decrease forb production relative to dormant season burns (Towne and Owensby 1984). In Oklahoma pine flatwoods communities, growing season burning increased herbaceous species richness and diversity and forb abundance (Sparks et al. 1998). That increases in herbaceous cover above pre-burn levels did not occur on dormant or growing season burn sites during this study may be a result of its short duration. However, as understory and shrub density and cover were not significantly reduced, competition for resources may have remained, restricting forb and graminoid growth.

A number of forb and graminoid characteristics only exhibited reductions following dormant and growing season burning when grazed. Certain native pine flatwoods grasses, including creeping (*Schizachyrium scoparium* [Michx.] Nash var. *stoloniferum* [Nash] Wipff) and chalky bluestem (*Andropogon capillipes* Nash) and wiregrass, have

been found to decline when grazed immediately following burning (White and Terry 1979, Sievers 1985). The attraction of livestock to new herbaceous growth produced following a fire (Hilmon and Hughes 1965) may have been responsible for a lowering of graminoid characteristics during this study. Grazing deferment following burning may benefit some grass species commonly found on pine flatwoods (White and Terry 1979, Sievers 1985, Orr and Paton 1997).

While dormant season roller chopping had little effect on herbaceous habitat characteristics, growing season roller chopping and roller chop/burning, like growing season burning, caused prolonged reductions in forbs and graminoids. Again, grazing deferment is recommended to maintain the herbaceous plant community, as reductions in many forb and graminoid characteristics were associated with grazing. Few studies have examined the response of herbaceous vegetation to roller chopping or roller chopping/burning. However, those that have suggest an increase in herbaceous characteristics following treatment application, with certain forbs flowering more profusely (Huffman and Werner 2000). Increases in certain grasses, including wiregrass, creeping bluestem, broomsedge (*Andropogon virginicus* L.), and lopsided Indiangrass (*Sorghastrum secundum* [Elliot] Nash) have been reported following dormant season roller chopping (Yarlett 1965, Yarlett and Roush 1970, Kalmbacher and Martin 1984). However, season of roller chopping comparisons have found no differences in yield of bluestems, other grasses, or forbs (Kalmbacher and Martin 1984). A study of Texas range found that after 2 years herbaceous cover was up to 135% greater on roller chopped than untreated sites (Bozzo et al. 1992).

The use of a single dormant season or growing season burn is not likely to be sufficient to reduce shrubs in pine flatwoods habitats, particularly when they occur at high densities and provide excessive cover, as is often the case after a period of fire exclusion. These treatments had no effect on shrub density and only minimal effect on shrub height and cover, which exhibited swift regrowth post-burn and returned to pre-treatment levels within a year. Dormant season burning may have minimal medium to long-term effects on many shrubs, particularly saw palmetto, with cover typically returning to pre-burn levels as soon as 1 year post-burn (Abrahamson 1984a, b; Fitzgerald 1990; Glitzenstein et al. 1995; Robbins and Myers 1992). Saw palmetto, a dominant shrub in pine flatwoods, is fire adapted with rhizomes that serve as large carbohydrate reserves and meristematic tissues that are protected from the flames during a burn. It is these characteristics that allow this species to resprout rapidly following a burn, regaining up to 80% of its pre-burn canopy cover after just 1 growing season (Hilmon 1968). Increases in shrub growth are often not as quick following growing season burning, as resprouting tends to be lower than for dormant season burning (Lewis and Harshbarger 1976, Fitzgerald 1990, Huffman and Blanchard 1991, Olson and Platt 1995, Drewa et al. 2002).

Shrub density tended to be lower on growing season burn than control sites when grazed. Shrubs, primarily saw palmetto, can comprise up to 20% of the diet of cattle grazing pine flatwoods in winter when grasses and forbs are scarce (Kalmbacher et al. 1984). There was a reduction in the cover of grasses and forbs following growing season roller chopping and livestock may have been forced to graze remaining shrubs, causing a reduction in their density. The effects of roller chopping/burning have not

been well studied. During this study, the majority of sites were burned after roller chopping, but it has been suggested that burning prior to roller chopping may remove some of the above ground shrub biomass making subsequent roller chopping more effective at shrub reduction (Kalmbacher and Martin 1984).

On pine flatwoods, in situations where significant, rapid shrub reduction is a primary goal, the use of alternative management techniques instead of or in addition to dormant season burning may be preferred. However, regular dormant season burning with a 2–3 year return interval, where fuel loads permit, will keep saw palmetto crown size small and can slow the spread of this species in situations where it is not dominant (Hilmon and Lewis 1962, Hilmon and Hughes 1965). A fire return interval of 4 years or more will typically result in cover increases of 2–3% per year (Hough 1968, Moore 1974). While not a suitable management activity in areas of high shrub growth, prescribed burning treatments provides a useful and cost effective means of preventing shrub spread in situations where low shrub levels are to be maintained.

Roller chopping may be the preferred technique for significant, rapid shrub reduction and the best method for initial treatment and control of shrubs in areas where they have high density and cover. It also has potential for use in controlling shrub increases in sites near urban developments and recreational areas where smoke management issues may arise during and following burning. Roller chopping and roller chopping/burning treatments resulted in prolonged lowering of shrub height and cover, and, in the case of growing season roller chopping, shrub density. The prolonged effects of these 2 treatments are largely what separated them from the practices of dormant and growing season burning. Similarly, other studies have found roller

chopping to be useful for improving poor condition flatwoods sites with dense shrub cover. In such situations, this treatment has resulted in considerable decreases in shrub abundance and cover; being particularly useful as a form of saw palmetto control (Moore 1974, Tanner et al. 1988). Roller chopping can result in 83% saw palmetto kill and shrub cover reduction of up to 3% on treated compared to control sites up to 2 years post-treatment (Moore 1974). These reductions often persist in the long-term, with 70% kill and crown cover reduced to less than 1-2% up to 5 years after roller chopping (Hilmon 1968, Lewis 1970). This study found no difference in the effects of dormant and growing season roller chopping on shrub height and cover. However, it has been suggested that chopping during periods of high soil moisture, i.e., during the growing season, may result in greater shrub reduction due to the greater penetration of the equipment's blades and deeper severing of the plant's roots that occur at this time of year (Tanner et al. 1988).

Prolonged reductions in a number of understory characteristics, including richness, density, and cover, on dormant season burn sites contradicts the findings of other studies, which suggest increases in hardwood stems following this treatment. These increases are attributed to resprouting of top-killed plants (Langdon 1981, Waldrop et al. 1987). In other studies, while increases in understory density were not observed, dormant season burning had no effect on density or cover of species due to insufficient top-kill (Robbins and Myers 1992). Decreases in understory density, basal area, and cover because of dormant season burning were often only observed on nongrazed sites. One possible explanation is that livestock, following a burn, may focus their attention on consumption of newly resprouting herbaceous species (Hilmon and Hughes

1965) rather than the consumption of less palatable understory browse, which may be consumed when there is reduced herbaceous forage availability (Kalmbacher et al. 1984).

I found understory richness, density, basal area, and cover to be largely unaffected by growing season burning and dormant season roller chopping. It is not possible to report on the effect growing season roller chopping and roller chop/burning had on understory characteristics as both control and treatment sites were largely free of understory trees. Understory stem density has been found to be lower on growing season burn compared to dormant season burn sites due to greater root kill from hotter summer fires (Grelan 1975, Langdon 1981, Waldrop et al. 1987, Fitzgerald 1990, Robbins and Myers 1992). In other situations, understory density has been found to be similar on both dormant and growing season burn sites (Lotti 1956, Robbins and Myers 1992).

Lower overstory canopy cover on dormant season burn than control sites suggests there may have been damage to tree crowns. Maintenance of the pine overstory is a typical goal of pine flatwoods management. Crown scorch, which can cause damage to developing buds and threaten growth, is often of particular concern following growing season burning (Robbins and Myers 1992). In the dormant season, buds are not present in the canopy and the chances of damage are lower (Robins and Myers 1992). Throughout the 2 years of the study, there were no changes in overstory pine density or basal area on dormant or growing season burn sites, suggesting scorching had no significant effect on overstory density or basal area.

Neither dormant nor growing season roller chopping affected overstory characteristics. There are concerns that roller chopping may cause damage to pine roots, leading to reduced growth or mortality (Willcox personal observation). However, no evidence of pine damage following roller chopping or roller chop/burning was observed during this study.

Management Implications

Dormant and growing season burning would be best utilized to suppress understory and shrub growth in pine flatwoods habitats where cover and height is not excessive and shrub species are not dominant. Where shrub levels are high, the single application of either of these treatments are unlikely to cause prolonged reductions in shrub density, height, or cover. In situations where shrub proliferation has not occurred, a single dormant or growing season burn will likely prove a cost-effective means of maintaining desirable understory, shrub, and herbaceous conditions, although care should be taken to protect the overstory canopy. Alternative management practices such as roller chopping may be needed in combination with, or instead of, dormant and growing season burning if there is a need to reduce excessive understory and shrub density and cover, and maintain or restore herbaceous plants following fire exclusion. As decreases in herbaceous characteristics were often associated with grazing on dormant burn sites, initial deferment from grazing following the use of dormant season or growing season burning, as for all practices discussed, is likely to be necessary to permit re-establishment and growth of forbs and graminoids. Consideration should be given to the use of growing over dormant season burning as a means of reducing shrub cover for up to 1 year post-burn. Although during this study, growing season burning resulted in decreases in herbaceous cover and height, it may in the longer-term

increase flowering and seeding of important flatwoods plants, maintaining and restoring areas previously dominated by shrubs.

In situations where shrubs have become a nuisance, a rapid reduction in their height and cover in the longer term (≥ 2 years) may be best achieved through the application of dormant or growing season roller chopping or roller chop/burning. If reductions in shrub density are required, growing season roller chopping may be the only suitable treatment.

Table 2-1. Effects of treatment on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat Characteristics by Treatment ^a	Treatment ($\bar{x} \pm SE$)		<i>P</i>
	Control	Treated	
Dormant season burn			
Visual obstruction (% from 10 m distance)	40.9 ± 2.3	28 ± 1.7	≤0.001
Overstory canopy cover (%)	11.9 ± 1.7	9.1 ± 1.3	0.070
Variance of litter cover (%)	0.6 ± 0.2	1.1 ± 0.2	0.040
Growing season burn			
Mean forb cover (%)	1.8 ± 0.2	1.2 ± 0.1	0.028
Variance of forb height (cm)	3.5.3 ± 44.9	129.5 ± 29.5	0.024
Mean graminoid richness (no. of spp.)	3.3 ± 0.2	3.0 ± 0.2	0.016
Visual obstruction (% from 10 m distance)	37.5 ± 2.6	26.8 ± 1.9	0.015
Mean litter cover (%)	5.4 ± 0.1	3.0 ± 0.3	≤0.001
Mean litter depth (cm)	10.3 ± 1.0	2.5 ± 0.7	≤0.001
Variance of litter depth (cm)	38.9 ± 7.7	10.0 ± 7.2	≤0.001
Mean soil pH	5.7 ± 0.1	5.9 ± 0.1	0.036
Dormant season roller chop			
Shrub height (cm)	101.8 ± 3.0	77.3 ± 2.8	≤0.001
Mean soil moisture (%)	8.0 ± 2.5	13.42 ± 2.8	0.017
Variance of soil moisture (%)	21.4 ± 6.0	82.9 ± 20.4	0.008
Growing season roller chop			
Shrub richness (no. of spp.)	5.3 ± 0.3	4.3 ± 0.4	0.033
Mean litter depth (cm)	10.1 ± 0.9	6.0 ± 0.8	0.055
Variance of litter depth (cm)	26.7 ± 5.0	32.0 ± 23.2	≤0.001
Roller chop/burn			
Shrub cover (%)	64.4 ± 4.4	30.2 ± 3.5	≤0.001
Visual obstruction (% from 10 m distance)	37.3 ± 3.0	18.8 ± 1.8	≤0.001
Understory richness (no. of spp.)	0.2 ± 0.1	0.0 ± 0.0	0.024
Understory density (no./ha)	153.8 ± 64.5	0.0 ± 0.0	0.025
Variance of litter depth (cm)	29.6 ± 8.3	10.7 ± 5.0	≤0.001

^a Only habitat characteristics significantly affected by treatment presented ($P \leq 0.1$).

Table 2-2. Effects of treatment * time interactions on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat Characteristics by Treatment ^a	Time ^b	Treatment ($\bar{x} \pm SE$) ^{c,d}		P
		Control	Treated	
Dormant season burn				
Shrub cover (%)	1	64.2 ± 4.6 _{Aa}	39.1 ± 4.6 _{Aa}	0.085
	2	67.5 ± 3.4 _{Aa}	67.9 ± 3.2 _{Ab}	
Variance of forb height (cm)	1	317.1 ± 47.7 _{Aa}	176.2 ± 42.5 _{Ba}	0.053
	2	273.4 ± 45.6 _{Ab}	253.0 ± 48.0 _{Aa}	
Mean graminoid cover (%)	1	2.5 ± 0.2 _{Aa}	2.0 ± 0.1 _{Aa}	0.035
	2	2.2 ± 0.1 _{Aa}	2.4 ± 0.1 _{Aa}	
Variance of graminoid cover (%)	1	0.5 ± 0.1 _{Aa}	0.3 ± 0.1 _{Ba}	0.006
	2	0.5 ± 0.2 _{Aa}	0.4 ± 0.1 _{Aa}	
Mean graminoid height (cm)	1	59.1 ± 4.2 _{Aa}	39.5 ± 3.4 _{Ba}	≤0.001
	2	56.8 ± 3.4 _{Aa}	56.5 ± 2.6 _{Aa}	
Variance of graminoid height (cm)	1	390.3 ± 77.3 _{Aa}	134.4 ± 27.7 _{Ba}	0.026
	2	338.1 ± 68.0 _{Aa}	222.9 ± 53.3 _{Aa}	
Visual obstruction (% from 5 m distance)	1	25.1 ± 2.9 _{Aa}	15.2 ± 3.2 _{Ba}	0.024
	2	28.2 ± 2.2 _{Aa}	26.3 ± 1.8 _{Ab}	
Understory richness (no. of spp.)	1	0.4 ± 0.2 _{Aa}	0.2 ± 0.1 _{Aa}	0.027
	2	0.6 ± 0.2 _{Ab}	0.2 ± 0.1 _{Aa}	
Mean litter cover (%)	1	5.5 ± 0.1 _{Aa}	3.4 ± 0.3 _{Ba}	0.016
	2	5.3 ± 0.2 _{Aa}	4.5 ± 0.2 _{Ba}	
Mean litter depth (cm)	1	8.2 ± 1.1 _{Aa}	4.8 ± 2.9 _{Ba}	0.005
	2	9.4 ± 1.1 _{Aa}	6.3 ± 0.7 _{Aa}	
Variance of litter depth (cm)	1	27.6 ± 11.8 _{Aa}	4.2 ± 2.2 _{Aa}	0.035
	2	24.7 ± 7.0 _{Aa}	38.3 ± 23.5 _{Ab}	
Growing season burn				
Shrub height (cm)	1	135.9 ± 20.8 _{Aa}	51.2 ± 5.7 _{Ba}	0.021
	2	133.3 ± 7.6 _{Aa}	92.2 ± 3.4 _{Ab}	
Shrub cover (%)	1	67.3 ± 6.6 _{Aa}	16.2 ± 5.9 _{Aa}	0.019
	2	73.8 ± 4.7 _{Aa}	76.6 ± 4.3 _{Ab}	
Variance of forb cover (%)	1	0.5 ± 0.1 _{Aa}	0.2 ± 0.1 _{Aa}	0.031
	2	6.8 ± 6.7 _{Aa}	0.3 ± 0.0 _{Aa}	
Mean graminoid cover (%)	1	2.7 ± 0.2 _{Aa}	1.6 ± 0.2 _{Ba}	0.035
	2	2.2 ± 0.1 _{Aa}	2.7 ± 0.4 _{Aa}	
Mean graminoid height (cm)	1	73.1 ± 4.2 _{Aa}	31.6 ± 5.4 _{Ba}	0.012
	2	56.3 ± 3.2 _{Aa}	52.6 ± 3.1 _{Ab}	
Visual obstruction (% from 5 m distance)	1	26.1 ± 3.8 _{Aa}	8.6 ± 2.0 _{Ba}	0.021
	2	28.2 ± 2.7 _{Aa}	21.5 ± 1.6 _{Ab}	
Variance of soil moisture (%)	1	29.9 ± 11.4 _{Aa}	36.6 ± 13.1 _{Aa}	0.070
	2	56.3 ± 17.8 _{Aa}	22.2 ± 8.2 _{Aa}	
Dormant season roller chop				
Shrub density (no./m ²)	1	3.7 ± 0.3 _{Aa}	3.2 ± 0.7 _{Aa}	0.074
	2	3.3 ± 0.3 _{Aa}	3.5 ± 0.5 _{Aa}	
Shrub cover (%)	1	58.3 ± 5.7 _{Aa}	19.4 ± 4.1 _{Ba}	0.072
	2	57.6 ± 3.1 _{Aa}	35.5 ± 4.0 _{Ba}	

Table 2-2. Continued

Habitat Characteristics by Treatment ^a	Time ^b	Treatment ($\bar{x} \pm SE$) ^{c,d}		P
		Control	Treated	
Mean forb richness (no. of spp.)	1	4.5 ± 0.3 _{Aa}	3.9 ± 0.3 _{Aa}	0.067
	2	4.4 ± 0.3 _{Aa}	5.0 ± 0.3 _{Aa}	
Mean forb cover (%) ^a	1	1.6 ± 0.1 _{Aa}	1.5 ± 0.1 _{Aa}	0.018
	2	1.6 ± 0.1 _{Aa}	1.9 ± 0.1 _{Aa}	
Mean forb height (cm)	1	29.3 ± 1.5 _{Aa}	24.8 ± 2.2 _{Aa}	0.002
	2	30.3 ± 1.5 _{Aa}	38.8 ± 2.6 _{Aa}	
Mean graminoid cover (%)	1	2.8 ± 0.2 _{Aa}	2.0 ± 0.1 _{Aa}	0.064
	2	2.5 ± 0.1 _{Aa}	2.3 ± 0.1 _{Aa}	
Visual obstruction (% from 5 m distance)	1	23.2 ± 2.1 _{Aa}	6.7 ± 1.2 _{Ba}	≤0.001
	2	20.0 ± 2.0 _{Aa}	14.6 ± 1.6 _{Ab}	
Visual obstruction (% from 10 m distance)	1	31.6 ± 2.0 _{Aa}	11.6 ± 1.3 _{Ba}	0.005
	2	30.5 ± 1.9 _{Aa}	20.5 ± 1.5 _{Ab}	
Mean litter depth (cm)	1	8.7 ± 1.0 _{Aa}	6.2 ± 0.6 _{Ba}	0.045
	2	7.7 ± 0.6 _{Aa}	3.7 ± 0.4 _{Ba}	
Variance of litter depth (cm)	1	35.5 ± 10.3 _{Aa}	11.8 ± 3.5 _{Ba}	0.064
	2	30.1 ± 5.2 _{Aa}	6.3 ± 1.5 _{Ba}	
Growing season roller chop				
Mean forb height (cm)	1	33.5 ± 2.4 _{Aa}	17.6 ± 3.7 _{Ba}	0.004
	2	29.8 ± 1.9 _{Aa}	30.7 ± 2.2 _{Ab}	
Mean graminoid richness (no. of spp.)	1	4.3 ± 0.4 _{Aa}	3.3 ± 0.4 _{Ba}	0.028
	2	4.0 ± 0.2 _{Aa}	4.6 ± 0.3 _{Ab}	
Mean graminoid cover (%)	1	2.6 ± 0.2 _{Aa}	1.9 ± 0.2 _{Ba}	0.099
	2	3.1 ± 0.7 _{Aa}	2.3 ± 0.1 _{Aa}	
Mean graminoid height (cm)	1	57.5 ± 2.4 _{Aa}	36.6 ± 3.6 _{Ba}	0.005
	2	58.4 ± 2.3 _{Aa}	49.6 ± 2.5 _{Aa}	
Visual obstruction (% from 5 m distance)	1	24.4 ± 4.7 _{Aa}	4.6 ± 1.0 _{Ba}	0.020
	2	17.8 ± 1.2 _{Aa}	7.8 ± 1.0 _{Ba}	
Visual obstruction (% from 10 m distance)	1	29.3 ± 4.1 _{Aa}	6.9 ± 1.5 _{Ba}	0.009
	2	23.0 ± 1.4 _{Aa}	10.8 ± 1.2 _{Aa}	
Variance of soil pH	1	0.1 ± 0.0 _{Aa}	0.2 ± 0.0 _{Aa}	0.034
	2	0.3 ± 0.1 _{Aa}	0.1 ± 0.0 _{Bb}	
Mean soil density (g/cm ³)	1	1.2 ± 0.0 _{Aa}	1.2 ± 0.0 _{Aa}	0.087
	2	1.2 ± 0.0 _{Aa}	1.2 ± 0.0 _{Aa}	
Roller chop/burn				
Shrub richness (no. of spp.)	1	5.1 ± 0.3 _{Aa}	4.8 ± 0.5 _{Aa}	0.084
	2	4.8 ± 0.2 _{Aa}	6.0 ± 0.4 _{Aa}	
Mean forb height (cm)	1	35.2 ± 2.6 _{Aa}	28.1 ± 4.4 _{Aa}	0.048
	2	37.6 ± 1.9 _{Aa}	38.7 ± 2.8 _{Ab}	
Variance of forb height (cm)	1	302.0 ± 74.2 _{Aa}	101.5 ± 26.2 _{Ba}	0.091
	2	233.4 ± 37.6 _{Aa}	286.5 ± 63.0 _{Aa}	
Mean graminoid richness (no. of spp.)	1	3.5 ± 0.3 _{Aa}	3.4 ± 0.4 _{Aa}	0.084
	2	3.2 ± 0.2 _{Aa}	4.0 ± 0.4 _{Aa}	
Mean graminoid cover (%)	1	2.4 ± 0.2 _{Aa}	1.7 ± 0.2 _{Ba}	0.089
	2	2.4 ± 0.2 _{Aa}	2.0 ± 0.1 _{Aa}	

Table 2-2. Continued

Habitat Characteristics by Treatment ^a	Time ^b	Treatment ($\bar{x} \pm SE$) ^{c,d}		<i>P</i>
		Control	Treated	
Mean graminoid height (cm)	1	61.4 ± 4.0 _{Aa}	37.1 ± 4.5 _{Ba}	0.055
	2	62.1 ± 4.4 _{Aa}	43.9 ± 3.8 _{Ab}	
Visual obstruction (% from 5 m distance)	1	28.5 ± 3.4 _{Aa}	8.2 ± 2.1 _{Ba}	0.085
	2	26.8 ± 2.7 _{Aa}	17.5 ± 2.2 _{Aa}	
Mean litter cover (%)	1	5.5 ± 0.1 _{Aa}	2.6 ± 0.3 _{Ba}	0.029
	2	5.0 ± 0.2 _{Aa}	3.5 ± 0.3 _{Aa}	
Variance of litter cover (%)	1	0.6 ± 0.4 _{Aa}	1.5 ± 0.4 _{Ba}	0.033
	2	1.2 ± 0.3 _{Aa}	1.4 ± 0.3 _{Aa}	
Mean litter depth (cm)	1	10.4 ± 1.2 _{Aa}	4.4 ± 2.8 _{Ba}	0.016
	2	8.8 ± 0.9 _{Aa}	3.9 ± 0.7 _{Ba}	

^a Only habitat characteristics significantly affected by a treatment * time interaction presented ($P \leq 0.1$).

^b Time since treatment application (years).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

^d Means in a column followed by the same lowercase letter not significantly different ($P > 0.1$).

Table 2-3. Effects of treatment * grazing interactions on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat characteristics by treatment ^a	Grazing	Treatment ($\bar{x} \pm SE$) ^b		P
		Control	Treated	
Dormant season burn				
Shrub cover (%)	Nongrazed	88.5 ± 7.7 _A	54.9 ± 7.5 _A	0.030
	Grazed	61.2 ± 2.6 _A	53.5 ± 3.8 _A	
Mean forb richness (no. of spp.)	Nongrazed	3.2 ± 0.2 _A	2.3 ± 0.4 _A	0.028
	Grazed	3.6 ± 0.3 _A	4.4 ± 0.3 _A	
Mean forb cover (%)	Nongrazed	1.9 ± 0.2 _A	1.1 ± 0.1 _B	≤0.001
	Grazed	1.4 ± 0.1 _A	1.7 ± 0.1 _B	
Mean forb height (cm)	Nongrazed	36.3 ± 2.9 _A	27.0 ± 3.9 _A	0.088
	Grazed	34.7 ± 2.2 _A	34.7 ± 2.7 _A	
Variance of forb height (cm)	Nongrazed	163.5 ± 51.9 _A	375.9 ± 91.3 _A	0.003
	Grazed	322.6 ± 37.0 _A	176.8 ± 31.5 _A	
Mean graminoid cover (%)	Nongrazed	1.8 ± 0.2 _A	2.2 ± 0.2 _B	0.061
	Grazed	2.5 ± 0.1 _A	2.2 ± 0.1 _A	
Understory richness (no. of spp.)	Nongrazed	1.1 ± 0.3 _A	0.1 ± 0.1 _B	0.009
	Grazed	0.4 ± 1.2 _A	0.2 ± 0.1 _A	
Understory density (no./ha)	Nongrazed	225.9 ± 74.5 _A	6.0 ± 6.0 _B	0.023
	Grazed	51.1 ± 17.5 _A	33.3 ± 11.4 _A	
Growing season burn				
Shrub richness (no. of spp.)	Nongrazed	7.1 ± 0.3 _A	5.0 ± 0.6 _A	0.069
	Grazed	5.3 ± 0.5 _A	4.8 ± 0.6 _A	
Mean forb height (cm)	Nongrazed	38.9 ± 2.8 _A	19.6 ± 2.6 _A	0.017
	Grazed	25.8 ± 5.6 _A	26.2 ± 5.3 _A	
Variance of graminoid height (cm)	Nongrazed	489.5 ± 121.6 _A	121.6 ± 27.2 _A	0.006
	Grazed	218.1 ± 77.4 _A	299.9 ± 64.6 _A	
Variance of soil pH	Nongrazed	0.1 ± 0.0 _A	0.1 ± 0.0 _B	0.047
	Grazed	0.1 ± 0.0 _A	0.0 ± 0.0 _A	
Mean soil density (g/cm ³)	Nongrazed	1.2 ± 0.0 _A	1.2 ± 0.0 _A	0.064
	Grazed	1.3 ± 0.0 _A	0.3 ± 0.0 _A	
Dormant season roller chop				
Shrub richness (no. of spp.)	Nongrazed	5.9 ± 0.4 _A	4.2 ± 0.3 _A	0.067
	Grazed	4.2 ± 0.2 _A	3.7 ± 0.3 _A	
Variance of graminoid cover (%)	Nongrazed	0.5 ± 0.1 _A	1.5 ± 1.0 _A	0.064
	Grazed	0.4 ± 0.1 _A	0.2 ± 0.0 _A	
Visual obstruction (% from 5 m distance)	Nongrazed	19.2 ± 1.7 _A	12.3 ± 2.0 _A	0.082
	Grazed	23.4 ± 2.2 _A	9.4 ± 1.2 _A	
Variance of litter depth (cm)	Nongrazed	26.4 ± 6.6 _A	10.4 ± 3.3 _B	0.046
	Grazed	38.1 ± 8.7 _A	7.6 ± 2.0 _B	
Mean soil pH	Nongrazed	6.0 ± 0.1 _A	5.9 ± 0.1 _A	0.052
	Grazed	5.8 ± 0.1 _A	6.0 ± 0.1 _A	
Growing season roller chop				
Shrub density (no./m ²)	Nongrazed	3.4 ± 0.4 _A	3.0 ± 0.6 _A	0.080
	Grazed	3.7 ± 0.8 _A	1.4 ± 0.3 _B	

Table 2-3. Continued

Habitat characteristics by treatment ^a	Grazing	Treatment ($\bar{x} \pm SE$) ^b		<i>P</i>
		Control	Treated	
Shrub height (cm)	Nongrazed	106.6 ± 5.3 _A	57.5 ± 3.4 _A	0.014
	Grazed	89.1 ± 5.1 _A	65.9 ± 3.4 _A	
Shrub cover (%)	Nongrazed	43.8 ± 4.7 _A	17.4 ± 4.2 _A	0.031
	Grazed	45.5 ± 3.3 _B	8.1 ± 2.0 _B	
Variance of graminoid height (cm)	Nongrazed	192.4 ± 47.8 _A	236.1 ± 60.5 _A	0.097
	Grazed	220.3 ± 91.0 _A	95.6 ± 18.0 _A	
Mean litter cover (%)	Nongrazed	5.4 ± 0.2 _A	4.6 ± 0.2 _B	0.098
	Grazed	4.8 ± 0.2 _A	4.4 ± 0.4 _B	
Roller chop/burn				
Shrub height (cm)	Nongrazed	128.4 ± 7.8 _A	73.0 ± 4.6 _B	0.054
	Grazed	111.4 ± 6.7 _A	82.1 ± 4.6 _A	
Mean forb richness (no. of spp.)	Nongrazed	5.1 ± 0.6 _A	4.4 ± 0.5 _A	0.068
	Grazed	4.5 ± 0.3 _A	5.6 ± 0.5 _A	
Mean forb cover (%)	Nongrazed	2.5 ± 0.7 _A	1.5 ± 0.1 _A	0.006
	Grazed	1.5 ± 0.1 _A	2.0 ± 0.1 _A	
Mean forb height (cm)	Nongrazed	36.8 ± 2.3 _A	28.9 ± 2.6 _A	0.081
	Grazed	36.5 ± 2.0 _A	42.0 ± 4.2 _A	
Mean graminoid cover (%)	Nongrazed	2.0 ± 0.2 _A	1.9 ± 0.2 _A	0.014
	Grazed	2.9 ± 0.2 _A	2.0 ± 0.2 _A	
Mean graminoid height (cm)	Nongrazed	54.0 ± 3.3 _A	40.0 ± 3.5 _A	0.090
	Grazed	72.1 ± 4.8 _A	43.2 ± 5.0 _A	
Variance of litter cover (%)	Nongrazed	1.3 ± 0.3 _A	1.3 ± 0.3 _A	0.011
	Grazed	0.6 ± 0.2 _A	1.6 ± 0.4 _A	
Mean litter depth (cm)	Nongrazed	8.1 ± 0.8 _A	2.9 ± 0.7 _A	0.087
	Grazed	11.2 ± 1.1 _A	3.2 ± 0.8 _A	
Mean soil density (g/cm ³)	Nongrazed	1.3 ± 0.0 _A	1.3 ± 0.0 _A	0.042
	Grazed	1.2 ± 0.0 _A	1.4 ± 0.2 _A	

^a Only habitat characteristics significantly affected by a treatment * grazing interaction presented ($P \leq 0.1$).

^b Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 2-4. Effects of treatment * season interactions on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat characteristics by treatment ^a	Season	Treatment ($\bar{x} \pm SE$) ^b		P
		Control	Treated	
Dormant-season burn				
Mean graminoid height (cm)	Winter	65.3 ± 5.2 _A	41.6 ± 4.9 _B	≤0.001
	Spring	51.8 ± 4.2 _A	44.9 ± 3.9 _A	
	Summer	58.0 ± 4.5 _A	57.0 ± 3.2 _A	
Growing season burn				
Variance of soil pH	Winter	0.1 ± 0.0 _A	0.1 ± 0.0 _A	0.042
	Spring	0.1 ± 0.0 _A	0.1 ± 0.0 _A	
	Summer	0.1 ± 0.0 _A	0.1 ± 0.0 _A	
Dormant season roller chopping				
Mean graminoid height (cm)	Winter	57.6 ± 5.2 _A	40.2 ± 3.1 _B	0.051
	Spring	44.2 ± 2.2 _A	39.7 ± 1.8 _A	
	Summer	49.7 ± 2.1 _A	49.6 ± 2.5 _A	
Growing season roller chop				
Mean graminoid height (cm)	Winter	56.1 ± 5.8 _A	43.4 ± 3.9 _B	0.014
	Spring	58.8 ± 2.5 _A	44.1 ± 2.3 _B	
	Summer	58.9 ± 1.8 _A	49.0 ± 4.0 _A	
Mean litter cover (%)	Winter	4.9 ± 0.3 _A	5.0 ± 0.3 _A	0.049
	Spring	4.9 ± 0.4 _A	4.4 ± 0.4 _A	
	Summer	5.4 ± 0.2 _A	4.4 ± 0.3 _A	
Roller chop/burn				
Mean forb height (cm)	Winter	35.9 ± 2.1 _A	30.6 ± 3.9 _A	0.061
	Spring	37.7 ± 2.8 _A	29.1 ± 4.3 _A	
	Summer	36.2 ± 0.7 _A	41.7 ± 3.9 _A	
Mean forb height (cm)	Winter	35.9 ± 2.1 _A	30.6 ± 3.9 _A	0.061
	Spring	37.7 ± 2.8 _A	29.1 ± 4.3 _A	
	Summer	36.2 ± 0.7 _A	41.7 ± 3.9 _A	
Mean forb height (cm)	Winter	35.9 ± 2.1 _A	30.6 ± 3.9 _A	0.061
	Spring	37.7 ± 2.8 _A	29.1 ± 4.3 _A	
	Summer	36.2 ± 0.7 _A	41.7 ± 3.9 _A	
Mean graminoid cover (%)	Winter	2.9 ± 0.2 _A	1.8 ± 0.2 _B	0.044
	Spring	2.4 ± 0.2 _A	35.0 ± 4.1 _A	
	Summer	2.2 ± 0.2 _A	2.1 ± 0.2 _A	
Mean graminoid height (cm)	Winter	71.9 ± 8.0 _A	30.9 ± 7.1 _B	0.020
	Spring	56.3 ± 5.4 _A	35.0 ± 4.1 _B	
	Summer	61.1 ± 3.4 _A	52.5 ± 3.5 _A	

^a Only habitat characteristics significantly affected by a treatment * season interaction presented ($P \leq 0.1$).

^b Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 2-5. Comparison of the effects of treatment type on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat characteristics ^a	Treatment type ($\bar{x} \pm SE$) ^b				P
	Control	Dormant burn	Growing burn	Roller chop/burn	
Understory richness (no. of spp.)	0.3 ± 0.0 _A	0.2 ± 0.1 _B	0.4 ± 0.1 _A	0.1 ± 0.0 _{AB}	0.019
Understory density (no./ha)	57.7 ± 14.5 _A	28.1 ± 9.3 _B	20.4 ± 6.5 _{AB}	3.8 ± 1.5 _B	0.010
Understory cover (%)	0.2 ± 0.1 _A	0.0 ± 0.0 _B	0.3 ± 0.2 _{AB}	0.0 ± 0.0 _A	0.020
Variance of litter depth (cm)	30.4 ± 3.0 _A	21.6 ± 12.1 _{AB}	10.1 ± 7.2 _{AB}	9.0 ± 1.9 _B	0.097

^a Only habitat characteristics significantly affected by treatment type presented ($P \leq 0.1$).

^b Means in a row followed by the same letter not significantly different ($P > 0.1$).

Table 2-6. Comparison of the effects of treatment type * time interactions on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat characteristics ^a	Time ^b	Treatment type ($\bar{x} \pm SE$) ^c						<i>P</i>
		Control	Dormant burn	Growing burn	Dormant roller chop	Growing roller chop	Roller chop/burn	
Visual obstruction (% from 5 m distance)	1	25.5 ± 1.4 _A	15.2 ± 3.2 _B	8.6 ± 2.0 _B	6.7 ± 1.2 _B	4.6 ± 1.0 _B	8.2 ± 2.1 _B	≤0.001
	2	24.0 ± 1.0 _A	26.3 ± 1.8 _{AB}	21.5 ± 1.5 _{AC}	14.6 ± 1.6 _{BD}	7.8 ± 1.0 _D	17.5 ± 2.2 _{BCD}	
Visual obstruction (% at 10 m distance)	1	35.6 ± 1.7 _A	21.6 ± 2.4 _B	18.9 ± 4.6 _B	11.6 ± 1.3 _B	6.9 ± 1.5 _B	12.6 ± 2.3 _B	≤0.001
	2	33.9 ± 1.3 _A	34.2 ± 1.8 _{AB}	29.5 ± 1.7 _{AB}	20.5 ± 1.5 _B	10.8 ± 1.2 _B	22.7 ± 2.2 _{AB}	
Shrub height (cm)	1	115.2 ± 3.9 _A	95.6 ± 7.1 _B	51.2 ± 5.7 _B	69.2 ± 4.1 _B	59.4 ± 6.5 _B	63.8 ± 2.0 _{AB}	0.013
	2	118.7 ± 3.0 _A	125.3 ± 3.7 _A	92.2 ± 3.4 _{AB}	84.8 ± 3.4 _B	60.9 ± 2.7 _B	85.2 ± 3.6 _{AB}	
Shrub cover (%)	1	60.1 ± 2.7 _A	39.1 ± 4.6 _B	16.2 ± 5.9 _B	19.4 ± 4.0 _B	9.4 ± 5.1 _B	20.1 ± 4.7 _B	≤0.001
	2	61.5 ± 2.1 _A	67.9 ± 3.2 _A	76.6 ± 4.3 _A	35.5 ± 3.6 _B	15.7 ± 3.4 _B	36.5 ± 4.6 _A	
Mean forb richness (no. of spp.)	1	3.9 ± 0.2 _A	3.8 ± 0.5 _A	1.2 ± 0.4 _A	3.9 ± 0.3 _A	3.2 ± 0.6 _B	4.0 ± 0.4 _A	0.042
	2	4.0 ± 0.2 _A	4.2 ± 0.4 _B	3.0 ± 0.4 _A	5.0 ± 0.3 _{AB}	4.4 ± 0.3 _{AB}	5.5 ± 0.5 _{AB}	
Mean forb cover (%)	1	1.6 ± 0.1 _{AC}	1.5 ± 0.1 _C	0.8 ± 0.2 _B	1.5 ± 0.1 _A	1.4 ± 0.2 _B	1.6 ± 0.2 _C	0.042
	2	1.8 ± 0.2 _A	1.8 ± 0.1 _B	1.3 ± 0.1 _A	2.0 ± 0.1 _{AB}	1.9 ± 0.2 _{AC}	1.7 ± 0.1 _{BC}	
Variance of forb height (cm)	1	251.4 ± 25.5 _A	176.2 ± 42.5 _B	44.1 ± 21.0 _A	164.2 ± 57.0 _A	121.5 ± 42.7 _{AB}	101.5 ± 26.2 _B	0.024
	2	234.0 ± 19.3 _A	253.0 ± 48.0 _B	157.9 ± 37.3 _{AB}	142.5 ± 21.0 _{AB}	172.6 ± 38.3 _{AB}	286.5 ± 63.0 _{AB}	
Mean graminoid richness (no. of spp.)	1	3.6 ± 0.1 _{AB}	3.1 ± 0.2 _{AB}	2.2 ± 0.4 _A	3.4 ± 0.1 _B	3.3 ± 0.4 _{AB}	37.1 ± 4.5 _{AB}	0.005
	2	3.7 ± 0.1 _A	4.1 ± 0.2 _A	3.2 ± 0.2 _B	4.2 ± 0.3 _A	4.6 ± 0.3 _A	3.9 ± 0.4 _A	
Mean graminoid height (cm)	1	57.3 ± 1.9 _A	39.5 ± 3.4 _B	31.6 ± 5.4 _B	40.4 ± 1.8 _B	36.6 ± 3.6 _B	37.1 ± 4.5 _{AB}	≤0.001
	2	57.4 ± 1.5 _A	56.5 ± 2.6 _B	52.7 ± 3.1 _{BC}	45.9 ± 2.4 _C	49.6 ± 2.5 _{BC}	43.9 ± 3.8 _{ABC}	
Mean litter cover (%)	1	5.6 ± 0.3 _A	3.4 ± 0.3 _B	2.7 ± 0.6 _C	5.3 ± 0.3 _B	4.9 ± 0.3 _{AB}	2.6 ± 0.3 _{BC}	0.052
	2	5.1 ± 0.1 _A	4.5 ± 0.2 _B	3.1 ± 0.3 _C	4.5 ± 0.2 _B	4.4 ± 0.2 _B	3.5 ± 0.3 _B	
Mean litter depth (cm)	1	9.9 ± 0.6 _A	4.8 ± 2.9 _B	1.2 ± 0.4 _B	6.2 ± 0.6 _C	8.5 ± 2.1 _{ABC}	1.7 ± 0.5 _{BC}	0.001
	2	8.8 ± 0.4 _A	6.3 ± 0.7 _B	2.9 ± 1.0 _C	3.7 ± 0.4 _D	5.1 ± 0.8 _{BD}	3.9 ± 0.7 _{BD}	

^a Only habitat characteristics significantly affected by a treatment type * time interaction presented ($P \leq 0.1$).

^b Time since treatment application (years).

^c Means in a row followed by the same letter not significantly different ($P > 0.1$).

Table 2-7. Comparison of the effects of treatment type * grazing interactions on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat characteristics ^a	Grazing	Treatment type ($\bar{x} \pm SE$) ^b						P
		Control	Dormant burn	Growing burn	Dormant roller chop	Growing roller chop	Roller chop/burn	
Visual obstruction (% from 5 m distance)	Grazed	26.0 ± 1.3 _A	20.2 ± 2.3 _A	25.5 ± 2.6 _A	9.4 ± 1.2 _{BC}	5.8 ± 0.9 _C	14.1 ± 2.1 _{AB}	≤0.001
	Nongrazed	23.1 ± 1.1 _A	23.5 ± 3.3 _A	16.2 ± 1.7 _{AB}	12.3 ± 2.0 _{AB}	7.6 ± 1.2 _B	13.8 ± 2.6 _{AB}	
Visual obstruction (% at 10 m distance)	Grazed	35.5 ± 1.5 _A	26.4 ± 1.7 _{AB}	35.0 ± 3.6 _A	14.2 ± 1.6 _B	8.9 ± 1.5 _B	19.4 ± 2.5 _{AB}	0.020
	Nongrazed	33.5 ± 1.5 _A	34.5 ± 4.6 _A	24.5 ± 2.0 _{AB}	18.5 ± 1.8 _{AB}	10.3 ± 1.3 _B	18.3 ± 2.5 _{AB}	
Overstory richness (no. of spp.)	Grazed	0.5 ± 0.0 _{AB}	0.8 ± 0.1 _A	0.5 ± 0.2 _{AB}	0.0 ± 0.0 _B	0.0 ± 0.0 _{AB}	0.8 ± 0.1 _A	≤0.001
	Nongrazed	0.5 ± 0.1 _A	0.5 ± 0.2 _A	0.4 ± 0.1 _A	0.6 ± 0.1 _A	0.6 ± 0.2 _A	0.4 ± 0.1 _A	
Overstory density (No./ha)	Grazed	32.2 ± 4.7 _{AB}	56.5 ± 7.7 _A	33.3 ± 12.6 _{AB}	0.0 ± 0.0 _B	0.0 ± 0.0 _{AB}	26.3 ± 3.2 _A	0.002
	Nongrazed	28.5 ± 4.5 _A	33.3 ± 10.1 _A	22.6 ± 6.2 _A	26.4 ± 6.3 _A	44.0 ± 13.4 _A	12.0 ± 3.3 _A	
Overstory basal area (cm ² /ha)	Grazed	2.6 ± 0.9 _{AB}	3.0 ± 0.3 _A	0.6 ± 0.2 _{AB}	0.0 ± 0.0 _B	0.0 ± 0.0 _{AB}	0.9 ± 0.2 _A	0.005
	Nongrazed	1.0 ± 0.2 _A	2.2 ± 0.7 _A	0.9 ± 0.3 _A	1.4 ± 0.4 _A	0.9 ± 0.3 _A	0.5 ± 0.2 _A	
Overstory cover (%)	Grazed	6.4 ± 0.9 _{AB}	10.3 ± 1.5 _A	3.0 ± 1.4 _{AB}	0.0 ± 0.0 _B	0.0 ± 0.0 _{AB}	7.7 ± 1.3 _A	≤0.001
	Nongrazed	7.6 ± 1.1 _A	3.8 ± 1.6 _A	4.4 ± 1.2 _A	7.5 ± 2.0 _A	6.2 ± 2.2 _A	1.9 ± 1.0 _A	
Shrub richness (no. of species)	Grazed	5.3 ± 0.2 _{ABC}	6.2 ± 0.3 _C	4.8 ± 0.6 _{AB}	3.7 ± 0.3 _B	3.9 ± 0.5 _{AC}	6.1 ± 0.5 _C	0.019
	Nongrazed	5.8 ± 0.2 _A	4.6 ± 0.4 _{AB}	5.0 ± 0.6 _B	4.2 ± 0.3 _{AB}	4.6 ± 0.5 _{AB}	5.1 ± 0.4 _A	
Shrub density (No./m ² .)	Grazed	4.0 ± 0.2 _A	5.3 ± 0.6 _B	3.4 ± 0.5 _{ABC}	4.1 ± 0.8 _{AB}	1.4 ± 0.3 _C	3.8 ± 0.5 _{AB}	0.015
	Nongrazed	3.7 ± 0.2 _A	3.8 ± 0.5 _A	5.6 ± 0.5 _A	2.5 ± 0.3 _A	3.0 ± 0.6 _A	2.9 ± 0.4 _A	
Shrub height (cm)	Grazed	111.8 ± 3.0 _A	104.2 ± 4.6 _A	92.5 ± 8.7 _{AB}	71.9 ± 3.2 _B	65.9 ± 3.4 _B	82.1 ± 4.6 _{AB}	0.010
	Nongrazed	123.0 ± 3.6 _A	117 ± 8.3 _A	79.0 ± 4.6 _B	85.6 ± 4.6 _B	57.5 ± 3.4 _B	73.0 ± 4.6 _B	
Mean forb richness (no. of spp.)	Grazed	3.8 ± 0.2 _A	4.4 ± 0.3 _B	2.7 ± 0.8 _A	4.6 ± 0.4 _{AB}	3.8 ± 0.4 _{AB}	5.6 ± 0.5 _{AB}	≤0.001
	Nongrazed	4.0 ± 0.2 _A	2.3 ± 0.4 _A	2.5 ± 0.3 _A	4.3 ± 0.3 _A	4.3 ± 0.4 _A	4.4 ± 0.5 _A	
Mean forb cover (%)	Grazed	1.5 ± 0.0 _A	1.7 ± 0.1 _B	1.1 ± 0.2 _A	1.7 ± 0.1 _{ABC}	1.4 ± 0.1 _{AC}	1.9 ± 0.1 _{BC}	≤0.001
	Nongrazed	2.0 ± 0.2 _A	1.1 ± 0.1 _A	1.2 ± 0.1 _A	1.7 ± 0.1 _A	2.0 ± 0.3 _A	1.5 ± 0.1 _A	
Variance of forb height (cm)	Grazed	258.3 ± 21.2 _A	176.8 ± 31.5 _B	210.4 ± 68.0 _{AB}	111.8 ± 21.2 _{AB}	111.2 ± 37.4 _{AB}	191.7 ± 65.6 _{AB}	0.017
	Nongrazed	222.8 ± 22.3 _A	376.0 ± 91.3 _A	106.4 ± 32.0 _A	200.9 ± 57.4 _A	186.6 ± 42.2 _A	232.8 ± 55.4 _A	
Variance of graminoid richness (no. of spp.)	Grazed	1.1 ± 0.1 _{AC}	1.4 ± 0.2 _{ABC}	0.7 ± 0.2 _C	1.4 ± 0.4 _{AB}	1.1 ± 0.3 _{ABC}	1.7 ± 0.3 _B	0.048
	Nongrazed	1.4 ± 0.2 _A	0.8 ± 0.3 _A	1.4 ± 0.6 _A	1.9 ± 0.4 _B	1.3 ± 0.2 _A	1.0 ± 0.2 _A	
Variance of graminoid height (cm)	Grazed	301.5 ± 39.5 _A	204.1 ± 36.8 _A	299.9 ± 64.6 _A	177.3 ± 40.4 _A	95.6 ± 18.1 _A	248.8 ± 86.0 _A	0.007
	Nongrazed	347.7 ± 43.8 _A	76.3 ± 23.0 _A	121.6 ± 27.2 _A	331.4 ± 98.0 _A	236.1 ± 60.5 _A	339.7 ± 124.5 _A	

Table 2-7. Continued

Habitat characteristics ^a	Grazing	Treatment type ($\bar{x} \pm SE$) ^b					<i>P</i>
		Control	Dormant burn	Growing burn	Dormant roller chop	Growing roller chop	
Mean litter cover (%)	Grazed	5.3 ± 0.2 _A	4.1 ± 0.2 _B	1.6 ± 0.3 _C	4.7 ± 0.6 _{BC}	4.4 ± 0.4 _{BC}	0.014
	Nongrazed	5.2 ± 0.1 _A	3.2 ± 0.3 _B	3.4 ± 0.3 _B	5.1 ± 0.1 _B	4.6 ± 0.2 _B	

^a Only habitat characteristics significantly affected by a treatment type * grazing interaction presented ($P \leq 0.1$).

^b Means in a row followed by the same letter not significantly different ($P > 0.1$).

Table 2-8. Comparison of the effects of treatment type * season interactions on habitat characteristics of flatwoods in Florida, 2007–2008.

Habitat characteristics ^a	Season	Treatment type ($\bar{x} \pm SE$) ^b					<i>P</i>
		Control	Dormant burn	Growing burn	Dormant roller chop	Growing roller chop	
Mean graminoid height (cm)	Winter	60.4 ± 2.8 _A	41.6 ± 4.8 _B	52.2 ± 7.2 _A	40.2 ± 3.1 _B	43.4 ± 4.0 _B	0.004
	Spring	52.6 ± 2.0 _A	44.9 ± 4.0 _{AB}	45.3 ± 5.0 _{AB}	40.0 ± 2.0 _B	44.1 ± 2.3 _B	
	Summer	59.0 ± 1.5 _A	57.0 ± 3.2 _{AB}	46.0 ± 4.4 _{AB}	65.1 ± 5.0 _B	49.0 ± 4.0 _{AB}	

^a Only habitat characteristics significantly affected by a treatment type * season interaction presented ($P \leq 0.1$).

^b Means in a row followed by the same letter not significantly different ($P > 0.1$).

CHAPTER 3 SEASONAL EFFECTS OF PRESCRIBED BURNING AND ROLLER CHOPPING ON SAW PALMETTO IN FLATWOODS

Introduction

In rangeland habitats across the United States (U.S.), including the southwestern arid and sub-arid grasslands, central and northern tall- and mixed-grass prairies, and southeastern pine savannas, a reduction in disturbance, most often fire, has resulted in shrub encroachment (Collins and Gibson 1990, Van Auken 2000, Lett and Knapp 2003). This encroachment has resulted in the degradation of many of these disturbance maintained rangelands systems and reduced their value for native wildlife species, particularly birds associated with more open herbaceous dominated habitats (Lloyd et al. 1998, Madden et al. 1999, Hunter et al. 2001).

Saw palmetto (*Serenoa repens* [Bartr.] Small), a low growing, branched, fan palm, has become a dominant shrub species in many pine savanna habitats of the southeastern Coastal Plain. This includes the pine flatwoods, a rangeland community with an open pine overstory and an often rich herbaceous layer (Hilmon 1968). Pine flatwoods cover approximately 50% of the land area of Florida (Abrahamson and Hartnett 1990). Unfortunately, due to shrub invasions, particularly of saw palmetto, large areas of this pine savanna habitat are in poor condition and currently exist in a highly degraded state (Means 1996, Florida Fish and Wildlife Conservation Commission 2005). Historically, flatwoods habitats were maintained by frequent, low intensity, lightning-ignited fires during the May–July thunderstorm season. These fires prevented encroachment by saw palmetto, a species that is able to spread prolifically in the absence of this disturbance (Hilmon 1968, Komarek 1968, Abrahamson and Hartnett 1990, Pyne et al. 1996). During the past 50 years, fire suppression, reductions in fire

frequency, or a shift in fire season, commonly a result of human intervention, have resulted in excessive saw palmetto growth on many pine flatwoods sites across Florida. On Florida's pine flatwoods, increases in saw palmetto height, cover, and density have become a concern, potentially resulting in the loss of many grass and forb species, and declines in the species rich herbaceous ground layer (Wade et al. 1980, Huffman and Blanchard 1991, Robbins and Myers 1992, Olson and Platt 1995). Such changes threaten the integrity of pine flatwoods and their suitability for many wildlife species of conservation concern. These include a variety of mammals, birds, amphibians, and reptiles such as Sherman's fox squirrel (*Sciurus niger shermanii* L.), Florida black bear (*Ursus americanus floridanus* Merriam), red-cockaded woodpecker (*Picoides borealis* Vieillot), Bachman's sparrow (*Aimophila aestivalis* Lichtenstein), flatwoods salamander (*Ambystoma cingulatum* Cope), gopher frog (*Rana capito* LeConte), and gopher tortoise (*Gopherus polyphemus* Daudin; FWC 2005). They have also resulted in declines in forage quantity and quality, potentially reducing the value of these areas for livestock production (Hilmon 1968, Moore 1974, Tanner et al. 1988).

In recent years, a common goal among managers of pine flatwoods has been to reduce the proliferation of shrubs, particularly saw palmetto. For most Florida landowners, many of whom are cattle ranchers, the objective of shrub reduction is to increase the growth and production of more palatable grasses and forbs as food for livestock (Yarlett 1965, Moore 1974, Kalmbacher and Martin 1984, Tanner et al. 1988). Concomitantly, most wildlife species that occupy pine flatwoods habitats benefit from increases in groundcover of grasses and forbs, as they provide diverse food and cover resources (Huber and Steuter 1984, Madden et al. 1999).

The United States Department of Agriculture's Environmental Quality Incentives Program and Wildlife Habitat Incentives Program are currently promoting management activities thought to reduce saw palmetto and other shrub species, and maintain areas of pine flatwoods in Florida. These programs provide landowners financial and technical assistance to implement management activities, including prescribed burning and roller chopping during dormant (November–March) and growing (April–September) seasons.

Prescribed burning and roller chopping can reduce shrubby vegetation in southeastern rangeland habitats (Wade et al. 1980, Kalmbacher and Martin 1984, Tanner et al. 1988, Glitzenstein et al. 1995, Watts and Tanner 2003). A number of studies have compared the effects of season of burning on shrub regeneration and growth, suggesting growing season burning results in greater reductions than dormant season burning (Robbins and Myers 1992). In addition, roller chopping has been shown to cause considerable declines in shrub cover. However, the results of some of these studies are contradictory (Lewis 1970, Moore 1974, Tanner et al. 1988, Watts and Tanner 2003, Watts et al. 2006). Studies that compare shrub responses to prescribed burning and roller chopping are few, as are those that specifically examine effects on saw palmetto, a problem species in many habitats (Watts and Tanner 2003, Watts et al. 2006). In addition, studies that have been conducted are extremely localized, typically being confined to a single study area. If we are to make general recommendations on the use of these practices to individuals managing pine flatwoods across the state, we need detailed research that compares saw palmetto response to prescribed burning and roller chopping practices over a larger area. Therefore, the objective of my study was to

fill recognized gaps in our understanding of how prescribed burning, roller chopping, and combinations of the two, applied during different seasons, affect the height, cover, and density of saw palmetto over a broad range of pine flatwoods habitats. The study was unique in that it examined saw palmetto response to these practices on pine flatwoods sites over a broad geographic area. The intention was to obtain information representative of the “real world”, where landowners use a range of burning and roller chopping techniques to control different levels of saw palmetto under a variety of conditions.

Methods

Study Sites

I conducted research on 50 privately- and publicly-owned, paired treatment and control sites across 6 counties (Desoto, Highlands, Lee, Manatee, Osceola, and Sarasota) in central and south Florida. Study sites consisted of pine flatwoods habitats with varying management histories and grazing regimes that were being prescribed burned and roller chopped by local landowners and land managers using varying, individual protocols. Florida’s pine flatwoods are characterized as having an overstory of scattered slash (*Pinus elliotti* Engelm.) and longleaf (*P. palustris* Mill.) pine, either in pure stands or in combination. The understory and shrub layer includes saw palmetto, wax myrtle (*Morella cerifera* [L.] Small), gallberry (*Ilex glabra* [Pursh] Chapm.), fetterbush (*Lyonia lucida* [Lam.] K. Koch), staggerbush (*Lyonia fruticosa* [Michx.] G. S. Torr), dwarf huckleberry (*Gaylussacia dumosa* [Andrews] Torr. & A. Gray), dwarf live oak (*Quercus mimima* [Sarg.] Small), and tarflower (*Bejaria racemosa* Vent.). An appreciable herbaceous layer exists when the shrub layer is relatively open. This layer contains a wide variety of grasses (e.g., *Agrostis*, *Andropogon*, *Aristida*, *Eragrostis*,

Panicum, and *Paspalum* spp.). Common forbs include legumes (e.g., *Cassia*, *Crotalaria*, *Galactia*, *Tephrosia* spp.), milkweeds (*Asclepias* spp.), milkworts (*Polygala* spp.), and a wide variety of composites (e.g., *Aster*, *Chrysopsis*, *Eupatorium*, *Liatris*, and *Solidago* spp.; Abrahamson and Hartnett 1990, U.S. Fish and Wildlife Service 1999).

Treatment Types

Treatment types included dormant season (November–March) burn, growing season (April–October) burn, dormant season roller chop, growing season roller chop, and a roller chop/burn combination treatment. The roller chop/burn combination treatment (hereafter referred to as roller chop/burn) involved roller chopping in the dormant season followed by burning within 6 months. A total of 11 dormant season burn, 9 growing season burn, 9 dormant season roller chop, 12 growing season roller chop, and 9 roller chop/burn sites were established, each paired with an adjacent untreated control.

Saw Palmetto Sampling

I used a paired-sample approach to assess the effects of management treatments (i.e., prescribed burning, roller chopping, and combinations of the two) on saw palmetto height, cover, and density. These 3 saw palmetto variables were compared between sampling points randomly located in paired treated (e.g., dormant season burned) and untreated (control) flatwoods sites. I randomly located control sampling points in untreated sites adjacent to those in treated sites. These control sites were of similar current and past management (e.g., grazing intensity), surrounding land-use, plant community (e.g., overstory cover), and soil conditions, to treated sites and were located in the same pasture or management unit. Within each site, I established 1 randomly

selected treatment or control sampling point. I rejected and randomly relocated sampling points that occurred within 50 m of the edge of a site to minimize edge effects. Sites, within which treatment and control sampling points were located, ranged from 2–20 ha in size.

Saw palmetto height, cover, and density at each sampling point were assessed following treatment, once in winter (February–March), spring (April–May), and summer (July–August), during each of 2 years (2007–2008). I counted all saw palmetto stems within 2 perpendicular 20-m² quadrats centered on the sampling point to determine density (No./m²; Hays et al. 1981, Bullock 1996, Higgins et al. 2005). Within these quadrats, I also obtained a single measurement of maximum saw palmetto height (cm). I assessed saw palmetto cover (%) along 2 perpendicular 20-m transects centered on the sampling point using the line intercept method (Stephenson and Buell 1965, Hays et al. 1981, Higgins et al. 2005).

Analyses

I performed repeated measures analyses using mixed model regressions, with season and time since treatment (time) as repeated measures and study site pair as a blocking factor, followed by Fisher's Protected LSD tests, to examine differences in saw palmetto height, cover, and density between untreated (control) and treated sites. Differences were examined within (e.g., dormant season burn) and among (i.e., dormant season burn, growing season burn, dormant season roller chop, growing season roller chop, and roller chop/burn) treatment types. In my results and discussion, I focus on treatment rather than repeated measures effects. I present results for treatment alone and 2- and 3-way treatment interactions in the text (Tables 3-1, 3-2, and 3-3). Due to the timing of data collection, it was not possible to test for 3-way interactions for growing

season burning and roller chopping treatments. When differences in linear combinations of groups or biologically meaningless comparisons (e.g., cover in dormant burn site in year 1 versus cover in control site in year 2) arose, I stated that post hoc comparisons revealed no differences based on treatment and the interacting factor. As 3-way interactions are difficult to reliably interpret, they were not discussed further (Zar 1999).

All data sets were rank transformed prior to analyses due to violations of normality and homogeneity of variance assumptions (Conover 1998, Zar 1999, SYSTAT 2007). I concluded statistical significance at $P \leq 0.1$ for all tests. This value was used, rather than the more common $P \leq 0.05$, to minimize the probability of making a Type II error (Mapstone 1995, Zar 1999). All statistical tests were performed using SYSTAT (2007) statistical software.

Results

Dormant Season Burn

Saw palmetto height was affected by a dormant season burning * time interaction (Table 3-2). Differences in height were not observed between dormant season burn and control sites the first or second year following treatment. However, saw palmetto height increased by 20% on burn sites from the first to the second year of the study. A dormant season burning * time interaction also affected saw palmetto cover (Table 3-2), which was 46% lower on burn than control sites the first year following treatment. No differences in saw palmetto cover between dormant season burn and control sites were observed the second year following treatment, with cover on burn sites increasing by 51% from the first to the second year of the study. Dormant season burning alone and

burning * time, burning * season, and burning * season * time interactions had no impact on saw palmetto density ($P \geq 0.593$).

Growing Season Burn

Growing season burning alone affected saw palmetto height (Table 3-1), which was 23% lower on burn than control sites. Saw palmetto cover was affected by a growing season burning * year interaction (Table 3-2), being 79% lower on growing season burn than control sites the first year following treatment. No differences in saw palmetto cover between growing season burn and control sites were observed the second year following treatment, with cover on burn sites increasing by 80% from the first to the second year of the study. Growing season burning alone and burning * time, burning * season interactions had no impact on saw palmetto density ($P \geq 0.161$).

Dormant Season Roller Chop

Dormant season roller chopping alone affected saw palmetto height (Table 3-1), which was 28% lower on dormant season roller chop than control sites. Saw palmetto cover was also affected by dormant season roller chopping alone (Table 3-1), being 56% lower on dormant season roller chop than control sites. Saw palmetto density was affected by a dormant season roller chopping * year interaction (Table 3-2). However, examination of post hoc comparisons revealed no differences in density based on roller chopping and year.

Growing Season Roller Chop

Growing season roller chopping alone affected saw palmetto height and cover (Table 3-1). Saw palmetto height was 40% lower on roller chop than control sites, and cover 70% lower on roller chop than control sites. Saw palmetto density was also

affected by growing-season roller chopping alone (Table 3-1), being 27% lower on roller chop than control sites.

Roller Chop/Burn

A roller chopping/burning * year interaction affected saw palmetto height (Table 3-2), which was 38% lower on roller chop/burn than control sites the first year following treatment. The second year following treatment, saw palmetto height was 23% lower on roller chop/burn than control sites. Saw palmetto cover was affected by a roller chopping/burning * season * year interaction ($P = 0.030$). Roller chopping/burning alone, and roller chopping/burning * time, roller chopping/burning * season, and roller chopping/burning * time * season interactions had no effect on saw palmetto density ($P \geq 0.265$).

Treatment Type Comparisons

A treatment * year interaction affected saw palmetto height (Table 3-3). The first year following treatment, height was lower on dormant and growing season burn, dormant and growing season roller chop, and roller chop/burn than control sites, but the effects of the 5 active treatments were similar. The second year following treatment, dormant season burning had no effect on saw palmetto height. However, height was lower on growing season burn, dormant and growing season roller chop, and roller chop/burn than control sites, with growing season roller chopping causing the greatest reduction. Saw palmetto height increased on dormant and growing season burn and roller chop/burn sites between the first and second year of the study.

Saw palmetto cover was also affected by a treatment * year interaction (Table 3-3). The first year following treatment, cover was lower on dormant and growing season burn, dormant and growing season roller chop, and roller chop/burn than control sites,

with similar effects of the 5 active treatments. The second year following treatment, dormant season burning had no effect on saw palmetto cover. However, cover was greater on growing season burn compared to control sites, and lower on dormant and growing season roller chop and roller chop/burn than control sites, with growing season roller chopping causing the greatest reduction. Saw palmetto cover increased on dormant and growing season burn and dormant season roller chop sites between the first and second year of the study.

Treatment type alone affected saw palmetto density (Table 3-3). Density was greater on dormant and growing season burn compared to control sites, but lower on dormant and growing season roller chop and roller chop/burn compared to control sites. The greatest reduction in saw palmetto density was observed on sites subject to growing season roller chopping.

Discussion

This study suggests a single dormant season burn is unlikely to be sufficient to reduce saw palmetto in pine flatwoods habitats, particularly when growth is dense and the cover excessive. Dormant season burning had no effect on saw palmetto density or height, and while cover was lower following dormant season burning, reductions were short-lived. Other studies suggest dormant season burning has minimal medium to long-term effects on many shrubs, with cover returning to pre-burn levels within 12 months post-burn (Abrahamson 1984a, b; Fitzgerald 1990; Robbins and Myers 1992; Glitzenstein et al. 1995). As a fire adapted species with rhizomes serving as large carbohydrate reserves and meristematic tissues that are protected from the flames during a burn, saw palmetto is able to resprout vigorously after fire and regain up to 80% of its pre-burn canopy cover after just 1 growing season (Hilmon 1968). Typically,

on pine flatwoods, in situations where rapid, significant shrub reduction is a primary goal, alternative management techniques should be utilized in place of or in addition to dormant season burning. However, regular dormant season burning with a 2–3 year return interval, where fuel loads permit, has been found sufficient to keep saw palmetto crown size small (Hilmon and Hughes 1965) and prevent saw palmetto proliferation. Therefore, this treatment is likely to be beneficial in reducing growth in situations where the shrub has not proliferated and maintenance at current levels is a goal. A fire return interval of ≥ 4 years can result in increases in saw palmetto cover (Hough 1968).

Growing season burning also had no effect on saw palmetto density. However, it appeared to cause greater reductions in saw palmetto height and cover in the short-term than did dormant season burning. Overall, treatment type comparisons suggest potentially greater and extended effects of growing season burning on shrub height and cover than dormant season burning. Studies examining shrub communities indicate regrowth is lower after growing than dormant season burning because of reduced shrub resprouting (Lewis and Harshbarger 1976, Fitzgerald 1990, Huffman and Blanchard 1991, Olson and Platt 1995, Drewa et al. 2002).

The effects of growing season burning on saw palmetto, although slightly greater than for dormant season burning, were still relatively temporary. However, the effects of both dormant and growing season roller chopping were greater and more prolonged than for either burn treatment. Dormant and growing season roller chopping resulted in lower saw palmetto height and cover than on control sites. However, saw palmetto density was only lower on growing season burn than control sites. In general, the effects of these treatments persisted through the duration of this study, although

comparisons suggest saw palmetto height and cover were lower on growing than dormant season roller chop sites. The longer-term effects of these 2 treatments are largely what separated them from the practice of growing season burning. Roller chopping can result in considerable decreases in shrub abundance and cover, and is useful on poor condition pine flatwoods with dense shrub cover, being particularly useful as a form of saw palmetto control (Moore 1974, Tanner et al. 1988). An 83% kill and reduction in crown cover to 3% of pre-treatment levels has been reported 2 years after practice implementation (Moore 1974), and a 70% kill and reduction in crown cover to 1–2% of pre-treatment levels 5 years after treatment (Hilmon 1968, Lewis 1970). However, many studies do not specifically examine effects of roller chopping on saw palmetto density. My results contradict those of a study which found saw palmetto density was lower on dry prairie roller chopped in the dormant versus the growing season (Watts and Tanner 2003, Watts et al. 2006). Chopping during periods of high soil moisture, i.e., during the growing season, may result in greater shrub reduction due to the greater penetration of the equipment's blades and deeper severing of plant roots (Moore 1974, Tanner et al. 1988). The additional stress placed on saw palmetto when roller chopped during the growing season, a period when it is actively producing new roots and stems, may result in greater reductions when compared to roller chopping conducted during the dormant season. Roller chopping, in either the dormant or growing season, is likely to be the best method for initial, rapid treatment and control of saw palmetto in high density and cover areas. It also has potential for use controlling saw palmetto on sites near urban developments and recreational areas, where smoke management issues may arise during and following burning.

Similar to dormant and growing season roller chopping, roller chopping/burning resulted in prolonged lowering of saw palmetto height compared to control sites. However, this treatment had no effect on saw palmetto density and its effect on saw palmetto cover was unclear due to an interaction with season and year. The effects of roller chopping/burning on plant communities have not been well studied. Other studies have found a reduction in saw palmetto density on burned and roller chopped plots (Watts and Tanner 2003, Watts et al. 2006). Burning prior to roller chopping may remove some of the above ground shrub biomass making subsequent roller chopping more effective in terms of shrub reduction (Kalmbacher and Martin 1984). During this study, combination treatment sites were burned after roller chopping and this may be why decreases in density were not observed, as fire can stimulate saw palmetto sprouting (Hilmon 1968). Further study of roller chopping/burning combination treatments is recommended to determine if there are differences based on timing of burning and roller chopping practices.

Management Implications

The single application of a dormant or growing season prescribed burn will likely be insufficient to reduce saw palmetto growth in areas where it has proliferated. In such situations, burn effects on saw palmetto height and cover are likely to be minor and short-lived, with no observable effect on density. Alternative management practices such as roller chopping will likely be needed in combination with, or in place of, dormant and growing season burning if there is a need to quickly and significantly reduce excessive saw palmetto height, cover, and density.

If rapid, more prolonged control and greater reductions in saw palmetto height, cover, and density are desired, dormant or growing season roller chopping should be

conducted. These practices result in decreases in saw palmetto height and cover for at least 2 years post-treatment. If reductions in shrub density are desired, growing season roller chopping may be the only suitable treatment.

Table 3-1. Effects of treatment on saw palmetto height, cover, and density in Florida flatwoods, 2007–2008.

Saw Palmetto Variables by Treatment ^a	Treatment ($\bar{x} \pm SE$)		P
	Control	Treated	
Growing season burning			
Cover (%)	60.5 ± 2.8	46.7 ± 4.3	0.001
Dormant season roller chop			
Height (cm)	101.8 ± 3.0	73.4 ± 2.5	≤0.001
Cover (%)	45.2 ± 1.7	20.0 ± 2.0	≤0.001
Growing season roller chop			
Height (cm)	95.8 ± 3.8	57.4 ± 2.7	≤0.001
Cover (%)	34.6 ± 2.2	10.1 ± 1.6	≤0.001
Density (no./m ²)	1.5 ± 0.0	1.1 ± 0.1	0.032
Roller chop/burn			
Height (cm)	105.2 ± 3.7	74.8 ± 3.2	≤0.001
Cover (%)	50.0 ± 3.1	20.2 ± 2.1	≤0.001

^a Only saw palmetto variables significantly affected by treatment presented ($P \leq 0.1$).

Table 3-2. Effects of treatment * time interactions on saw palmetto height, cover, and density in Florida flatwoods, 2007–2008.

Saw Palmetto Variables by Treatment ^a	Time ^b	Treatment ($\bar{x} \pm SE$) ^{c,d}		P
		Control	Treated	
Dorman season burn				
Height (cm)	1	96.6 ± 7.2 _{Aa}	84.4 ± 4.8 _{Aa}	0.073
	2	114.2 ± 4.4 _{Aa}	118.2 ± 4.2 _{Ab}	
Cover (%)	1	45.2 ± 4.7 _{Aa}	24.4 ± 3.8 _{Ba}	0.021
	2	51.2 ± 4.4 _{Aa}	50.2 ± 3.9 _{Ab}	
Growing season burn				
Cover (%)	1	57.5 ± 5.4 _{Aa}	12.1 ± 3.6 _{Ba}	0.005
	2	61.5 ± 3.3 _{Aa}	58.2 ± 3.3 _{Ab}	
Dormant season roller chop				
Density (no./m ²)	1	1.7 ± 0.1 _{Aa}	1.2 ± 0.1 _{Aa}	0.023
	2	1.4 ± 0.1 _{Aa}	1.4 ± 0.1 _{Aa}	
Roller chop/burn				
Height (cm)	1	100.8 ± 7.9 _{Aa}	62.1 ± 4.8 _{Ba}	0.041
	2	108.0 ± 3.5 _{Aa}	82.8 ± 3.5 _{Ba}	

^a Only saw palmetto variables significantly affected by a treatment * time interaction presented ($P \leq 0.1$).

^b Time since treatment application (years).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

^d Means in a column followed by the same lowercase letter not significantly different ($P > 0.1$).

Table 3-3. Comparisons of the effects of treatment and treatment * time interactions on saw palmetto height, cover, and density in Florida flatwoods, 2007–2008.

Saw Palmetto Characteristics by Treatment and Interaction ^a	Time ^b	Treatment Type ($\bar{x} \pm SE$) ^{b,c}						<i>P</i>
		Control	Dormant Season Burn	Growing Burn	Dormant Season Roller Chop	Growing Season Roller Chop	Roller Chop/Burn	
Treatment								
Density (No./m ²)	N/A	1.6 ± 0.0 _A	1.7 ± 0.1 _B	2.2 ± 0.1 _B	1.3 ± 0.1 _C	1.1 ± 0.1 _D	1.5 ± 0.1 _C	≤0.001
Treatment * Year								
Height (cm)	1	98.3 ± 3.3 _{Aa}	84.4 ± 4.8 _{Ba}	51.2 ± 5.7 _{Ba}	118.2 ± 4.2 _{Ba}	59.1 ± 6.4 _{Ba}	62.1 ± 4.8 _{Ba}	≤0.001
	2	106.1 ± 1.9 _{Aa}	118.2 ± 4.2 _{ABb}	92.2 ± 3.4 _{BCb}	67.9 ± 3.7 _{Ca}	56.8 ± 2.9 _{Da}	82.8 ± 3.5 _{Cb}	
Cover (%)	1	45.3 ± 2.2 _{Aa}	24.4 ± 3.8 _{Ba}	12.1 ± 3.6 _{Ba}	12.6 ± 1.8 _{Ba}	6.9 ± 3.0 _{Ba}	11.5 ± 2.0 _{Bb}	≤0.001
	2	48.7 ± 1.6 _{Aa}	50.2 ± 3.9 _{ABb}	58.2 ± 3.3 _{Bb}	26.7 ± 2.8 _{Cb}	11.2 ± 1.9 _{Da}	25.7 ± 2.7 _{Cc}	

^a Only saw palmetto variables significantly affected by treatment or a treatment * time interaction presented ($P \leq 0.1$).

^b Time since treatment application (years).

^c Means in a row followed by the same letter not significantly different ($P > 0.1$).

^d Means in a column followed by the same lowercase letter not significantly different ($P > 0.1$).

CHAPTER 4
INFLUENCE OF ROLLER CHOPPING AND BURNING ON ARTHROPOD
COMMUNITIES OF FLORIDA RANGELANDS

Introduction

Frequent fires are considered essential to maintain the structure and diverse herbaceous groundcover of many southeastern rangeland habitats (Christensen 1981, Abrahamson and Hartnett 1990, Platt 1998). However, in many situations, fire exclusion, reductions in fire frequency, and/or a shift in fire season have resulted in excessive shrub growth and declines in the species rich herbaceous ground layer of these habitats (Wade et al. 1980, Platt et al. 1988, Huffman and Blanchard 1991, Glitzenstein et al. 1999), potentially reducing their value to livestock and certain wildlife species. Depending on season of application, roller chopping and prescribed burning have been shown to improve southeastern rangeland condition by reducing the cover of shrubs such as saw palmetto (*Serenoa repens* [bartr.] Small), gallberry (*Ilex glabra* [Pursh] Chapm.), and wax myrtle (*Morella cerifera* [L.] Small), and promoting the growth and seeding of herbaceous groundcover species (Chapter 2; Wade et al. 1980, Kalmbacher and Martin 1984, Tanner et al. 1988, Glitzenstein et al. 1995, Watts and Tanner 2003). In Florida, the use of these practices in a variety of rangeland habitats is promoted as a means to maintain or enhance wildlife habitat, increase livestock forage quantity and quality, and reduce fuel build up and wildfire risk.

Arthropods are a critical component of rangeland systems and their management and conservation should be considered when implementing roller chopping and prescribed burning activities. They are major contributors to biodiversity and play an important role in ecosystem processes as pollinators and predators, providing benefits to agricultural and livestock producers (Warren et al. 1987, Triplehorn and Johnson

1995). In addition, they provide an important food source for rangeland-associated wildlife. In Florida, this includes numerous avian species considered of conservation priority e.g., northern bobwhite (*Colinus virginianus* L.), Bachman's sparrow (*Aimophila aestivalis* Lichtenstein), common ground-dove (*Columbina passerina* L.), and grasshopper sparrow (*Ammodramus savannarum* Gmelin; Vickery 1996, Brennan 1999, Bowman 2002, Dunning 2006). In contrast, some arthropods cause extensive damage to grasslands and crops. Nymphal, larval, and adult arthropods can cause considerable injury to the leaves, stems, roots, and reproductive structures of plants (Warren et al. 1987), while some phytophagous species transmit plant diseases (Hardwood and James 1979). In addition, many arthropods are parasites of humans, livestock, and wildlife (Hardwood and James 1979). Despite their importance, the effects roller chopping and prescribed burning have on the arthropod communities of Florida's rangelands have not been extensively studied (Robbins and Myers 1992, Hanula and Wade 2003).

In other rangeland systems across the United States, arthropod response to fire has been shown to be highly variable and influenced by a variety of factors including order, family, or species examined, mobility, life stage at time of burning, burn frequency, degree of flame exposure, and reaction to changes in community composition and habitat (Lussenhop 1976; Seastedt 1984; Warren et al. 1987; Anderson et al. 1989; Siemann et al. 1997; Swengel 1996, 1998; Hanula and Wade 2003). Many rangeland, particularly grassland and savanna, arthropods are fire adapted (Evans 1984, Anderson et al. 1989, Siemann et al. 1997). However, the application of prescribed burning to Florida's rangeland habitats may cause alterations

to associated arthropod communities as a result of changes in vegetative structure and composition (Herman et al. 1998). The response of the vegetative community to roller chopping is also likely to result in alterations to associated arthropod communities. A more comprehensive understanding of how Florida rangeland arthropod communities respond to roller chopping and prescribed burning is needed to ensure they are used appropriately in situations where arthropod management or conservation are a consideration. The objectives of my study were to 1) compare composition (e.g., abundance and richness) of arthropod communities on treated (management activities implemented) and untreated (no management activities implemented) pine flatwoods sites during dormant (November–March) and growing (April–October) seasons and 2) examine the effects local pine flatwoods habitat characteristics (e.g., shrub density, forb cover, and graminoid height) have on the composition (e.g., abundance and richness) of arthropod communities.

Methods

Study Sites

I conducted research on 50 paired treatment and control pine flatwoods sites with varying management (i.e., roller chopping and prescribed burning) histories and grazing regimes. When grazed, both the treatment and paired control study sites were subject to similar grazing pressures at similar times. Flatwoods sites were located on privately- and publically-owned lands across a 6 county area (Desoto, Highlands, Lee, Manatee, Osceola, and Sarasota) in central and south Florida, and were being managed by landowners and managers using a variety of prescribed burning and roller chopping protocols. Florida's pine flatwoods are rangelands characterized as having a pure or combined overstory stand of scattered longleaf (*Pinus palustris* Mill.) and slash (*P.*

elliotti Engelm.) pine and an often diverse herbaceous layer. This herbaceous layer is comprised of a wide variety of grasses (e.g., *Agrostis* spp., *Andropogon* spp., *Aristida* spp., *Eragrostis* spp., *Panicum* spp., and *Paspalum* spp.). Common forbs include legumes (e.g., *Cassia* spp., *Crotalaria* spp., *Galactia* spp., *Tephrosia* spp.), milkweeds (*Asclepias* spp.), milkworts (*Polygala* spp.), and a range of composites (e.g., *Aster* spp., *Chrysopsis* spp., *Eupatorium* spp., *Liatrix* spp., and *Solidago* spp.). The understory and shrub layer includes saw palmetto, gallberry, wax myrtle, fetterbush (*Lyonia lucida* [Lam.] K. Koch), staggerbush (*Lyonia fruticosa* [Michx] G. S. Torr), dwarf huckleberry (*Gaylussacia dumosa* [Andrews] Torr. & A. Gray), dwarf live oak (*Quercus mimima* [Sarg] Small), and tarflower (*Befaria racemosa* Vent.; Abrahamson and Hartnett 1990, United States Fish and Wildlife Service 1999).

Treatment Types

Treated sites were subject to 1 of 5 treatment types: dormant season (November–March) roller chop, growing season (April–October) roller chop, dormant season burn, growing season burn, or a roller chop/burn combination treatment. The roller chop/burn combination treatment (hereafter referred to as roller chop/burn) involved roller chopping in the dormant season followed by burning within 6 months. I established a total of 9 dormant season roller chop, 12 growing season roller chop, 11 dormant season burn, 9 growing season burn, and 9 roller chop/burn and control pairs.

Arthropod Sampling

I used a paired sampling approach to assess the effects of treatment type (i.e., roller chopping, prescribed burning, and roller chopping/burning) on arthropod familial richness and abundance. Richness and abundance were compared between sampling points randomly located in paired treated (e.g., dormant season roller chopped) and

untreated (control) flatwoods sites. Paired treatment and control sampling points were adjacent, being located in the same pasture or management unit, and of similar current and past management (e.g., grazing intensity), surrounding land-use, plant community (e.g., overstory cover), and soil conditions. Within each treatment and control site, I established 1 randomly selected sampling point. To minimize edge effects, sampling points that fell within 50 m of the edge of a treatment or control site were rejected and randomly relocated. Sites within which treatment and control sampling points were located ranged from 2-20 ha.

I collected arthropods at each sampling point once in winter (February–March), spring (April–May), and summer (July–August), during each of 2 years (2007–2008) following treatment. Sub-samples of arthropods occupying vegetation less than 30 cm above the ground were taken from within 4 1-m² plots, randomly located in each quadrant of a 0.03-ha nested circular plot centered on the sampling point (Dueser and Shugart 1978, Higgins et al. 2005). Arthropods were sampled using a suction sampler (Wright and Stewart 1993, Ausden 1996). Within each 1-m² plot, the suction sampler was turned on and systematically moved around the sub-sample area, no more than 30 cm above the ground, for a 3-minute period collecting arthropods. Suction sampling was not conducted if vegetation was damp or had been flattened by wind, rain, or trampling (Ausden 1996). I separated arthropods collected in each suction sub-sample from coarse vegetation and combined them in a vial containing a preservation agent of 75% ethanol and 25% distilled water (Schauff 1986).

I collected sub-samples of mobile arthropods and arthropods occupying vegetation more than 30 cm above the ground along 2 perpendicular 20-m transects centered on

the sampling point (Dueser and Shugart 1978, Higgins et al. 2005). Arthropods were sampled using a sweep net (Ausden 1996). I made 50 sweeps (1 sweep comprising a forward and backward stroke of the sweep net) along each of the 20 m transects, ensuring the sweep net did not pass within 30 cm of the ground (Schauff 1986). Arthropods collected in each sweep net sub-sample were combined and preserved as described for those collected using suction sampling. In the laboratory, I identified arthropods contained in each suction and sweep net sample to the family level using a microscope and appropriate identification keys (Triplehorn and Johnson 2005, Ubick et al. 2005).

Habitat Sampling

Habitat sampling was conducted once in winter (February–March), spring (April–May), and summer (July–August), during each of 2 years (2007–2008) following treatment. Within each of the 0.03-ha nested circular plots established at arthropod sampling points, I examined a variety of habitat characteristics including herbaceous and shrub community composition and structure and ground layer variables (Dueser and Shugart 1978, Higgins et al. 2005).

Ground layer. I assessed litter cover (%; ocular estimate) within 4 1-m² sub-sample plots, 1 randomly located in each quadrant of the 0.03-ha circular plot, along with soil density (g/cm³), moisture (%), and pH. Litter cover was recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Donhaue et al. 1971, Hays et al. 1981, Higgins et al. 2005). I recorded soil density as the dry weight density (g/cm³) of a 5-cm diameter 10-cm deep soil core sample after oven drying at 45°C for 48 hours. Soil pH and moisture were measured using a Kelway soil tester (Rodewald and Yahner 2001).

Herbaceous layer. I measured species richness (no. of species), cover (%; ocular estimate), and maximum height (cm) of forbs and graminoids within the 1-m² subsample plots. Forb and graminoid cover were recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Hays et al. 1981, Krebs 1999, Higgins et al. 2005).

Shrub layer. I counted and determined the height of all shrubs (woody vegetation <2.0 m in height) in 2 perpendicular 20-m² quadrats centered on the 0.03-ha plot to estimate species richness (no. of species), density (no./m²), and maximum height (cm) for individual species and all combined (Hays et al. 1981, Krebs 1999, Higgins et al. 2005). Shrub cover (%) was assessed along 2 perpendicular 20-m transects centered on the 0.03-ha circular plot using the line intercept method (Hays et al. 1981, Higgins et al. 2005).

I recorded the presence of livestock on study sites for incorporation into analyses.

Analyses

Prior to analyses, I combined data collected from suction and sweep-net samples. I analyzed differences in total arthropod familial richness and abundance (all families combined) and familial richness and abundance by order (orders with abundance ≥ 500 individuals) between treated and untreated (control) sites. Differences were examined both within (e.g., dormant season roller chop) and among (i.e., dormant season roller chop, growing season roller chop, dormant season burn, growing season burn, and roller chop/burn) treatment types using repeated measures mixed model regressions. Repeated measures were season and time since treatment (time). Study site pair was included as a blocking factor and presence of grazing as an additional influential independent variable. I used Fisher's Protected LSD tests to make post hoc

comparisons. In my results and discussion, I focused on treatment rather than repeated measures or grazing effects. I presented results for 2-, 3-, and 4- way treatment interactions. Due to the timing of data collection, it was not possible to test for 3-way interactions for growing season burning and roller chopping treatments. As 3- and 4-way treatment interaction effects are difficult to reliably interpret, they were not discussed further (Zar 1999). If, when examining two-way treatment interactions, differences in linear combinations of groups or biologically meaningless comparisons (e.g., arthropod abundance in growing season roller chop sites in year 1 versus arthropod abundance in control sites in year 2) arose, I stated that post hoc comparisons revealed no differences based on treatment and the interacting factor.

Multiple linear regression was used to examine which combination of habitat characteristics best described changes in arthropod family richness and abundance. I reduced multicollinearity problems by subjecting all predictor variables involved in pairwise correlations with $r \geq 0.7$ to a univariate, one-way analysis of variance (ANOVA) with each dependent variable. For each pair of highly correlated predictor variables, I retained the one with the greatest F value (Noon 1981, McGarigal et al. 2000). All regression models were fit using a backward stepwise procedure with Tolerance = 0.001, F -to-enter = 0.15, and F -to-remove = 0.15. These values are considered appropriate for predictor variables that are relatively independent (SYSTAT 2007). I considered regression models statistically and biologically significant at $P \leq 0.1$ and $R^2 \geq 0.2$. Only models considered significant were presented. The relative importance of each variable in the best model was assessed by examining standardized regression

coefficients (SC; i.e., variables with higher coefficients made greater individual contributions to the explanatory power of the model).

I rank transformed all data prior to analyses due to violations of normality and homogeneity of variance assumptions (Conover 1998, Zar 1999, SYSTAT 2007). To minimize the probability of making a Type II error, I concluded statistical significance for all tests at $P \leq 0.1$, rather than the more common $P \leq 0.05$ (Zar 1999). All statistical tests were performed using SYSTAT (2007) statistical software.

Results

During the study, I collected arthropods from 24 orders and 162 families (Table 4-1). Of these orders, Orthoptera, Coleoptera, Hemiptera, Diptera, Araneida, Hymenoptera, and Blattodea had an abundance ≥ 500 individuals and were subject to further analyses.

Dormant Season Roller Chop

Total arthropod familial richness was affected by dormant season roller chopping alone (Table 4-2), being 15% lower on roller chop than control sites. Dormant season roller chopping also affected Araneida familial richness, which was 18% lower on roller chop than control sites. Hemiptera, Diptera, and Hymenoptera familial richness were affected by a dormant season roller chopping * time interaction (Table 4-3). However, post hoc comparisons revealed no differences in richness between roller chop and control sites for these arthropod orders based on time. A dormant season roller chopping * season interaction also affected Hemiptera, Diptera, and Hymenoptera familial richness (Table 4-4). In winter, Hemiptera familial richness was 38% lower on roller chop than control sites. Diptera familial richness was 14% greater on roller chop than control sites in spring and 9% lower on roller chop than control sites in winter. In

spring, Hymenoptera familial richness was 67% lower on roller chop than control sites. Blattodea familial richness was affected by a dormant season roller chopping * grazing interaction (Table 4-5). However, no differences in Blattodea familial richness between roller chop and control sites based on grazing were observed from post hoc comparisons. Dormant season roller chopping alone and in all combinations with time, season, and grazing had no effect on Coleoptera familial richness ($P \geq 0.260$).

Dormant season roller chopping alone affected total arthropod abundance (Table 4-2), which was 20% lower on roller chop than control sites. Orthoptera, Hemiptera, and Araneida abundance were also affected by dormant season roller chopping alone. Orthoptera abundance was 16% lower, Hemiptera abundance 22% lower, and Araneida abundance 35% lower on roller chop than control sites. Diptera and Hymenoptera abundance were affected by a dormant season roller chopping * time interaction (Table 4-3). However, post hoc comparisons revealed no differences in abundance between roller chop and control sites for these arthropod orders based on time. Diptera abundance was also affected by a dormant season roller chopping * season interaction, being 18% lower on roller chop than control sites in spring and 28% greater on roller chop than control sites in winter. A dormant season roller chopping * grazing interaction affected Diptera and Blattodea abundance (Table 4-5). Diptera abundance was 17% higher on nongrazed dormant season roller chop than control sites, but 46% lower on grazed dormant season roller chop than control sites. Examination of post hoc comparisons revealed no differences in Blattodea abundance between roller chop and control sites based on grazing. Coleoptera abundance was unaffected by dormant

season roller chopping alone and in all combinations with time, season, and grazing ($P \geq 0.327$).

Growing Season Roller Chop

Total arthropod familial richness was affected by a growing season roller chopping * time interaction (Table 4-3), being 38% lower on roller chop than control sites the first year following treatment. However, no differences in total arthropod richness were observed the second year following treatment. Growing season roller chopping alone affected Blattodea familial richness, which was 57% lower on roller chop than control sites (Table 4-2). A growing season roller chopping * time interaction also affected Orthoptera familial richness, but post hoc comparisons revealed no differences in richness between roller chop and control sites based on time. Hemiptera familial richness was affected by a growing season roller chopping * grazing interaction (Table 4-5). However, no differences in Hemiptera familial richness between roller chop and control sites based on grazing were observed from post hoc comparisons. Dormant season roller chopping alone and in all combinations with time, season, and grazing had no effect on Coleoptera familial richness ($P \geq 0.338$).

Diptera and Blattodea abundance were affected by growing season roller chopping (Table 4-2), being 32% and 50% lower on roller chop than control sites, respectively. A growing season roller chopping * time interaction affected Orthoptera abundance (Table 4-3), which was 55% lower on roller chop than control sites the first year following treatment. However, no differences in Orthoptera abundance were observed between roller chop and control sites the second year following treatment. Total arthropod abundance and Coleoptera abundance were unaffected by growing

season roller chopping alone and in all combinations with time, season, or grazing ($P \geq 0.254$).

Dormant Season Burn

Total arthropod familial richness was affected by a dormant season burning * time interaction (Table 4-3), being 49% lower on burn than control sites the first year following treatment. However, no differences in total arthropod familial richness were observed between burn and control sites the second year following treatment. Total arthropod familial richness was also affected by a dormant season burning * grazing interaction (Table 4-5). However, post hoc comparisons revealed no differences in total arthropod familial richness between burn and control sites based on grazing. A dormant season burning * time interaction also affected Orthoptera, Hemiptera, Diptera, and Hymenoptera familial richness. Orthoptera familial richness was 42% lower, Hemiptera richness 62% lower, Hemiptera familial richness 64% lower, and Hymenoptera familial richness 45% lower on burn than control sites the first year following treatment. However, there were no differences in richness between burn and control sites for these arthropod orders the second year following treatment. Hymenoptera familial richness were affected by a dormant season burning * grazing interaction (Table 4-5). However, post hoc comparisons revealed no differences in richness for this order between burn and control sites based on grazing. Dormant season burning alone and in all combinations with time, season, and grazing had no effect on Coleoptera familial richness ($P \geq 0.147$).

A dormant season burning * time interaction affected total arthropod abundance, which was 64% lower on burn than control sites the first year following treatment (Table 4-3). The second year following treatment, there were no differences in total arthropod

abundance between burn and control sites. Orthoptera, Hemiptera, Diptera, Araneida, and Hymenoptera abundance were also affected by a dormant season burning * time interaction. Orthoptera abundance was 68% lower, Hemiptera abundance 74% lower, and Hymenoptera abundance 73% lower on burn than control sites the first year following treatment. However, there were no differences in abundance between burn and control sites for these orders the second year following treatment. Post hoc comparisons revealed no differences in Diptera or Araneida abundance between burn and control sites based on time. A dormant season burning * grazing interaction affected Coleoptera and Blattodea abundance (Table 4-5). Coleoptera abundance was 69% greater and Blattodea abundance 125% greater on Nongrazed burn than control sites. There was no difference in abundance between grazed burn and control sites for these arthropod orders.

Growing Season Burn

A growing season burning * time interaction affected total arthropod familial richness (Table 4-3), which was 56% lower on burn than control sites the first year following treatment. There were no differences in total arthropod familial richness between burn and control sites the second year following treatment. Hemiptera familial richness was affected by growing season burning alone (Table 4-2), being 23% lower on burn than control sites. Araneida familial richness was affected by a growing season burning * time interaction, being 57% lower on burn than control sites the first year following treatment. The second year following treatment there was no differences in Araneida familial abundance between burn and control sites. Blattodea familial richness was affected by a growing season burning * season interaction (Table 4-4). In winter, Blattodea familial richness was 89% lower on burn than control sites. Coleoptera

familial richness was affected by a growing season burning * season * grazing interaction ($P = 0.063$). Orthoptera, Diptera, and Hymenoptera familial richness were unaffected by growing season burning alone and in all combinations with time, season, and grazing ($P \geq 0.105$).

Total arthropod abundance was affected by growing season burning alone (Table 4-2), being 40% lower on burn than control sites. Hemiptera and Araneida abundance were also affected by growing season burning. Hemiptera abundance was 40% lower and Araneida abundance 50% lower on burn than control sites. A growing season burning * season interaction affected Blattodea abundance (Table 4-4). In winter, Blattodea abundance was 96% lower on burn than control sites. Orthoptera abundance was affected by a growing season burning * grazing interaction (Table 4-5). However, post hoc comparisons revealed no differences in abundance for this order between burn and control sites based on grazing. Coleoptera and Diptera abundance were affected by a growing season burning * time * season * grazing interaction ($P \leq 0.069$). Dormant season burning alone and in all combinations with time, season, and grazing had no effect on Hymenoptera abundance ($P \geq 0.363$).

Roller Chop/Burn

Orthoptera and Blattodea familial richness were affected by a roller chopping/burning * time interaction (Table 4-3). Blattodea familial richness was 88% lower on roller chop/burn sites the first year following treatment. However, there was no difference in Blattodea familial richness between roller chop/burn sites the second year following treatment. Examination of post hoc comparisons revealed no differences in Orthoptera familial richness between roller chop/burn and control sites based on time. Coleoptera familial richness was affected by a roller chopping/burning * season

interaction (Table 4-4). In spring, Coleoptera familial richness was 43% lower on roller chop/burn than control sites. A roller chopping/burning * time * season interaction affected Hemiptera and Araneida familial richness ($P \leq 0.040$). Roller chopping/burning alone and in all combinations with time, season, and grazing had no effect on total arthropod familial richness and Diptera and Hymenoptera familial richness ($P \geq 0.174$).

Roller chopping/burning alone affected Diptera abundance (Table 4-2), which was 32% lower on roller chop/burn than control sites. A roller chopping/burning * time interaction affected Blattodea abundance, but examination of post hoc comparisons revealed no differences in abundance between roller chop/burn and control sites based on time. Coleoptera abundance was affected by a roller chopping/burning * season interaction. In winter, Coleoptera abundance was 343% greater on roller chop/burn than control sites. Total arthropod abundance and Orthoptera, Hemiptera, Araneida, and Hymenoptera abundance were affected by a roller chopping/burning * time * season interaction ($P \leq 0.098$)

Treatment Type Comparisons

A treatment type * time interaction affected total arthropod familial richness (Table 4-6). The first year following treatment total arthropod familial richness was lower on growing season roller chop and dormant and growing season burn compared to control sites, but the effects of the 3 treatments were similar. There were no differences in total arthropod familial richness between treatment and control sites during the second year of the study. Orthoptera, Diptera, Araneida, Hymenoptera, and Blattodea familial richness were also affected by a treatment type * time interaction. Orthoptera familial richness was lower on growing season roller chop and dormant and growing season burn compared to control sites during the first year of the study. However, the 3

treatments had similar effects. The second year following treatment, there were no differences in Orthoptera familial richness between treatment and control sites. Hymenoptera familial richness was lower on dormant season burn compared to control sites the first year following treatment and greater on dormant season burn compared to control sites the second year following treatment. Blattodea familial richness was lower on dormant and growing season burn and roller chop/burn sites during the first year of the study. There were no differences in Blattodea familial richness between treatment and control sites during the second year of the study. Post hoc comparisons revealed no differences in Diptera and Araneida familial richness based on treatment * time. Orthoptera and Hymenoptera familial richness were affected by a treatment type * season interaction (Table 4-7). Orthoptera familial richness was greater on roller chop/burn sites in winter, and lower on growing season roller chop sites in spring. Hymenoptera familial richness was also lower on dormant season roller chop and roller chop/burn sites in spring.

Total arthropod abundance was affected by a treatment type * time interaction (Table 4-6), being lower on dormant and growing season burn compared to control sites the first year following treatment. However, the effects of the 2 treatments were similar. The second year following treatment there were no differences in total arthropod abundance based on treatment type. Orthoptera, Hemiptera, Diptera, Araneida, Hymenoptera, and Blattodea abundance were also affected by a treatment * time interaction. During the first year of the study, Orthoptera abundance was lower on growing season roller chop and dormant and growing season burn than control sites, with similar effects of the 3 treatments. Hemiptera abundance was lower on dormant

and growing season burn than control sites the first year following treatment. However, the effects of the 2 treatments were similar. Araneida abundance was lower on dormant season roller chop and growing season burn compared to control sites during the first year of the study, but there were no differences in the effects of these 2 treatments. The first year following treatment, Hymenoptera abundance was lower on dormant season burn sites. There was no difference in Orthoptera, Hemiptera, Araneida, and Hymenoptera abundance based on treatment type during the second year of the study. Blattodea abundance was lower on dormant and growing season burn and roller chop/burn sites the first year following treatment, with similar effects of the 3 treatments. The second year following treatment Blattodea abundance was lower on growing season roller chop compared to control sites. Examination of post hoc comparisons revealed no difference in Diptera abundance based on treatment type and year. Coleoptera and Araneida abundance were affected by a treatment type * season interaction (Table 4-7). Coleoptera abundance was lower on roller chop/burn sites in winter, and greater on growing season roller chop compared to control sites in summer. In spring, Araneida abundance was lower on growing season burn, growing season roller chop, and roller chop/burn than control sites.

Arthropod-Habitat Relationships

Habitat characteristics that best predicted total arthropod familial richness were mean forb height (SC = 0.215), mean litter depth (SC = 0.201), mean soil moisture (SC = 0.129), mean graminoid height (SC = 0.124), shrub density (SC = 0.120), variance of soil density (SC = -0.104), variance of forb cover (SC = 0.074), mean graminoid species richness (SC = 0.073), and mean soil density (SC = -0.064; $P \leq 0.001$, $R^2 = 0.270$). The combined effects of mean litter depth (SC = 0.203), mean soil moisture (SC = 0.177),

mean forb cover (SC = 0.134), mean graminoid species richness (SC = 0.124), shrub height (SC = 0.121), mean graminoid height (SC = 0.117), variance of soil density (SC = -0.105), shrub species richness (SC = 0.101), and mean soil pH (SC = -0.076; $P \leq 0.001$, $R^2 = 0.262$) best explained Araneida richness.

Habitat characteristics that best predicted total arthropod abundance were mean litter depth (SC = 0.193), mean forb height (SC = 0.181), mean graminoid height (SC = 0.159), mean soil moisture (SC = 0.141), variance of soil density (SC = -0.135), shrub density (SC = 0.104), variance of litter depth (SC = -0.096), variance of forb cover (SC = 0.092), mean forb species richness (SC = 0.062; $P \leq 0.001$, $R^2 = 0.284$). Shrub height (SC = -0.236), mean litter depth (SC = 0.233), mean forb richness (SC = 0.195), mean forb height (SC = 0.133), mean graminoid cover (SC = 0.129), shrub cover (SC = 0.107), mean soil pH (SC = 0.103), and variance of litter cover (SC = -0.085; $P \leq 0.001$, $R^2 = 0.242$) were the habitat characteristics that best described Orthoptera abundance. The combined effects of mean litter depth (SC = 0.210), mean soil moisture (SC = 0.180), mean graminoid height (SC = 0.173), mean forb height (SC = 0.152), variance of soil density (SC = -0.120), shrub species richness (SC = 0.096), mean soil pH (SC = -0.082), shrub height (SC = 0.080), variance of litter cover (SC = -0.056; $P \leq 0.001$, $R^2 = 0.308$) best explained Araneida abundance.

Discussion

Dormant season roller chopping led to a prolonged decrease in total arthropod familial richness and abundance that lasted throughout the 2 years of this study. Growing season roller chopping resulted in lower total arthropod familial richness, but the effect was short-lived occurring the first year following treatment only. Based on research by Panzer (1988) in prairie remnants, Hall and Schweitzer (1992) suggested

that, in southeastern flatwoods, many arthropods would survive and recolonize best after growing season burning, as this is when they are typically most mobile. This study suggests growing season roller chopping may also be preferable to dormant season roller chopping to maintain arthropod familial richness, possibly because it occurs when the majority of arthropods are mobile and able to flee roller chopped areas where conditions are unsuitable. During the dormant season, many species are in an inactive, nymphal stage below ground or beneath dead wood (Swengel 2001). Disturbance of soil and logs by a passing roller chopper during this period may result in the loss of many of these dormant individuals, later reducing the abundance of more mobile life stages.

Examination of arthropod orders indicated that dormant season roller chopping caused prolonged reductions in Araneida familial richness and Araneida and Hemiptera abundance that lasted throughout the 2 years of the study. Growing season roller chopping caused prolonged reductions in Blattodea familial richness and Blattodea and Diptera abundance that also lasted throughout duration of the study. In addition, short-term reductions in Orthoptera abundance that lasted throughout the first year of the study were observed following this treatment. Studies examining the effects of roller chopping on arthropods are scarce. Roller chopping in both seasons causes short-term reductions in litter depth and cover, graminoid and forb cover and height, and shrub cover and height (Chapter 2), potentially reducing food and cover availability for members of these orders and in turn their familial richness and abundance. Reductions in graminoid and forb cover are likely to be a particular concern for many Orthoptera, the majority of which are herbivorous and rely on herbaceous plants as a food source

(Warren et al. 1987). Blattodea and certain Orthoptera are ground dwelling, living in the litter layer (Warren et al. 1987). Removal of litter and disturbance of the soil during roller chopping may contribute to reductions in abundance within these orders both directly and indirectly.

Dormant and growing season burning resulted in lower total arthropod familial richness and abundance. However, reductions in richness and abundance were more prolonged on growing season burn sites, lasting throughout the 2 years of the study. On dormant season burn sites, changes were relatively short-lived, occurring only during the first year of the study. These findings do not support the suggestion that southeastern flatwoods arthropods survive and recolonize best following growing season burning, a period when they are most mobile (Hall and Schweitzer 1992). In contrast arthropods may be better able to survive dormant than growing season burning. During the dormant season, many arthropods are in an inactive, nymphal state below ground and beneath woody debris. This may offer them protection from passing flames during a burn (Swengal 2001).

Examination of arthropod orders indicated that dormant season burning led to short-term reductions in Orthoptera, Hemiptera, and Hymenoptera familial richness and abundance. Growing season burning had a prolonged effect on Hemiptera familial richness and Hemiptera and Araneida abundance, which lasted throughout the 2 years of the study, and a short-term affect on Araneida familial richness. The majority of Orthopterans are voracious plant feeders and can be negatively affected by fire due to a reduction in herbaceous plant foods (Evans 1984). Dormant and growing season burning have both been found to cause reductions in graminoid and forb height and

cover on pine flatwoods sites for at least the first year post-treatment (Chapter 2). Reduced Orthopteran numbers following burning have also been attributed to reduced litter cover (Tester and Marshall 1961). Reductions in litter depth and cover are considerable following prescribed burning of pine flatwoods habitats and can be prolonged, lasting for at least 2 years post-treatment (Chapter 2). Accumulation of litter has been found to restore Orthoptera populations within 1 year of treatment (Tester and Marshall 1961). Other studies suggest that Orthoptera abundance may increase on burned sites in the first to second year post-treatment (Nagel 1973, Reed 1997, Chambers and Samways 1998). Short-term reductions in Hemiptera abundance have been observed following burning (Morris 1975, Anderson et al. 1989). Like Orthoptera, many Hemiptera are reliant on herbaceous vegetation as a food source (Warren et al. 1987) and burning significantly alters availability of graminoids and forbs (Chapter 2). Unlike Orthoptera, the majority of Hemiptera occupy aerial habitats (Warren et al. 1987) and, as a result, the loss of litter following burning may not be such a concern. In other studies, Hemiptera abundance has been found to be greater on burned areas in the short and/or intermediate term (Rice 1932, Cancelado and Yonke 1970, Hurst 1971). In the case of Hymenoptera, declines in abundance have been observed in the short-term (Bulen and Barrett 1971). However, increases in predaceous species often occur, presumably because of greater numbers and vulnerability of prey on burned areas (Van Amburg et al. 1981). Other studies indicate an increase in Hymenoptera abundance on burned areas a few months following treatment (Hurst 1971, Nagel 1973). Araneida are only able to survive the combustion phase of a burn by seeking refuge in the soil or under nonflammable debris (Warren et al. 1987). Immediately following a burn, their

survival is likely to be largely dependent on prey density and diversity (Warren et al. 1987), which the results of this study suggest may be reduced.

Roller chopping/burning had a short-term effect on Blattodea familial richness and abundance, which lasted for 1 year post-treatment. As mentioned previously Blattodea occur largely in the litter layer (Warren et al. 1987), the cover and depth of which is severely reduced after roller chopping/burning (Chapter 2). In addition, the passing of the roller choppers blades may cause soil disturbance that directly and indirectly leads to reductions in the abundance of members of this order.

Habitat characteristics that were most often identified as positively related to total arthropod familial richness and abundance, Araneida richness and abundance, and Orthoptera abundance were related to graminoid and forb cover and height, and litter cover and depth. All of these characteristics are lower on dormant and growing season burn and roller chop than control sites, some for 2 years post-treatment (Chapter 2). Therefore, the application of these treatments should be applied with caution in areas where arthropod conservation and management are a priority.

Managment Implications

Studies examining the response of arthropods to prescribed burning are often contradictory. Generally, total arthropod abundance and richness in Florida flatwoods are lower following prescribed burning and roller chopping treatments, at least in the short-term. However, greater variability in response to these treatments is observed when individual orders are examined. Certainly, prescribed burning and roller chopping may be viable management tools for the control of certain arthropod pests. However, further investigation of pest phenology and habits is needed to ensure treatments applied are the most appropriate for control. When maintenance of overall arthropod

biodiversity is a priority, care should be taken regarding the use of prescribed burning and roller chopping treatments over large areas. Consideration should also be given to the season of application, with growing season burning limited to minimize negative effects on certain orders and families. Until further research is conducted, arthropod biodiversity may be best promoted by applying a combination of the treatments examined in a mosaic across the landscape. In addition, it may be necessary to implement techniques that help protect populations of immobile arthropods. Leaving untreated areas adjacent to treated areas to serve as refugia may be beneficial to these populations. Ultimately, the application of prescribed burning and roller chopping has the potential to harm beneficial arthropods, which may include pollinators, scavengers, and predators of parasites. Future research should take into account the relative value of all arthropods and how they are affected by prescribed burning and roller chopping activities to allow treatment application to be appropriately tailored to benefit desired orders or families.

Table 4-1. Arthropod abundance (no of individuals) within orders and families collected on prescribed burned, roller chopped, and control sites in Florida flatwoods, 2007–2008.

Order ^a	Family ^b
Araneida (5298)	Agelenidae (983), Araneidae (513), Corinnidae (5), Ctenidae (5), Ctenizidae (2), Clubionidae (5), Cyrtaucheniidae (1), Deinopidae (7), Dictynidae (2), Filistatidae (18), Gnaphosidae (815), Lycosidae (62), Nephilidae (287), Oonopidae (14), Oxyopidae (316), Pisauridae (6), Pholcidae (13), Salticidae (1853), Segestriidae (10), Tetragnathidae (14), Theraphosidae (4), Theridiidae (80), Thomisidae (220), Unknown (63)
Hemiptera (2511)	Acanaloniidae (5), Alydidae (6), Anthocoridae (7), Aphididae (63), Aradidae (3), Berytidae (64), Blissidae (1), Cercopidae (26), Cicadellidae (1385), Cicadidae (11), Cixiidae (15), Coreidae (8), Cydnidae (3), Delphacidae (20), Derbidae (172), Dictyopharidae (7), Flatidae (51), Hydrometridae (2), Issidae (171), Lygaeidae (18), Membracidae (77), Miridae (147), Nabidae (28), Pachygronthidae (72), Pachytroctidae (2), Pentatomidae (33), Pyrrhocoridae (5), Reduviidae (44), Rhopalidae (7), Scolytidae (1), Scutelleridae (6), Tinigidae (26), Tropiduchidae (1), Unknown (24)
Orthoptera (2465)	Acrididae (749), Gryllidae (1415), Rhaphidophoridae (10), Tetrigidae (98), Tettigonidae (190), Unknown (3)
Hymenoptera (1528)	Apidae (34), Braconidae (2), Chalcididae (1), Cimbicidae (1), Crabronidae (2), Eulophidae (1), Formicidae (1038), Halictidae (27), Ichneumonidae (75), Megachilidae (2), Mutillidae (3), Pteromalidae (1), Sphecidae (2), Vespidae (8), Unknown (60)
Diptera (1027)	Bibionidae (138), Bombyliidae (1), Cecidomyiidae (4), Chamaemyiidae (2), Chironomidae (7), Chloropidae (12), Curtonotidae (1), Culicidae (150), Dolichopodidae (176), Drosophilidae (183), Empididae (1), Hippoboscidae (2), Lauxaniidae (1), Muscidae (17), Mycetophilidae (2), Otitidae (2), Phoridae (202), Pipunculidae (2), Platypezidae (1), Platystomatidae (6), Sciomyzidae (6), Simuliidae (1), Stratiomyiidae (2), Syrphidae (1), Tabanidae (1), Tachinidae (11), Tephritidae (4), Tipulidae (5), Ulidiidae (6), Unknown (92)

Table 4-1. Continued

Order ^a	Family ^b
Coleoptera (942)	Anthicidae (12), Cantharidae (11), Carabidae (9), Cerambycidae (2), Chrysomelidae (573), Cleridae (43), Cucjidae (11), Curculionidae (106), Coccinellidae (61), Elateridae (13), Endomychidae (1), Erotylidae (11), Eucnemidae (3), Euglenidae (1), Gyrinidae (1), Lampyridae (17), Latridiidae (7), Lyctidae (1), Meloidae (1), Mordellidae (4), Nitidulidae (3), Pselaphidae (1), Scarabaeidae (9), Scolytidae (7), Silphidae (7), Staphylinidae (2), Tenebrionidae (6), Unknown (17)
Blattodea (552)	Blattellidae (511), Blattidae (32). Unknown (9)
Psocoptera (156)	Pachytroctidae (143), Unknown (13)
Collembola (150)	Isotomidae (134) , Poduridae (10), Sminthuridae (2), Entomobryidae (4)
Phasmatodea (103)	Heteronemiidae (12), Phasmatidae (40), Pseudophasmatidae (38), Unknown (13)
Trichoptera (82)	Hydroptilidae (65), Unknown (17)
Acari (66)	Ixodidae (2) Trombiculidae (64)
Scorpiones (58)	Buthidae (1), Unknown (57)
Lepidoptera (53)	Geometridae (23), Hepialidae (1), Nymphalidae (1), Unknown (28)
Parasitengona (47)	Trombiculidae (47)
Mantodea (35)	Mantidae (23), Photinae (12)
Thysanoptera (25)	Thripidae 24, Unknown (1)
Neuroptera (19)	Ascalaphidae (2), Chrysopidae (14), Corydalidae (1), Megcorydalidae (1), Unknown (3)
Pseudoscorpionida (17)	Chernetidae (17)
Isoptera (15)	Rhinotermitidae (15)
Opiliones (5)	Stylocellidae (1), Unknown (4)

Table 4-1. Continued

Order ^a	Family ^b
Spirobolida (3)	Spirobolidae (1), Unknown (2)
Diplura (3)	Campodeidae (1), Unknown (2)
Embiidina (1)	Teratembiiidae (1)

^a Abundance by order presented in parentheses.

^b Abundance by family presented in parentheses.

Table 4-2. Effects of treatment on arthropod familial richness and abundance in Florida flatwoods, 2007–2008.

Arthropod Richness and Abundance by Treatment ^a	Treatment ($\bar{x} \pm SE$)		<i>P</i>
	Control	Treated	
Dormant season roller chop			
Richness (no. of families)			
Total	12.3 ± 0.6	10.4 ± 0.7	0.017
Araneida	3.7 ± 0.2	3.0 ± 0.2	0.020
Abundance (no. of individuals)			
Total	37.0 ± 3.0	29.5 ± 3.1	0.032
Orthoptera	8.8 ± 0.7	7.4 ± 1.0	0.085
Hemiptera	6.3 ± 0.7	4.9 ± 0.8	0.042
Araneida	12.4 ± 1.4	8.0 ± 1.0	0.007
Growing season roller chop			
Richness (no. of families)			
Blattodea	0.7 ± 0.1	0.3 ± 0.1	0.010
Abundance (no. of individuals)			
Diptera	1.9 ± 0.1	1.3 ± 0.6	0.082
Blattodea	2.6 ± 0.7	1.3 ± 0.4	0.030
Growing season burn			
Richness (no. of families)			
Hemiptera	2.2 ± 0.2	1.7 ± 0.2	0.021
Abundance (no. of individuals)			
Total	40.0 ± 4.2	24.0 ± 2.8	0.001
Hemiptera	6.3 ± 1.0	3.8 ± 0.7	0.002
Araneida	14.9 ± 1.7	7.5 ± 1.0	≤0.001
Roller chop/burn			
Abundance (no. of individuals)			
Diptera	3.7 ± 1.0	2.0 ± 0.7	0.043

^a Only arthropod families with richness or abundance significantly affected by treatment presented ($P \leq 0.1$).

Table 4-3. Effects of treatment * time interactions on arthropod familial richness and abundance in Florida flatwoods, 2007–2008.

Arthropod Richness and Abundance by Treatment ^a	Time ^b	Treatment ($\bar{x} \pm SE$) ^c		P
		Control	Treated	
Dormant season roller chop				
Richness (no. of families)				
Hemiptera	1	2.3 ± 0.3 _A	1.5 ± 0.2 _A	0.022
	2	2.1 ± 0.2 _A	2.3 ± 0.2 _A	
Diptera	1	1.1 ± 0.2 _A	0.4 ± 0.1 _A	0.022
	2	1.0 ± 0.2 _A	1.1 ± 0.2 _A	
Hymenoptera	1	1.2 ± 0.2 _A	0.6 ± 0.1 _A	0.007
	2	0.8 ± 0.2 _A	0.8 ± 0.1 _A	
Abundance (no. of individuals)				
Diptera	1	1.7 ± 0.4 _A	0.7 ± 0.3 _A	0.014
	2	1.9 ± 0.5 _A	1.9 ± 0.4 _A	
Hymenoptera	1	3.6 ± 0.9 _A	3.4 ± 1.3 _A	0.052
	2	1.8 ± 0.4 _A	2.4 ± 0.7 _A	
Growing season roller chop				
Richness (no. of families)				
Total	1	15.0 ± 1.2 _A	9.3 ± 1.1 _B	0.097
	2	11.6 ± 1.1 _A	11.0 ± 1.0 _A	
Orthoptera	1	2.3 ± 0.3 _A	1.7 ± 0.2 _A	0.042
	2	1.7 ± 0.2 _A	2.0 ± 0.3 _A	
Abundance (no. of individuals)				
Orthoptera	1	7.5 ± 1.7 _A	3.4 ± 0.7 _B	0.070
	2	4.7 ± 0.9 _A	5.1 ± 1.0 _A	
Dormant season burn				
Richness (no. of families)				
Total	1	13.8 ± 1.1 _A	7.0 ± 0.9 _B	≤0.001
	2	12.0 ± 1.1 _A	14.2 ± 1.0 _A	
Orthoptera	1	1.2 ± 0.2 _A	0.7 ± 0.1 _B	≤0.001
	2	1.5 ± 0.2 _A	1.9 ± 0.2 _A	
Hemiptera	1	2.6 ± 0.3 _A	1.0 ± 0.2 _B	0.003
	2	2.5 ± 0.3 _A	2.8 ± 0.3 _A	
Diptera	1	1.4 ± 0.2 _A	0.5 ± 0.1 _B	0.004
	2	1.3 ± 0.3 _A	1.5 ± 0.2 _A	
Hymenoptera	1	1.1 ± 0.1 _A	0.6 ± 0.1 _B	0.009
	2	0.8 ± 0.1 _A	1.0 ± 0.2 _A	
Abundance (no. of individuals)				
Total	1	51.4 ± 9.0 _A	18.3 ± 3.8 _B	≤0.001
	2	36.2 ± 5.6 _A	43.8 ± 5.2 _A	
Orthoptera	1	4.0 ± 0.7 _A	1.3 ± 0.4 _B	≤0.001
	2	4.2 ± 0.8 _A	6.0 ± 1.0 _A	
Hemiptera	1	8.1 ± 1.8 _A	2.1 ± 0.6 _B	0.001
	2	6.3 ± 1.1 _A	7.0 ± 1.2 _A	

Table 4.3. Continued

Arthropod Richness and Abundance by Treatment ^a	Time ^b	Treatment ($\bar{x} \pm SE$) ^c		<i>P</i>
		Control	Treated	
Diptera	1	3.7 ± 0.9 _A	1.1 ± 0.4 _A	0.008
	2	2.7 ± 0.8 _A	4.3 ± 1.1 _A	
Araneida	1	17.3 ± 2.8 _A	7.1 ± 1.6 _A	≤0.001
	2	14.7 ± 2.5 _A	17.8 ± 2.5 _A	
Hymenoptera	1	8.3 ± 3.1 _A	2.2 ± 0.7 _B	0.028
	2	2.4 ± 0.6 _A	3.3 ± 1.0 _A	
Growing season burn				
Richness (no. of families)				
Total	1	13.4 ± 0.6 _A	5.9 ± 1.1 _B	0.017
	2	11.2 ± 0.6 _A	10.1 ± 0.8 _A	
Araneida	1	4.6 ± 0.5 _A	2.0 ± 0.3 _B	0.018
	2	3.3 ± 0.3 _A	3.3 ± 0.3 _A	
Roller chop/burn				
Richness (no. of families)				
Orthoptera	1	2.0 ± 0.3 _A	1.4 ± 0.3 _A	0.027
	2	1.4 ± 0.2 _A	2.2 ± 0.2 _A	
Blattodea	1	0.9 ± 0.1 _A	0.1 ± 0.1 _B	0.001
	2	0.5 ± 0.1 _A	0.3 ± 0.1 _A	
Abundance (no. of individuals)				
Blattodea	1	2.4 ± 0.7 _A	0.1 ± 0.1 _B	≤0.001
	2	1.0 ± 0.3 _A	1.0 ± 0.6 _A	

^a Only arthropod families with richness or abundance significantly affected by a treatment * time interaction presented ($P \leq 0.1$).

^b Time since treatment application (years).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 4-4. Effects of treatment * season interactions on arthropod familial richness and abundance in Florida flatwoods, 2007–2008.

Arthropod Richness and Abundance by Treatment ^a	Season	Treatment ($\bar{x} \pm SE$) ^b		<i>P</i>
		Control	Treated	
Dormant season roller chop				
Richness (no. of families)				
Hemiptera	Winter	2.1 ± 0.2 _A	1.3 ± 0.2 _B	0.058
	Spring	2.5 ± 0.3 _A	2.3 ± 0.3 _A	
	Summer	2.0 ± 0.3 _A	2.2 ± 0.3 _A	
Diptera	Winter	1.5 ± 0.3 _A	0.5 ± 0.2 _A	0.026
	Spring	0.7 ± 0.2 _A	0.8 ± 0.2 _B	
	Summer	1.1 ± 0.3 _A	1.0 ± 0.2 _B	
Hymenoptera	Winter	0.5 ± 0.1 _A	0.5 ± 0.1 _A	0.005
	Spring	1.5 ± 0.2 _A	0.5 ± 0.2 _B	
	Summer	1.1 ± 0.2 _A	1.1 ± 0.1 _A	
Abundance (no. of individuals)				
Diptera	Winter	2.5 ± 0.6 _A	0.6 ± 0.3 _A	0.009
	Spring	1.1 ± 0.5 _A	1.0 ± 0.2 _B	
	Summer	1.8 ± 0.5 _A	2.3 ± 0.7 _B	
Growing season burn				
Richness (no. of families)				
Blattodea	Winter	0.9 ± 0.1 _A	0.1 ± 0.1 _B	0.062
	Spring	0.1 ± 0.1 _A	0.1 ± 0.1 _A	
	Summer	0.6 ± 0.1 _A	0.0 ± 0.0 _A	
Abundance (no. of individuals)				
Blattodea	Winter	2.7 ± 0.7 _A	0.1 ± 0.1 _B	0.052
	Spring	0.1 ± 0.1 _A	0.2 ± 0.2 _A	
	Summer	2.1 ± 0.7 _A	0.0 ± 0.0 _A	
Roller chop/burn				
Richness (no. of families)				
Coleoptera	Winter	0.5 ± 0.2 _A	1.2 ± 0.3 _A	0.016
	Spring	1.7 ± 0.3 _A	0.9 ± 0.2 _B	
	Summer	0.9 ± 0.2 _A	1.2 ± 0.2 _A	
Abundance (no. of individuals)				
Coleoptera	Winter	0.7 ± 0.4 _A	3.1 ± 1.1 _A	0.003
	Spring	2.8 ± 0.5 _A	1.6 ± 0.6 _B	
	Summer	1.8 ± 0.5 _A	3.7 ± 1.3 _B	

^a Only arthropod families with richness or abundance significantly affected by a treatment * season interaction presented ($P \leq 0.1$).

^b Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 4-5. Effects of treatment * grazing interactions on arthropod familial richness and abundance in Florida flatwoods, 2007–2008.

Arthropod Richness and Abundance by Treatment ^a	Grazing	Treatment ($\bar{x} \pm SE$) ^b		<i>P</i>
		Control	Treated	
Dormant season roller chop				
Richness (no. of families)				
Blattodea	Nongrazed	0.7 ± 0.1 _A	0.5 ± 0.1 _A	0.034
	Grazed	0.3 ± 0.1 _A	0.5 ± 0.1 _A	
Abundance (no. of individuals)				
Diptera	Nongrazed	1.2 ± 0.3 _A	1.4 ± 0.4 _B	0.069
	Grazed	2.4 ± 0.5 _A	1.3 ± 0.4 _B	
Blattodea	Nongrazed	2.1 ± 0.5 _A	1.5 ± 0.4 _A	0.033
	Grazed	0.4 ± 0.1 _A	1.4 ± 0.4 _A	
Growing season roller chop				
Richness (no. of families)				
Hemiptera	Nongrazed	2.2 ± 0.4 _A	2.3 ± 0.4 _A	0.017
	Grazed	2.8 ± 0.4 _A	1.9 ± 0.6 _A	
Dormant season burn				
Richness (no. of families)				
Total	Nongrazed	9.7 ± 1.4 _A	10.0 ± 1.8 _A	≤0.001
	Grazed	13.7 ± 0.8 _A	10.5 ± 0.9 _A	
Hymenoptera	Nongrazed	0.8 ± 0.1 _A	1.0 ± 0.2 _A	0.094
	Grazed	1.0 ± 0.1 _A	0.7 ± 0.1 _A	
Abundance (no. of individuals)				
Coleoptera	Nongrazed	1.6 ± 1.1 _A	2.7 ± 0.7 _B	0.060
	Grazed	3.8 ± 0.9 _A	2.6 ± 0.4 _A	
Blattodea	Nongrazed	0.4 ± 0.2 _A	0.9 ± 0.3 _B	0.040
	Grazed	2.1 ± 0.5 _A	0.7 ± 0.3 _A	
Growing season burn				
Abundance (no. of individuals)				
Orthoptera	Nongrazed	3.6 ± 0.9 _A	6.2 ± 1.2 _A	0.080
	Grazed	3.9 ± 1.1 _A	4.9 ± 1.6 _A	

^a Only arthropod families with richness or abundance significantly affected by a treatment * grazing interaction presented ($P \leq 0.1$).

^b Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 4-6. Comparison of the effects of treatment * time interactions on arthropod familial richness and abundance in Florida flatwoods, 2007–2008.

Arthropod Richness and Abundance ^a	Time ^b	Treatment ($\bar{x} \pm SE$) ^c						P
		Control	Dormant Burn	Growing burn	Dormant Roller Chop	Growing Roller chop	Roller Chop/Burn	
Richness (no. of individuals)								
Total	1	13.8 ± 0.5 _A	7.1 ± 0.9 _{BD}	6.1 ± 1.2 _B	9.4 ± 0.9 _{AC}	9.3 ± 1.1 _{BCD}	9.8 ± 1.9 _{AD}	≤0.001
	2	11.7 ± 0.4 _A	14.3 ± 0.9 _A	10.3 ± 0.8 _A	11.4 ± 0.9 _A	11.1 ± 1.0 _A	13.0 ± 1.1 _A	
Orthoptera	1	1.9 ± 0.1 _A	0.7 ± 0.1 _B	0.6 ± 0.3 _B	1.9 ± 0.2 _A	1.7 ± 0.2 _B	1.4 ± 0.3 _A	≤0.001
	2	1.6 ± 0.1 _A	1.9 ± 0.2 _A	1.7 ± 0.2 _A	2.1 ± 0.2 _A	2.0 ± 0.3 _A	2.2 ± 0.2 _A	
Diptera	1	1.2 ± 0.1 _A	0.5 ± 0.1 _A	0.3 ± 0.2 _A	0.4 ± 0.1 _A	0.8 ± 0.5 _A	0.9 ± 0.3 _A	0.001
	2	1.1 ± 0.1 _A	1.5 ± 0.2 _A	1.2 ± 0.2 _A	1.1 ± 0.2 _A	0.7 ± 0.2 _A	1.0 ± 0.2 _A	
Araneida	1	4.1 ± 0.2 _A	2.5 ± 0.3 _A	2.0 ± 0.3 _A	2.6 ± 0.3 _A	2.9 ± 0.4 _A	2.6 ± 0.6 _A	≤0.001
	2	3.6 ± 0.1 _A	4.2 ± 0.3 _A	3.3 ± 0.3 _A	3.3 ± 0.2 _A	3.1 ± 0.2 _A	3.9 ± 0.4 _A	
Hymenoptera	1	1.0 ± 0.1 _A	0.6 ± 0.1 _B	0.8 ± 0.2 _A	0.6 ± 0.2 _A	0.7 ± 0.1 _{AB}	0.7 ± 0.1 _A	0.006
	2	0.8 ± 0.1 _A	1.0 ± 0.2 _B	0.1 ± 0.1 _A	0.8 ± 0.1 _A	0.7 ± 0.2 _A	1.0 ± 0.2 _{AB}	
Blattodea	1	0.6 ± 0.1 _A	0.2 ± 0.1 _B	0.0 ± 0.0 _B	0.6 ± 0.1 _A	0.3 ± 0.2 _{AB}	0.1 ± 0.1 _B	0.040
	2	0.5 ± 0.0 _A	0.5 ± 0.1 _A	0.1 ± 0.1 _A	0.4 ± 0.1 _A	0.3 ± 0.1 _A	0.3 ± 0.1 _A	
Abundance (no. of individuals)								
Total	1	48.3 ± 4.0 _A	18.3 ± 3.8 _B	23.4 ± 5.7 _B	26.3 ± 3.7 _{AB}	40.6 ± 16.6 _{AB}	31.6 ± 9.4 _{AB}	≤0.001
	2	34.5 ± 2.1 _A	43.8 ± 5.2 _A	24.2 ± 3.3 _A	32.1 ± 4.8 _A	24.8 ± 2.8 _A	42.3 ± 7.3 _A	
Orthoptera	1	6.5 ± 0.5 _A	1.3 ± 0.4 _B	1.4 ± 0.9 _B	7.0 ± 1.2 _A	3.4 ± 0.7 _{BC}	4.0 ± 1.1 _{AC}	≤0.001
	2	5.3 ± 0.4 _A	6.0 ± 1.0 _A	4.8 ± 0.9 _A	7.9 ± 1.4 _A	5.1 ± 1.0 _A	11.3 ± 2.0 _A	
Hemiptera	1	6.7 ± 0.8 _A	2.1 ± 0.6 _{BC}	1.7 ± 0.7 _B	3.3 ± 1.0 _{AB}	2.4 ± 1.0 _{AB}	6.1 ± 2.7 _{AC}	0.001
	2	6.4 ± 0.5 _A	7.0 ± 1.2 _A	4.6 ± 0.9 _A	6.3 ± 1.1 _A	7.0 ± 1.4 _A	7.8 ± 1.6 _A	
Diptera	1	2.5 ± 0.4 _A	1.1 ± 0.4 _A	0.4 ± 0.2 _A	0.7 ± 0.3 _A	2.2 ± 1.9 _A	3.3 ± 1.7 _A	0.010
	2	2.9 ± 0.4 _A	4.3 ± 1.1 _A	2.6 ± 0.8 _A	1.9 ± 0.4 _A	1.0 ± 0.3 _A	2.1 ± 0.6 _A	
Araneida	1	17.6 ± 1.6 _A	7.1 ± 1.7 _{AB}	5.3 ± 1.2 _B	6.4 ± 1.0 _B	7.4 ± 1.9 _{AB}	10.5 ± 3.6 _A	≤0.001
	2	12.9 ± 1.0 _A	17.8 ± 2.5 _A	8.4 ± 1.2 _A	9.3 ± 1.7 _A	6.4 ± 0.8 _A	13.6 ± 2.7 _A	
Hymenoptera	1	5.8 ± 1.2 _A	2.2 ± 0.7 _B	12.4 ± 4.0 _{AC}	3.5 ± 1.3 _{AC}	4.6 ± 1.7 _{AC}	3.1 ± 0.8 _C	0.041
	2	2.9 ± 0.6 _A	3.3 ± 1.0 _A	1.9 ± 0.7 _A	2.4 ± 0.8 _A	1.4 ± 0.4 _A	2.2 ± 0.7 _A	
Blattodea	1	2.4 ± 0.4 _A	0.2 ± 0.1 _B	0.0 ± 0.0 _B	1.9 ± 0.5 _A	1.2 ± 0.8 _{AB}	0.1 ± 0.1 _B	0.003
	2	1.3 ± 0.2 _A	1.3 ± 0.5 _A	0.1 ± 0.1 _A	1.0 ± 0.4 _A	1.3 ± 0.5 _B	1.0 ± 0.6 _A	

^a Only arthropod families with richness or abundance significantly affected by a treatment * time interaction ($P \leq 0.1$).

^b Time since treatment application (years).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 4-7. Comparison of the effects treatment * season interactions on arthropod familial richness and abundance in Florida flatwoods, 2007–2008.

Arthropod Richness and Abundance ^a	Season	Treatment ($\bar{x} \pm SE$) ^b					<i>P</i>	
		Control	Dormant Burn	Growing Burn	Dormant Roller Chop	Growing Roller Chop		Roller Chop/Burn
Richness (no. of families)								
Orthoptera	Winter	1.5 ± 0.1 _A	1.1 ± 0.2 _A	1.8 ± 0.4 _A	1.7 ± 0.2 _A	2.0 ± 0.4 _A	2.7 ± 0.3 _B	0.078
	Spring	1.5 ± 0.1 _A	1.3 ± 0.3 _{AB}	1.8 ± 0.3 _A	1.9 ± 0.2 _A	1.3 ± 0.4 _B	1.4 ± 0.3 _{AB}	
	Summer	2.1 ± 0.1 _A	1.5 ± 0.2 _A	1.0 ± 0.3 _A	2.3 ± 0.2 _A	2.2 ± 0.3 _A	2.0 ± 0.3 _A	
Hymenoptera	Winter	0.7 ± 0.1 _{ABC}	0.6 ± 0.1 _A	0.7 ± 0.2 _{BC}	0.5 ± 0.1 _A	1.2 ± 0.3 _C	1.2 ± 0.3 _C	0.001
	Spring	1.1 ± 0.1 _A	1.0 ± 0.2 _B	0.7 ± 0.3 _A	0.5 ± 0.2 _B	0.4 ± 0.2 _A	0.6 ± 0.2 _B	
	Summer	0.9 ± 0.1 _{AB}	0.7 ± 0.1 _{AB}	0.6 ± 0.1 _A	1.1 ± 0.1 _B	0.8 ± 0.1 _{AB}	0.9 ± 0.1 _{AB}	
Abundance (no. of individuals)								
Coleoptera	Winter	1.0 ± 0.2 _A	1.5 ± 0.3 _A	0.6 ± 0.2 _A	0.7 ± 0.3 _A	0.5 ± 0.3 _A	3.1 ± 1.1 _B	0.086
	Spring	1.8 ± 0.2 _A	2.4 ± 0.5 _A	0.7 ± 0.3 _A	1.4 ± 0.6 _A	1.4 ± 0.8 _A	1.6 ± 0.6 _A	
	Summer	3.3 ± 0.6 _{AB}	4.0 ± 0.8 _{AB}	1.3 ± 0.3 _A	4.3 ± 1.4 _{AB}	2.6 ± 0.7 _B	3.7 ± 1.3 _{AB}	
Araneida	Winter	14.8 ± 1.6 _A	11.9 ± 3.0 _A	8.1 ± 2.0 _A	7.9 ± 1.8 _A	7.8 ± 1.2 _A	17.9 ± 6.7 _A	0.070
	Spring	12.0 ± 1.1 _A	7.7 ± 1.9 _{AB}	1.6 ± 1.3 _B	7.9 ± 2.1 _{AB}	4.2 ± 0.8 _B	5.2 ± 1.7 _B	
	Summer	17.0 ± 1.6 _A	16.9 ± 3.0 _A	5.8 ± 0.9 _A	8.2 ± 1.6 _A	8.1 ± 1.3 _A	15.6 ± 3.0 _A	

^a Only arthropod families with richness or abundance significantly affected by a treatment * season interaction presented ($P \leq 0.1$).

^b Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

CHAPTER 5
EFFECTS OF PRESCRIBED BURNING AND ROLLER CHOPPING ON AVIAN
COMMUNITIES AND THEIR HABITAT ASSOCIATIONS IN FLATWOODS

Introduction

Peninsular Florida's rangeland habitats, including its pine flatwoods, provide habitat for a variety of resident and migratory bird species (United States Fish and Wildlife Service [USFWS] 1999, Alsop 2002, Engstrom et al. 2005, Florida Fish and Wildlife Conservation Commission [FWC] 2005). Many of these avian species, some of which are federally and/or state listed as endangered or threatened, have decreasing populations and are considered by FWC to be of conservation concern (Engstrom 2005, FWC 2005).

Declines in bird populations, particularly those of obligate and facultative species associated with open savanna and grassland habitats, have been reported across the eastern United States and attributed to a loss of suitable habitat (Askins 1993, Murphy 2003, Brennan and Kuvlesky 2005). Certainly, the pine flatwoods, which provide habitat for many of Florida's resident and migratory savanna and grassland affiliated species, are in poor condition and declining in quality and quantity (Davis 1967, Abrahamson and Hartnett 1990, Kautz 1993, Cox et al. 1997, USFWS 1999, FWC 2005). As a result, they are listed by FWC (2005) as terrestrial habitats with a high relative threat status. Changes in land management techniques (e.g., alteration of fire regimes and increases in grazing pressure) are thought to be contributing to the degradation of these areas (Abrahamson and Hartnett 1990, FWC 2005, Gordon et al. 2005). Historically, pine flatwoods, were subject to high frequency, low intensity, lightning ignited fires during the thunderstorm season (May–July; Komarek 1968, Christensen 1981, Abrahamson and Hartnett 1990, Pyne et al. 1996). Frequent fires are considered essential to maintain

the structure and diverse herbaceous groundcover of many southeastern rangeland systems, such as pine flatwoods (Christensen 1981, Abrahamson and Hartnett 1990, Platt 1998). Fire exclusion, reductions in fire frequency, and/or a shift in fire season, commonly a result of human intervention, can result in excessive shrub growth and declines in the species rich herbaceous ground layer of these areas (Wade et al. 1980, Platt et al. 1988, Fitzgerald and Tanner 1992, Robbins and Myers 1992, Watts and Tanner 2003, Watts and Tanner 2006). On many private pine flatwoods in Florida, the majority of which are used for livestock production, changes in land management have resulted in alterations to the natural fire regime. This has frequently permitted the proliferation of shrubby vegetation such as saw palmetto (*Serenoa repens* [Bartram] Small), gallberry (*Ilex glabra* [L.] A. Gray), and wax myrtle (*Morella cerifera* [L.] Small) and caused reductions in grass and forb species. Such changes threaten the integrity of remaining pine flatwoods habitats, potentially reducing their suitability for a variety of bird species, many of which are considered of conservation priority (Engstrom 2005, FWC 2005).

In Florida, FWC and the United States Department of Agriculture (USDA) are utilizing assistance-based programs such as the Farm Bill's Environmental Quality Incentives Program and Wildlife Habitat Incentives Program, to encourage private landowners to better manage remaining areas of pine flatwoods. Management activities currently being promoted under these programs include the use of prescribed fire and roller chopping during dormant (November–March) and growing (April–October) seasons. These practices, depending on season of application, have been shown to improve the quality of southeastern rangelands by reducing shrub cover and promoting

the growth and flowering of herbaceous vegetation (Chapter 2; Kalmbacher and Martin 1984, Platt et al. 1988, Tanner et al. 1988, Robbins and Myers 1992, Watts and Tanner 2003, Watts and Tanner 2006).

The impacts prescribed-burning and roller chopping have on Florida's pine flatwoods-associated avian communities are largely unknown. However, these practices are likely to lead to alterations in avian community composition through changes in vegetation composition and structure and available habitat (Johnston and Odum 1956, Anderson 1980, Norris et al. 2003, Venier and Pearce 2005). Certainly, in other North American grassland systems, disturbances such as prescribed burning have influenced bird abundance and species richness through changes to the plant community (Huber 1984, Madden et al. 1999). In Florida, a study examining the effect of management activities on bird communities found the application of prescribed burning and roller chopping to a shrub-dominated, former dry prairie site in south Florida had an acute effect on bird abundance and species composition, with notable declines on roller-chopped sites (Fitzgerald 1990, Fitzgerald and Tanner 1995). In light of current efforts to promote the use of prescribed burning and roller chopping on rangelands through the use of assistance-based programs, a clearer understanding of the effects these activities have on avian communities is needed (Berkland et al. 2005, FWC 2005, Gray et al. 2005). Such research will establish whether these management practices are beneficial to the numerous bird species of greatest conservation need commonly associated with pine flatwoods habitats (FWC 2005), and determine whether their use to maintain and restore rangeland vegetation is also an effective avian conservation strategy.

The objectives of my study were to 1) compare avian species richness and abundance in treated (management activities implemented) and untreated (no management activities implemented) pine flatwoods during winter, migratory, and breeding seasons and 2) examine the relative importance of local pine flatwoods habitat characteristics (e.g., shrub cover, forb, cover, graminoid height, arthropod abundance) in determining avian species richness and abundance.

Methods

Study Sites

I conducted research on 50 privately- and publically-owned pine flatwoods sites that were being prescribed burned and roller chopped by local landowners and land managers using a variety of individual protocols. These sites had varying management and grazing histories and were located across 6 counties (Desoto, Highlands, Lee, Manatee, Osceola, and Sarasota) in central and south Florida. When grazed, both the treatment and paired control study sites were subject to similar grazing pressures at similar times.

Florida's pine flatwoods are characterized as having a pure or combined overstory stand of scattered longleaf (*Pinus palustris* Mill.) and slash (*P. elliotti* Engelm.) pine and, when the shrub layer is relatively open, an often diverse herbaceous layer. This herbaceous layer contains a wide variety of grasses (e.g., *Agrostis*, *Andropogon*, *Aristida*, *Eragrostis*, *Panicum*, and *Paspalum* spp.). Common forbs include legumes (e.g., *Cassia*, *Crotalaria*, *Galactia*, *Tephrosia* spp.), milkweeds (*Asclepias* spp.), milkworts (*Polygala* spp.), and a wide variety of composites (e.g., *Aster*, *Chrysopsis*, *Eupatorium*, *Liatris*, and *Solidago* spp.). Understory and shrub species include saw palmetto, wax myrtle (*Morella cerifera* [L.] Small), gallberry (*Ilex glabra* [Pursh] Chapm.),

fetterbush (*Lyonia lucida* [Lam.] K. Koch), staggerbush (*Lyonia fruticosa* [Michx.], G. S. Torr), dwarf huckleberry (*Gaylussacia dumosa* [Andrews] Torr. & A. Gray), dwarf live oak (*Quercus mimima* [Sarg.] Small), and tarflower (*Bejaria racemosa* Vent; Abrahamson and Hartnett 1990, USFWS 1999.).

Treatment Types

One of 5 treatment types was applied to treated sites: dormant season (November–March) burn, growing season (April–October) burn, dormant season roller chop, growing season roller chop, or a roller chop/burn combination treatment. The roller chop/burn combination treatment (hereafter referred to as roller chop/burn) involved roller chopping in the dormant season followed by burning within 6 months. I established a total of 11 dormant season burn, 9 growing season burn, 9 dormant season roller chop, 12 growing season roller chop, and 9 roller chop/burn and control pairs.

Bird Surveys

I assessed the effects of treatment (prescribed burning, roller chopping, and roller chopping/burning) on avian richness and abundance using a paired sampling approach. Richness and abundance were compared between sampling points randomly located in paired treated (e.g., dormant season burned) and untreated (control) flatwoods sites. Paired treatment and control sampling points were of similar current and past management (e.g., grazing), surrounding land use, plant community (e.g., overstory cover), and soil conditions, being located in the same pasture or management unit. I established 1 randomly selected sampling point within each treatment and control site. To minimize edge effects, I rejected and randomly relocated sampling points that fell

within 50 m of the edge of a treatment or control site. Sites within which treatment and control sampling points were located ranged from 2–20 ha.

I surveyed the avian community at each sampling point 3 times each year, during each of 2 years (2007 and 2008) following treatment. Sampling periods corresponded to presumed seasonal differences in avian habitat utilization: wintering (January–March) and breeding (June–September) seasons and spring migration (April–May). I conducted bird surveys at each sampling point using unlimited radius point count methods (Gibbons et al. 1996, Bibby et al. 2000). After arriving at each point count location, I waited 3 minutes before commencing sampling. This provided time for birds to settle following my arrival. Point counts were conducted for 5 minutes, during which all birds detected to an unlimited distance were recorded (Gibbons et al. 1996, Bibby et al. 2000). Study sites were surveyed within 4 hours of sunrise, on mornings with little rain, no wind, and no fog, and with paired sampling points always being counted on the same day (Gibbons et al. 1996, Bibby et al. 2000). During each sampling period, the paired point sampled first was alternated. I used careful observation, including recording the approximate position of detected birds and flyovers to reduce the likelihood of double counting (Gibbons et al. 1996, Gregory et al. 2004).

I divided counts of avian abundance and richness into 4 categories and 7 guilds based on residency and breeding status. Depending on residency status, counts were assigned to either a permanent resident or migrant category. The migrant category was divided into 2 guilds, short-distance migrant, and neo-tropical migrant. Based on their breeding status in Florida, birds were also assigned to 1 of 2 categories, non-breeding (overwintering) or breeding. The breeding category was divided into 5 breeding habitat

guilds: woodland, urban, successional-scrub, wetland, and grassland (Table 5-1; (Peterjohn and Sauer 1993)).

Habitat Sampling

I conducted habitat sampling once in winter (January–March), spring (April–May), and summer (June–September) during each of 2 years (2007–2008) following treatment at the same points used for avian surveys. At each sampling point, plant community composition and structure, litter and soil variables, and vertical obstruction were examined in several strata (i.e., ground, herbaceous, shrub, understory, and overstory levels), using a 0.03-ha nested circular plot design similar to that described by Dueser and Shugart (1978) and Higgins et al. (2005).

Ground layer. I assessed litter cover (%; ocular estimate) within 4 1-m² sub-sample plots, 1 randomly located in each quadrant of the 0.03-ha circular plots, along with soil density (g/cm³), moisture (%), and pH. Litter cover was recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Donhaue et al. 1971, Hays et al. 1981, Higgins et al. 2005). I measured soil density as the dry weight density (g/cm³) of a 5-cm diameter, 10-cm deep soil core sample after oven drying at 45°C for 48 hours. Soil pH and moisture were measured using a Kelway soil tester (Rodewald and Yahner 2001).

Herbaceous layer. I determined species richness, cover (%; ocular estimate), and maximum height (cm) of forbs and graminoids within the 1-m² subsample plots. Forb and graminoid cover were recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Hays et al. 1981, Krebs 1999, Higgins et al. 2005).

Vertical Obstruction. Vertical obstruction (%) from 0–2 m above ground was measured using a cover pole (Griffin and Youtie 1988) centered on the 0.03-ha circular plot at a distance of 5 and 10 m.

Shrub layer. I counted and measured the height of all shrubs (woody vegetation <2.0 m in height) in 2 perpendicular 20-m² quadrats centered on the 0.03-ha plot to estimate species richness (no. of species), density (no./m²), and maximum height (cm) for individual species and all combined (Hays et al. 1981, Krebs 1999, Higgins et al. 2005). Shrub cover (%) was assessed along 2 perpendicular 20-m transects centered on the 0.03-ha circular plot using the line intercept method (Hays et al. 1981, Higgins et al. 2005).

Understory. I counted, identified the species, and measured the diameter at breast height (dbh) of all understory (woody vegetation <7.5 cm dbh, ≥2.0 m in height) plants within the 0.03-ha circular plots to estimate species richness, density (no./ha), and basal area (cm²/ha) for individual species and all combined (Krebs 1999). Understory canopy cover (%) was estimated from 41 evenly spaced, vertical ocular tube sightings taken at a height of 0.75 m along 2 perpendicular 20-m transects centered on the 0.03-ha circular plot (James and Shugart 1971).

Overstory. All overstory (woody vegetation ≥7.5 cm dbh) plants were also counted, species identified, and dbh measured within the 0.03-ha circular plot to estimate species richness, density (no./ha), and basal area (cm²/ha) for individual species and all combined (Krebs 1999). I estimated overstory canopy cover (%) from 41 evenly spaced, vertical ocular tube sightings taken at a height of 1.5 m along 2

perpendicular 20-m transects centered on the 0.03-ha circular plot (James and Shugart 1971).

Arthropods. I collected arthropods occupying vegetation less than 30 cm above the ground from within 4 1-m² sub-sample plots, randomly located in each quadrant of a 0.03-ha nested circular plot centered on the sampling point (Dueser and Shugart 1978, Higgins et al. 2005). Arthropods were sampled using a suction sampler (Wright and Stewart 1993, Ausden 1996). Within each 1-m² plot, the suction sampler was turned on and systematically moved around the sub-sample area, no more than 30 cm above the ground, for a 3-minute period collecting arthropods. Suction sampling was not conducted if vegetation was damp or had been flattened by wind, rain, or trampling (Ausden 1996). I separated arthropods collected in each suction sub-sample from coarse vegetation and combined them in a vial containing a preservation agent of 75% ethanol and 25% distilled water (Schauff 1986).

I collected sub-samples of mobile arthropods and arthropods occupying vegetation more than 30 cm above the ground along 2 perpendicular 20-m transects centered on the sampling point (Dueser and Shugart 1978, Higgins et al. 2005). Arthropods were sampled using a sweep net (Ausden 1996). I made 50 sweeps (1 sweep comprising a forward and backward stroke of the sweep net) along each of the 20 m transects, ensuring the sweep net did not pass within 30 cm of the ground (Schauff 1986). I combined arthropods collected in each sweep net sub-sample and preserved as described for those collected using suction sampling. In the laboratory, I identified arthropods contained in each suction and sweep net sample to the family level using a

microscope and appropriate identification keys (Triplehorn and Johnson 2005, Ubick et al. 2005). Suction and sweep net samples were combined prior to analyses.

I recorded the presence of livestock on study sites for incorporation into analyses.

Analyses

I performed repeated measures mixed model regressions to examine differences in total avian abundance and species richness, and avian abundance and species richness by category and guild, between treated and control sites. Differences were examined both within (e.g., dormant season roller chop) and among (i.e., dormant season roller chop, growing season roller chop, dormant season burn, growing season burn, and roller chop/burn) treatment types. Repeated measures were season and time since treatment (time). Study site pair was included as a blocking factor and presence of grazing as an additional influential independent variable. I used Fisher's Protected LSD tests to make post hoc comparisons. In my results and discussion, I focus on treatment rather than repeated measures or grazing effects. I presented results for 2-, 3-, and 4-way treatment interactions when they occurred. However, as 3- and 4-way treatment interaction effects are difficult to reliably interpret, they were not discussed further (Zar 1999). Due to the timing of data collection, it was not possible to test for 3-way interactions for growing season burning and roller chopping treatments. If, when examining 2-way treatment interactions, differences in linear combinations of groups or biologically meaningless comparisons (e.g., avian abundance in dormant season burn sites in year 1 versus avian abundance in control sites in year 2) arose, I stated that post hoc comparisons revealed no differences based on treatment and the interacting factor.

To determine the combination of habitat characteristics that best described changes in avian abundance and species richness, both overall and by category and guild, I used backward stepwise multiple regression. Multicollinearity problems were reduced by subjecting all predictor variables involved in pairwise correlations with $r \geq 0.7$, to a univariate, one-way analysis of variance (ANOVA) with each dependent variable. For each pair of highly correlated predictor variables, I retained the one with the greatest F value (Noon 1981, McGarigal et al. 2000). All regression models were fit using a Tolerance = 0.001, F-to-enter = 0.15, and F-to-remove = 0.15. These values are considered appropriate for predictor variables that are relatively independent (SYSTAT 2007). I considered regression models statistically and biologically significant at $P \leq 0.1$ and $R^2 \geq 0.2$. Only models considered significant are presented. The relative importance of each variable in the best model was assessed by examining standardized regression coefficients (SC; i.e., variables with higher coefficients made greater individual contributions to the explanatory power of the model).

All data sets were rank transformed prior to analyses due to violations of normality and homogeneity of variance assumptions (Conover 1998, Zar 1999, SYSTAT 2007). I concluded statistical significance at $P \leq 0.1$, rather than the more common $P \leq 0.05$, to minimize the probability of making a Type II error (Mapstone 1995, Zar 1999). All statistical tests were performed using SYSTAT (2007) statistical software.

Results

Dormant Season Burn

I recorded 52 bird species on dormant season burn sites over the course of the study (Table 5-1). Non-breeding category richness was affected by dormant season burning alone (Table 5-2). Within the breeding category, a dormant season burn *

season * grazing interaction affected woodland guild abundance ($P = 0.071$).

Successional-scrub and wetland guild abundance were affected by a dormant season burning * time * season interaction ($P = 0.099$ and $P = 0.063$, respectively). Total avian richness, permanent resident, migrant and breeding category richness, and short distance migrant, neo-tropical migrant, urban, and grassland guild richness were unaffected by dormant season burning alone and in all combinations with time, season, and grazing ($P \geq 0.118$).

Permanent resident category abundance was affected by a dormant season burning * season * grazing interaction ($P = 0.082$). Migrant and non-breeding category abundance were affected by a dormant season burning * season interaction (Table 5-3). Within the migrant category, a dormant season burning * season interaction also affected neo-tropical migrant guild richness (Table 5-3). Within the breeding category, a dormant season burning * grazing interaction affected successional-scrub abundance (Table 5-4). However, no differences in abundance for this guild based on burning and grazing were observed from post hoc comparisons. A dormant season burning * time * season interaction affected wetland guild abundance ($P = 0.072$). Dormant season burning alone and in all combinations with time, season, and grazing had no effect on total avian abundance, breeding category abundance, and short distance migrant and urban guild abundance ($P \geq 0.128$).

Growing Season Burn

I identified 46 bird species on growing season burn sites throughout the 2 years of the study (Table 5-1). Migrant category richness was affected by growing season burning alone (Table 5-2). Within the migrant category, growing season burning alone also affected neo-tropical migrant guild richness (Table 5-2). Non-breeding category

richness was affected by a growing season burning * season interaction (Table 5-3). Within the breeding category, urban guild richness was affected by growing season burning alone (Table 5-2). Wetland guild richness was affected by a growing season burning * season interaction (Table 5-3). However, examination of post hoc comparisons revealed no differences in richness for this guild based on burning and season. Total avian richness, permanent resident and breeding category richness, and short-distance migrant, woodland, successional-scrub, and grassland guild richness were unaffected by growing season burning alone and in all combinations with time, season, and grazing ($P \geq 0.170$)

Non-breeding category abundance was affected by a growing season burning * season interaction (Table 5-3). Within the breeding category, wetland guild abundance was affected by growing season burning alone (Table 5-2). Growing season burning alone and in all possible combinations with time, season, and grazing had no effect on total avian abundance, permanent resident, migrant, and breeding category abundance, and short-distance migrant, neo-tropical migrant, woodland, successional-scrub, and grassland guild abundance ($P \geq 0.119$).

Dormant Season Roller Chop

I detected 48 bird species on dormant season roller chop sites over the study period (Table 5-1). Permanent resident and breeding category richness were affected by a dormant season roller chopping * grazing interaction (Table 5-4). However, examination of post hoc comparisons revealed no differences in richness for these categories based on roller chopping and season. Within the breeding category, grassland guild richness was affected by a dormant season roller chopping * time interaction (Table 5-5). Total avian richness, migrant and breeding category richness,

and short-distance migrant, neo-tropical migrant, woodland, urban, successional-scrub, and wetland guild richness were unaffected by dormant season roller chopping alone and in all combinations with time, season, and grazing ($P \geq 0.113$).

Breeding category abundance was affected by a dormant season roller chopping * time interaction (Table 5-5). However, no differences in abundance for this guild based on roller chopping and time were observed from post hoc comparisons. Dormant season roller chopping alone and in all possible combinations with time, season, and grazing had no affect on total avian abundance, permanent resident, migrant, and non-breeding category abundance, and short-distance migrant, neo-tropical migrant, woodland, urban, successional-scrub, wetland, and grassland guild abundance ($P \geq 0.101$).

Growing Season Roller Chop

I observed 39 bird species on growing season roller chop sites over the course of the study (Table 5-1). Within the migrant category, short-distance migrant guild richness was affected by a growing season roller chopping * season * grazing interaction ($P = 0.090$). Within the breeding category grassland guild richness was affected by a growing season roller chopping * time interaction (Table 5-5). However, examination of post hoc comparisons revealed no differences in richness for this guild based on roller chopping and time. A growing season roller chopping * time * season * grazing interaction affected urban guild richness ($P = 0.087$). Total avian richness, permanent resident, migrant, non-breeding, and breeding category richness, and neo-tropical migrant, woodland, successional-scrub, wetland, and grassland guild richness were unaffected by growing season roller chopping alone and in all combinations with time, season, and grazing ($P \geq 0.280$).

Permanent resident category abundance was affected by growing season roller chopping alone (Table 5-2). Within the migrant guild, short-distance migrant guild abundance was affected by a growing season roller chopping * season * grazing interaction ($P = 0.090$). Growing season roller chopping alone affected breeding category abundance (Table 5-2). With the breeding category, successional-scrub, and grassland guild abundance were also affected by growing season roller chopping alone (Table 5-2). Growing season roller chopping alone and in all combinations with time, season, and grazing had no effect on total avian abundance, migrant and non-breeding category abundance, and neo-tropical migrant, woodland, urban, and wetland guild abundance ($P \geq 0.137$).

Roller Chop/Burn

I recorded 48 bird species were recorded on roller chop/burn sites during the 2 years of the study (Table 5-1). Non-breeding category richness was affected by a roller chopping/burning * grazing interaction (Table 5-4). However, post hoc comparisons revealed no differences in richness for this category based on roller chopping/burning and grazing. Within the breeding category, grassland guild richness was affected by roller chopping/burning alone (Table 5-2). Total avian richness, permanent resident, migrant, and breeding category richness, and short-distance migrant, neo-tropical migrant, woodland, urban, successional-scrub, and wetland guild abundance were unaffected by roller chopping/burning alone and in all combinations with time, season, and grazing ($P \geq 0.109$).

Total avian abundance was affected by roller chopping/burning alone (Table 5-2). Roller chopping/burning alone also affected permanent resident category abundance (Table 5-2). Migrant category abundance was affected by a roller chopping/burning *

time * season * grazing interaction ($P = 0.063$). A roller chopping/burning * grazing interaction affected non-breeding category abundance. However, no differences in abundance for this category based on roller chopping and grazing were observed from post hoc comparisons. Breeding category abundance was affected by roller chopping/burning alone (Table 5-2). Within the breeding category, grassland guild abundance was also affected by roller chopping/burning alone (Table 5-2). Urban guild abundance was affected by a roller chopping/burning * season * grazing interaction ($P = 0.026$). Roller chopping/burning alone and in all combinations with time, season, and grazing had no effect on short-distance migrant, woodland, successional-scrub, and wetland guild abundance ($P \geq 0.124$).

Treatment Type Comparisons

Total avian richness was affected by treatment type alone (Table 5-6). Treatment type alone also affected permanent resident category richness (Table 5-6). Non-breeding category richness was affected by a treatment type * season interaction (Table 5-7). Treatment type alone affected breeding category richness (Table 5-6). Within the breeding category, woodland guild richness was affected by a treatment type * time (Table 5-8) and a treatment type * grazing (Table 5-9) interaction. Urban guild richness was also affected by a treatment type * grazing interaction (Table 5-9). A treatment type * time * season interaction affected successional scrub, wetland and grassland guild richness ($P \leq 0.087$). Migrant category ($P = 0.045$) and neo-tropical migrant guild ($P = 0.022$) richness were affected by a treatment type * time * season * grazing interaction. Treatment type alone and in all combinations with time, season, and grazing had no effect on short-distance migrant guild richness ($P \geq 0.366$).

Total avian abundance was affected by a treatment type* grazing interaction (Table 5-9). A treatment type * season * grazing interaction affected permanent resident category abundance ($P = 0.030$). Migrant category ($P = 0.013$) and neotropical migrant guild ($P = 0.011$) abundance were affected by a treatment type * time * season * grazing interaction. A treatment type * season interaction had an effect on non-breeding category abundance (Table 5-7). Breeding category abundance was affected by a treatment type * season * grazing interaction ($P = 0.033$). Within the breeding category, woodland guild abundance was affected by a treatment type* time (Table 5-8) and a treatment type * grazing interaction (Table 5-9). Successional-scrub and grassland guild abundance were also affected by a treatment type * grazing interaction (Table 5-9). However, post hoc comparisons revealed no differences in grassland guild abundance based on treatment type and grazing. Grassland abundance was affected by a treatment type * grazing interaction (Table 5-7). A treatment type * season * grazing interaction affected urban guild abundance ($P = 0.018$). Short-distance migrant and wetland guild abundance were unaffected by treatment type alone and in all combinations with time, season, and grazing ($P \geq 0.108$).

Avian-Habitat Relationships

Relationships among avian species richness and abundance and habitat characteristics were generally weak and best models contained large numbers of variables. The combination of vegetation characteristics having the greatest affect on avian abundance and richness differed by category and guild (Tables 5-10). Vegetation characteristics that positively affected richness and abundance with many avian categories and guilds were related to shrub richness, cover, and density, forb cover and height, and arthropod familial richness. Vegetation characteristics that negatively

affected species richness and abundance within many avian categories and guilds were shrub richness, cover, and density, forb richness, cover, and height, and graminoid cover.

Discussion

There are negative effects of dormant season burning on the non-breeding, overwintering avian community of pine flatwoods habitats, including declines in species richness and abundance. Few studies have shown strong effects of burn season on avian communities (Engstrom et al. 1996, King et al. 1998). However, it has been proposed that declines in the survival of overwintering species can occur following dormant season burning, due in part to decreases in herbaceous ground cover important as a food and cover source (Robbins and Myers 1992, Engstrom et al. 2005, Thatcher et al. 2006). In pine flatwoods, dormant season burning has little impact on understory and shrubby vegetation density and cover, while causing reductions in graminoid and forb species richness, cover, and height and litter cover and depth for at least the first year post-treatment (Chapter 2). It is suspected that dormant season burning may exert a negative impact on non-breeding, overwintering species by removing herbaceous cover and foraging substrate of species that are active close to the ground (Engstrom et al. 2005, Thatcher et al. 2006). The effects of growing season burning on the non-breeding, overwintering avian community of pine flatwoods were in contrast largely positive, with increases observed in species richness and abundance. Growing season burning tends to cause a greater reduction in understory and shrubby vegetation than dormant season burning (Chapter 2 and 3), especially if repeated fires are implemented over multiple years (Waldrop et al. 1987). In addition, although declines in forb and graminoid species richness, cover, and height may be observed for

2 years following treatment application (Chapter 2), growing season burning can stimulate the growth, flowering, and seeding of certain herbaceous species common in pine flatwoods. These include wiregrass (*Aristida stricta* Michx.), cutthroat grass (*Panicum abscissum* Swallen), toothache grass (*Ctenium aromaticum* (Walter) Alph. Wood), little bluestem (*Schizachyrium rhizomatum* [Swallen] Gould), and other bluestem grasses (*Andropogon* spp.; Myers and Boettcher 1987, Platt et al. 1988, Outcault 1990, Streng et al. 1993). Resulting increases in seed and insect production following growing season burning may be of potential benefit to ground foraging species during the winter months (Engstrom et al. 2006, Thatcher et al. 2006), helping to maintain and even increase species richness and abundance of non-breeding, overwintering species.

Growing season burning had no effect on species richness and abundance within the breeding category. However, 2 guilds within this category, urban and wetland, exhibited increases in richness or abundance following growing season burning. In addition, richness and abundance of all other breeding category guilds were maintained. Growing season burning also resulted in increases in migrant category and neo-tropical migrant guild species richness. These increases may reflect changes in the occurrence of some migratory breeding birds. Although research suggests growing season burning may be beneficial to non-breeding, overwintering birds, concerns have been raised that the practice may be detrimental to nesting species. This concern arises from the potential threat fires set during the growing season may have on the nests of breeding birds, with damage to the nests of game species often of particular issue (Stoddard 1931, Sisson et al. 1990, Hermann et al. 1998, Engstrom et al. 2005). However, the indirect benefits of habitat alteration are likely beneficial to breeding species and are

thought to outweigh potential nest losses (Engstrom et al. 2005, Robbins and Myers 1992, Cox and Widner 2008). Growing season burning may lead to improved late summer and fall brood habitat (Cox and Widener 2008) through potential increases in herbaceous growth (Myers and Boettcher 1987, Platt et al. 1988, Outcault 1990, Streng et al. 1993, Provencher et al. 1998) and insect food sources (Hanula and Fanzeb 1998, Provencher et al. 1998, Collins et al. 2003, Hardy 2003). However, in pine flatwoods growing season burning can result in reduced herbaceous growth through the second year post-treatment (Chapter 2). In addition, insect response to growing season burning can be highly variable and declines in familial richness and abundance, particularly within the Hemiptera, Araneida, and Blattodea, have been observed (Chapter 4), potentially causing reductions in food. Therefore, there is a need to be cautious in assuming improved habitat following growing season burning as declines in avian abundance have been observed soon after treatment application (Fitzgerald and Tanner 1992). However, in general, recent research has concluded growing season burning may not be as problematic to breeding birds as once thought (Cox and Jones 2007, Tucker et al. 2004).

The results of this study suggest avian communities were largely unaffected by dormant season roller chopping. However, growing season roller chopping resulted in increases in permanent resident and breeding category abundance. Within the breeding category, increases in abundance were observed for successional scrub and grassland guilds. The effects of roller chopping on avian communities have been examined in dry prairie habitats, with reductions in species richness and abundance observed on dormant and growing season roller-chopped sites (Fitzgerald and Tanner

1992). Proposed reasons for these declines include reduced perch availability and vertical vegetation diversity. Reductions in shrub cover following treatment application significantly reduced the abundance of shrub-level inhabitants such as white-eyed vireo (*Vireo griseus* Boddaert), northern cardinal (*Cardinalis cardinalis* L.), and grey catbird (*Dumetella caroliniensis* L.; Fitzgerald and Tanner 1992). In pine flatwoods, growing season roller chopping causes prolonged (≥ 2 year) reductions in shrub cover, height, and density and herbaceous plant species richness, cover, and height (Chapter 2 and 3). However, these changes appear to maintain or enhance abundance and species richness within certain avian categories and guilds. This includes increases in successional scrub guild abundance, even though this guild contains species observed to decline following roller chopping of dry prairie (Fitzgerald and Tanner 1992).

Roller chopping/burning had a positive effect on a number of avian categories and guilds. Grassland guild richness and total avian, permanent resident and breeding category, and grassland guild abundance increased following the use of this practice. No other study examining the effect of this combination treatment on avian communities has been conducted. Prescribed burns were applied following roller chopping, typically in the early growing season. This, as for growing season burning and roller chopping, resulted in declines in herbaceous species richness, cover, and abundance. However, due to the timing of burning there is potential for increased seed (Myers and Boettcher 1987, Platt et al. 1988, Outcault 1990, Streng et al. 1993, Hardy 2003) and insect production (Streng et al. 1993, Hanula and Fanzeb 1998, Provencher et al. 1998, Collins et al. 2003, Hardy 2003). This may improve food and cover conditions for certain species and explain the increases in species richness and abundance observed.

Grazing did not interact with the effects of burning and roller chopping to impact avian communities. However, grazing should be applied cautiously in areas where avian conservation and management are a priority. Grazing can result in decreased spatial heterogeneity through reductions in plant biomass and cover and changes in structural conditions (e.g., plant density and height, and litter cover and depth; Chapter 2; Vallentine 1990, Milchunas and Lauenroth 1993, Fuhlendorf and Engle 2001, Derner et al. 2009). In addition, species such as creeping bluestem (*Schizachyrium scoparium* [Michx.] Nash var. *stoloniferum* [Nash] Wipff), chalky bluestem (*Andropogon capillipes* Nash) and wiregrass decline when grazed immediately following burning (White and Terry 1979, Sievers 1985). These changes in the plant community can reduce the suitability and availability of food and cover resources for a variety of avian species (Saab et al. 1995, Brennan and Kuvaesky 2005, Coppedge et al. 2008, Derner et al. 2009). However, rather than managing livestock for uniform use of vegetation or “management to the middle,” with extremes in vegetations structure (e.g., low sparse and high dense) absent (Derner et al. 2009), there is the potential for them to be used as ecosystem engineers. Herbivores naturally exhibit preference for the consumption of certain plants over others (Van Soest 1996). If stocking rates are appropriate and pastures of a sufficient size, this results in differential patterns of use of individual plant species across a pasture (Launchbaugh and Howery 2005), maintaining or increasing the heterogeneity of vegetation and potential benefiting avian species (Derner et al. 2009).

Habitat characteristics that were most often identified as positively related to avian category and guild species richness were shrub richness, cover, and density, forb cover

and height, and arthropod familial richness. All of the treatments examined caused reductions in some, if not all, of these habitats characteristics (Chapter 2, 3, and 5). However, some avian categories and guilds were negatively affected by these habitat characteristics as well as graminoid cover and forb richness. Treatments, therefore, stand to benefit some avian communities through reductions in these habitat characteristics. Generally, treatment applications need to be carefully considered to most appropriately benefit the avian community of primary concern. To maintain biodiversity, it seems most appropriate to adopt a strategy of fire and roller chopping application that is diverse in season, frequency, and space.

Management Implications

Dormant season burning should be used cautiously in situations where conservation and management of non-breeding, overwintering species is a priority. The use of this practice can result in decreases in species richness and abundance within this category, potentially due to reductions in ground level food and cover resources. In situations where the maintenance or enhancement of the non-breeding avian community is desired, growing season burning appears more beneficial and should be used in preference to dormant season burning. This practice also has the potential to benefit the migrant bird community and promote species richness and abundance within certain breeding bird guilds.

The use of growing season roller chopping appears to provide a potential treatment alternative in situations where positive effects on permanent resident and breeding species are desired. However, studies examining the effects of this treatment on avian communities have been contradictory and further research should be conducted before it is widely applied as an avian management and conservation

strategy. The same applies to roller chopping/prescribed burning, the effects of which have been little studied. Maintenance of avian biodiversity may be best achieved through the diverse application of fire and roller chopping based on season, frequency, and space.

Table 5-1. Avian migratory and breeding category composition and abundance in prescribed burned and roller chopped Florida flatwoods, 2007–2008.

Guild ^a	Common Name	Scientific Name	Abundance (no. of individuals)									
			Dormant Burn		Growing Burn		Dormant Roller Chop		Growing Roller Chop		Roller Chop/Burn	
			C ^b	T ^c	C	T	C	T	C	T	C	T
RE, BD(WD)	American crow	<i>Corvus brachyrhynchos</i>	66	84	32	27	21	23	13	10	34	14
MI(SD), NBD	American robin	<i>Turdus migratorius</i>	21	50	0	6	0	3	2	1	1	4
RE, BD(WD)	Bachman's sparrow	<i>Aimophila botterii</i>	42	86	41	39	24	29	28	33	33	32
MG(NM), NBD	Barn swallow	<i>Hirundo rustica</i>	6	0	0	0	3	1	0	0	1	5
RE, BD(WD)	Barred owl	<i>Strix varia</i>	0	0	1	0	1	1	0	0	0	0
MG(NM), NBD	Black and white warbler	<i>Mniotilta varia</i>	0	1	0	0	0	0	0	0	0	0
RE, BD(WT)	Black-bellied whistling duck	<i>Dendrocygna autumnalis</i>	0	0	0	6	0	0	0	0	0	0
RE, BD(WD)	Blue-gray gnatcatcher	<i>Poliotilta caerulea</i>	18	5	4	1	1	5	7	2	8	7
MG(SM), BD(SS)	Brown-headed cowbird	<i>Molothrus ater</i>	17	1	0	0	0	0	0	0	0	0
RE, BD(WD)	Brown-headed nuthatch	<i>Sitta pusilla</i>	18	13	6	7	14	8	14	33	8	14
RE, BD(SS)	Black vulture	<i>Coragyps atratus</i>	3	0	0	0	0	0	0	0	1	0
RE, BD(UB)	Blue jay	<i>Cyanocitta cristata</i>	19	20	16	15	11	14	14	5	7	16
RE, BD(WT)	Boat-tailed grackle	<i>Quiscalus major</i>	1	38	0	0	5	3	0	0	0	1
RE, BD (SS)	Brown thrasher	<i>Toxostoma rufum</i>	1	1	0	1	2	0	1	0	3	0
RE, BD(WD)	Carolina chickadee	<i>Poecile carolinensis</i>	1	2	0	0	0	0	0	1	0	0
RE, BD(SS)	Carolina wren	<i>Thryothorus ludovicianus</i>	26	26	21	13	12	10	8	6	18	11
RE, BD(WT)	Cattle egret	<i>Bubulcus ibis</i>	4	5	0	1	1	1	0	0	0	3
MG(NM), NBD	Chipping sparrow	<i>Spizella passerina</i>	7	9	3	21	2	7	1	2	2	2
RE, BD(UR)	Common grackle	<i>Quiscalus quiscula</i>	72	3	6	1	6	5	3	4	4	16
RE, BD(SS)	Common ground dove	<i>Columbina passerine</i>	11	20	7	8	15	23	7	7	12	14
MG(NM), BR(SS)	Common nighthawk	<i>Chordeiles minor</i>	14	15	12	3	9	9	4	5	11	10
MG(SM), NBD	Common snipe	<i>Gallinago gallinago</i>	0	1	0	0	0	0	0	0	2	3
RE, BD(SS)	Common yellowthroat	<i>Geothlypis trichas</i>	83	76	46	46	27	25	23	20	22	29
RE, BD(WD)	Downy woodpecker	<i>Picoides pubescens</i>	6	5	2	4	3	2	5	3	0	1
RE, BD(GR)	Eastern bluebird	<i>Sialia sialis</i>	11	22	6	6	25	8	22	30	8	21
MG(NM), BD(GR)	Eastern kingbird	<i>Tyrannus tyrannus</i>	4	0	0	0	0	0	0	0	2	0
RE, BD(GR)	Eastern meadowlark	<i>Sturnella magna</i>	67	87	53	69	160	178	121	132	96	142

Table 5-1. Continued

Guild ^a	Common Name	Scientific Name	Abundance (no. of individuals)										
			Dormant Burn		Growing Burn		Dormant Roller Chop		Growing Roller Chop		Roller Chop/Burn		
			C ^b	T ^c	C	T	C	T	C	T	C	T	
MG(SM), NBD	Eastern phoebe	<i>Sayornis phoebe</i>	1	0	0	0	0	0	0	0	0	0	1
RE, BD(SS)	Eastern towhee	<i>Pipilo erythrophthalmus</i>	174	213	77	116	105	106	91	132	119	154	
RE, BD(UB)	Eurasian collard-dove	<i>Streptopelia decaocto</i>	0	0	0	3	0	0	0	0	0	0	
RE, BD(UB)	European starling	<i>Sturnus vulgaris</i>	0	0	0	0	0	0	2	0	0	0	
RE, BD(WD)	Fish crow	<i>Corvus ossifragus</i>	4	3	2	1	2	2	1	0	1	1	
MG(NM), NBD	Gray catbird	<i>Dumetella carolinensis</i>	31	19	2	5	5	4	2	1	3	1	
MG(NM), BD(WD)	Great-crested flycatcher	<i>Myiarchus crinitus</i>	19	16	5	2	12	4	15	13	10	20	
RE, BD(WD)	Great horned owl	<i>Bubo virginianus</i>	0	0	0	0	0	0	1	0	0	0	
MG(NM), NBD	House wren	<i>Troglodytes aedon</i>	0	0	0	0	0	1	0	0	0	0	
RE, BD(GR)	Killdeer	<i>Charadrius vociferous</i>	5	6	0	0	2	1	1	0	1	4	
RE, BD(GR)	Loggerhead shrike	<i>Lanius ludovicianus</i>	5	5	4	6	9	10	9	10	5	4	
RE, BD(WT)	Mottled duck	<i>Anas fulvigula</i>	4	5	2	4	3	4	1	0	3	3	
RE, BD(UB)	Mourning dove	<i>Zenaida macroura</i>	22	38	23	23	42	37	19	27	23	27	
RE, BD(SS)	Northern bobwhite	<i>Colinus virginianus</i>	38	44	38	34	36	36	24	18	22	24	
RE, BD(SS)	Northern cardinal	<i>Cardinalis cardinalis</i>	69	68	45	57	30	28	10	8	15	19	
RE, BD(UB)	Northern mockingbird	<i>Mimus polyglottos</i>	9	20	4	6	27	25	10	1	14	21	
MG(NM), BD(WD)	Northern parula	<i>Parula americana</i>	5	5	6	8	4	3	3	2	4	2	
RE, BD(WD)	Palm warbler	<i>Dendroica palmarum</i>	13	15	0	2	6	3	2	1	0	0	
RE, BD(WD)	Pileated woodpecker	<i>Dryocopus pileatus</i>	7	11	10	10	7	7	7	10	2	6	
RE, BD(WD)	Pine Warbler	<i>Dendroica pinus</i>	4	9	0	1	2	1	0	1	4	3	
RE, BD(WD)	Red-bellied woodpecker	<i>Melanerpes carolinus</i>	51	70	34	35	37	34	34	33	23	36	
RE, BD(WD)	Red-cockaded woodpecker	<i>Picoides borealis</i>	4	7	6	6	6	3	4	4	1	2	
RE, BD(WD)	Red-headed woodpecker	<i>Melanerpes erythrocephalus</i>	6	4	1	2	0	1	2	1	1	1	
RE, BD(WD)	Red-shouldered hawk	<i>Buteo lineatus</i>	8	4	3	4	1	3	4	3	3	2	
RE, BD(WD)	Red-tailed hawk	<i>Buteo jamaicensis</i>	8	2	2	6	7	10	17	12	10	16	
RE, BD(WT)	Red-winged blackbird	<i>Agelaius phoeniceus</i>	45	42	7	12	64	71	44	62	65	74	
RE, BD(WT)	Sandhill crane	<i>Grus canadensis</i>	28	32	18	19	23	22	32	20	36	28	
MG(SM), NBD	Savannah sparrow	<i>Passerculus sandwichensis</i>	17	13	2	2	8	6	0	0	1	1	

Table 5-1. Continued

Guild ^a	Common Name	Scientific Name	Abundance (no. of individuals)									
			Dormant Burn		Growing Burn		Dormant Roller Chop		Growing Roller Chop		Roller Chop/Burn	
			C ^b	T ^c	C	T	C	T	C	T	C	T
MG(NM), BD(WD)	Swallow-tailed kite	<i>Elanoides forficatus</i>	0	0	0	0	0	0	1	0	0	0
MG(SM), NBD	Tree swallow	<i>Tachycineta bicolor</i>	6	0	0	0	0	0	0	0	0	0
RE, BD(WD)	Tufted titmouse	<i>Baeolophus bicolor</i>	6	8	6	9	4	5	2	1	2	8
RE, BD(WD)	Turkey vulture	<i>Cathartes aura</i>	0	2	2	3	0	0	1	0	1	1
RE, BD(SS)	White-eyed vireo	<i>Vireo giseus</i>	24	16	15	11	4	5	2	12	12	6
RE, BD(WD)	Wild turkey	<i>Meleagris gallopavo</i>	7	6	0	1	0	1	2	2	1	1
RE, BD(WT)	Wood stork	<i>Mycteria americana</i>	0	0	1	0	0	0	1	0	0	0
MG(SM), NBD	Yellow-bellied sapsucker	<i>Sphyrapicus varius</i>	0	1	0	0	0	0	0	0	0	0
MG(NM), NBD	Yellow-rumped warbler	<i>Dendroica coronate</i>	3	5	0	0	1	0	0	1	0	1
RE, BD(WD)	Yellow-shafted flicker	<i>Colaptes auratus</i>	3	4	6	4	3	5	3	0	3	2
MG(NM), NBD	Yellow-throated warbler	<i>Dendroica dominica</i>	1	0	0	0	0	0	0	0	0	0

^a Migrant status categories and guilds: RE = Resident category, MG = Migrant category (SM = Short-distance migrant guild, NM = Neo-tropical migrant guild). Breeding status categories and guilds: NBD = Non-breeding category, BD = Breeding category (WD = Woodland guild, UB = Urban guild, SS Successional scrub guild, WT = Wetland guild, GR = Grassland guild).

^b C = Control.

^c T = Treated.

Table 5-2. Effects of treatment on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Abundance and Richness by Treatment Type ^{a,b}	Treatment ($\bar{x} \pm SE$)		<i>P</i>
	Control	Treated	
Dormant season burn			
Richness (no. of species)			
Non-breeding	1.2 ± 0.2	0.9 ± 0.2	0.004
Growing season burn			
Richness (no. of species)			
Migrant	0.6 ± 0.1	1.1 ± 0.2	0.057
Neo-tropical migrant	0.6 ± 0.1	0.9 ± 0.1	0.085
Urban	1.1 ± 0.1	0.8 ± 0.1	0.092
Abundance (no. of individuals)			
Wetland	0.8 ± 0.3	1.2 ± 0.4	0.078
Growing season roller chop			
Abundance (no. of individuals)			
Permanent resident	14.2 ± 1.0	15.6 ± 1.1	0.058
Breeding	14.7 ± 1.0	16.0 ± 1.0	0.060
Successional-scrub	4.2 ± 0.5	4.8 ± 0.4	0.054
Grassland	3.2 ± 0.5	3.5 ± 0.4	0.086
Roller chop/burn			
Richness (no. of species)			
Grassland	0.9 ± 0.1	1.1 ± 1.1	0.021
Abundance (no. of individuals)			
Total	15.4 ± 1.4	19.1 ± 1.1	
Permanent resident	14.6 ± 1.4	17.9 ± 1.1	0.017
Breeding	15.2 ± 1.4	18.7 ± 1.1	0.017
Grassland	2.5 ± 0.4	3.5 ± 0.4	0.058

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by treatment presented ($P \leq 0.1$).

Table 5-3. Effects of treatment * season interactions on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Abundance and Richness by Treatment Type ^{a,b}	Season	Treatment ($\bar{x} \pm SE$) ^c		<i>P</i>
		Control	Treated	
Dormant season burn				
Abundance (no. of individuals)				
Migrant	Winter	4.6 ± 0.9 _A	5.3 ± 1.7 _A	0.038
	Spring	2.1 ± 0.5 _A	0.8 ± 0.2 _B	
	Summer	1.6 ± 0.3 _A	1.3 ± 0.2 _A	
Neo-tropical migrant	Winter	1.9 ± 0.3 _A	2.2 ± 0.4 _A	0.024
	Spring	2.1 ± 0.5 _A	0.8 ± 0.2 _B	
	Summer	1.3 ± 0.3 _A	1.1 ± 0.1 _A	
Non-breeding	Winter	3.8 ± 0.8 _A	5.3 ± 1.7 _A	0.007
	Spring	1.1 ± 0.4 _A	0.1 ± 0.1 _B	
	Summer	0.4 ± 0.1 _A	0.2 ± 0.1 _A	
Growing season burn				
Richness (no. of species)				
Non-breeding	Winter	0.3 ± 0.2 _A	1.3 ± 0.4 _B	0.039
	Spring	0.0 ± 0.0 _A	0.2 ± 0.1 _A	
	Summer	0.1 ± 0.1 _A	0.1 ± 0.1 _A	
Wetland	Winter	0.3 ± 0.2 _A	0.6 ± 0.3 _A	0.075
	Spring	0.3 ± 0.2 _A	0.6 ± 0.3 _A	
	Summer	0.5 ± 0.2 _A	0.5 ± 0.1 _A	
Abundance (no. of individuals)				
Non-breeding	Winter	0.6 ± 0.3 _A	3.0 ± 1.0 _B	0.058
	Spring	0.0 ± 0.0 _A	0.8 ± 0.7 _A	
	Summer	0.1 ± 0.1 _A	0.1 ± 0.1 _A	

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by a treatment * season interaction presented ($P \leq 0.1$).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 5-4. Effects of treatment * grazing interactions on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Abundance and Richness by Treatment Type ^{a,b}	Grazing	Treatment ($\bar{x} \pm SE$) ^c		<i>P</i>
		Control	Treated	
Dorman season burn				
Abundance (no. of individuals)				
Successional scrub	Nongrazed	7.1 ± 1.1 _A	9.6 ± 1.2 _A	0.080
	Grazed	8.0 ± 0.7 _A	7.3 ± 0.5 _A	
Dormant season roller chop				
Richness (no. of species)				
Permanent resident	Nongrazed	8.1 ± 0.8 _A	9.1 ± 0.9 _A	0.072
	Grazed	8.0 ± 0.5 _A	7.8 ± 0.6 _A	
Breeding	Nongrazed	8.7 ± 0.9 _A	9.6 ± 0.9 _A	0.053
	Grazed	8.4 ± 0.5 _A	8.2 ± 0.7 _A	
Roller chop/burn				
Richness (no. of species)				
Non-breeding	Nongrazed	0.3 ± 0.1 _A	0.2 ± 0.1 _A	0.025
	Grazed	0.1 ± 0.1 _A	0.3 ± 0.1 _A	
Abundance (no. of individuals)				
Non-breeding	Nongrazed	0.3 ± 0.1 _A	0.3 ± 0.2 _A	0.031
	Grazed	0.2 ± 0.1 _A	0.5 ± 0.3 _A	

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by a treatment * grazing interaction presented ($P \leq 0.1$).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 5-5. Effects of treatment * time interactions on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Abundance and Richness by Treatment Type ^{a,b}	Time ^c	Treatment ($\bar{x} \pm SE$) ^d		<i>P</i>
		Control	Treated	
Dorman season roller chop				
Abundance (no. of individuals)				
Non-breeding	1	1.1 ± 0.4 _A	0.7 ± 0.3 _A	0.087
	2	0.1 ± 0.1 _A	0.4 ± 0.2 _A	
Growing season roller chop				
Richness (no. of species)				
Grassland	1	1.0 ± 0.1 _A	1.2 ± 0.2 _A	0.059
	2	1.2 ± 0.1 _A	1.1 ± 0.1 _A	

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by a treatment * time interaction presented ($P \leq 0.1$).

^c Time since treatment application (years).

^d Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 5-6. Comparison of the effects of treatment on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Richness and Abundance ^{a,b}	Treatment Type ($\bar{x} \pm SE$) ^c						<i>P</i>
	Control	Dormant Burn	Growing Burn	Dormant Roller Chop	Growing Roller Chop	Roller Chop/Burn	
Richness (no. of species)							
Total	9.7 ± 0.3 _{ABC}	11.0 ± 0.5 _{AB}	10.2 ± 0.6 _{AB}	9.3 ± 0.6 _{AB}	8.6 ± 0.5 _B	9.7 ± 0.5 _C	0.001
Permanent resident	8.5 ± 0.2 _{ABC}	9.2 ± 0.4 _A	9.1 ± 0.6 _{BC}	8.4 ± 0.5 _A	8.0 ± 0.5 _C	8.7 ± 0.5 _{AC}	0.059
Breeding	9.1 ± 0.2 _{AB}	9.8 ± 0.5 _{AC}	9.7 ± 0.7 _B	8.8 ± 0.6 _{AC}	8.5 ± 0.5 _B	9.4 ± 0.5 _C	0.039

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by treatment presented ($P \leq 0.1$).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 5-7. Comparison of the effects of treatment * season interactions on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Richness and Abundance ^{a,b}	Season	Treatment Type ($\bar{x} \pm SE$) ^c					<i>P</i>	
		Control	Dormant Burn	Growing Burn	Dormant Roller Chop	Growing Roller Chop		Roller Chop/Burn
Richness (no. of species)								
Non-breeding	Winter	1.2 ± 0.2 _A	2.3 ± 0.3 _{AB}	1.3 ± 0.4 _B	1.3 ± 0.4 _{AB}	0.4 ± 0.3 _A	0.8 ± 0.3 _{AB}	0.095
	Spring	0.3 ± 0.1 _A	0.1 ± 0.1 _B	0.2 ± 0.1 _{AB}	0.1 ± 0.1 _{AB}	0.0 ± 0.0 _{AB}	0.1 ± 0.1 _{AB}	
	Summer	0.2 ± 0.0 _A	0.3 ± 0.1 _A	0.1 ± 0.1 _A	0.1 ± 0.1 _A	0.0 ± 0.0 _A	0.1 ± 0.1 _A	
Abundance (no. of individuals)								
Non-breeding	Winter	1.7 ± 0.3 _A	5.3 ± 1.7 _{AB}	3.0 ± 1.0 _B	1.8 ± 0.6 _{AB}	0.5 ± 0.3 _A	1.1 ± 0.5 _{AB}	0.058
	Spring	0.5 ± 0.1 _A	0.1 ± 0.1 _B	0.8 ± 0.7 _{AB}	0.1 ± 0.1 _{AB}	0.0 ± 0.0 _{AB}	0.4 ± 0.3 _{AB}	
	Summer	0.2 ± 0.1 _A	0.2 ± 0.1 _A	0.1 ± 0.1 _A	0.1 ± 0.1 _A	0.0 ± 0.0 _A	0.1 ± 0.0 _A	
Grassland	Winter	2.0 ± 0.4 _{AB}	1.4 ± 0.4 _B	1.6 ± 0.6 _B	2.4 ± 0.6 _{AB}	4.3 ± 0.8 _A	4.9 ± 1.0 _A	0.016
	Spring	2.8 ± 0.4 _A	1.3 ± 0.4 _B	2.9 ± 1.1 _{AB}	4.8 ± 0.8 _A	3.3 ± 1.0 _{AB}	2.0 ± 0.4 _{AB}	
	Summer	2.0 ± 0.4 _A	2.0 ± 0.4 _A	1.9 ± 0.5 _A	4.0 ± 0.1 _A	3.2 ± 0.5 _A	2.5 ± 0.4 _A	

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by a treatment * season interaction presented ($P \leq 0.1$).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 5-8. Comparison of the effects of treatment * time interactions on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Richness and Abundance ^{a,b}	Time ^c	Treatment Type ($\bar{x} \pm SE$) ^d						<i>P</i>
		Control	Dormant Burn	Growing Burn	Dormant Roller Chop	Growing Roller Chop	Roller Chop/Burn	
Richness (no. of species)								
Woodland	1	3.5 ± 0.2 _A	4.0 ± 0.4 _A	5.1 ± 0.9 _A	2.6 ± 0.5 _A	2.8 ± 0.7 _B	2.9 ± 0.6 _A	0.085
	2	3.0 ± 0.2 _A	3.4 ± 0.3 _A	3.2 ± 0.3 _A	3.0 ± 0.4 _A	3.1 ± 0.3 _A	3.0 ± 0.4 _A	
Abundance (no. of individuals)								
Woodland	1	4.0 ± 0.3 _A	6.1 ± 0.5 _{AB}	5.4 ± 0.9 _B	2.9 ± 0.3 _{AB}	2.4 ± 0.6 _A	3.5 ± 0.7 _{AB}	0.022
	2	4.5 ± 0.3 _{AC}	5.6 ± 0.6 _A	4.5 ± 0.5 _A	4.4 ± 0.6 _B	5.3 ± 0.7 _C	4.9 ± 0.8 _{ABC}	

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by a treatment * time interaction presented ($P \leq 0.1$).

^c Time since treatment application (years).

^d Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 5-9. Comparison of the effects of treatment * grazing interactions on avian richness and abundance in Florida flatwoods, 2007–2008.

Avian Richness and Abundance ^{a,b}	Grazing	Treatment Type ($\bar{x} \pm SE$) ^c						P
		Control	Dormant Burn	Growing Burn	Dormant Roller Chop	Growing Roller Chop	Roller Chop/ Burn	
Richness (no. of species)								
Woodland	Nongrazed	3.6 ± 0.2 _{AB}	3.4 ± 0.7 _A	3.9 ± 0.4 _{AB}	3.7 ± 0.5 _B	3.9 ± 0.3 _C	3.0 ± 0.4 _{ABC}	0.010
	Grazed	2.8 ± 0.2 _A	3.8 ± 0.3 _A	3.0 ± 0.7 _A	2.1 ± 0.3 _A	1.7 ± 0.5 _A	3.0 ± 0.5 _A	
Urban	Nongrazed	1.0 ± 0.1 _{AB}	1.5 ± 0.3 _A	0.9 ± 0.2 _{AB}	1.0 ± 0.2 _{AB}	1.3 ± 0.3 _{AB}	0.7 ± 0.2 _B	0.004
	Grazed	0.9 ± 0.1 _A	0.9 ± 0.1 _A	0.5 ± 0.3 _A	1.0 ± 0.2 _A	0.9 ± 0.1 _A	1.5 ± 0.2 _A	
Abundance (no. of individuals)								
Total	Nongrazed	16.4 ± 0.8 _A	23.0 ± 1.1 _A	19.4 ± 1.8 _A	15.9 ± 1.3 _A	16.1 ± 1.4 _A	18.9 ± 1.7 _A	0.053
	Grazed	17.4 ± 1.0 _A	21.7 ± 2.1 _B	15.7 ± 1.9 _{AB}	17.9 ± 1.4 _A	16.2 ± 1.7 _{AB}	19.4 ± 1.5 _A	
Woodland	Nongrazed	4.8 ± 0.3 _A	4.7 ± 0.9 _B	4.9 ± 0.5 _A	4.4 ± 0.6 _C	5.6 ± 0.7 _D	4.3 ± 0.7 _A	0.034
	Grazed	4.0 ± 0.3 _A	6.1 ± 0.4 _A	4.1 ± 1.2 _A	3.2 ± 0.6 _B	3.0 ± 0.9 _{AB}	6.1 ± 0.4 _A	
Successional-scrub	Nongrazed	5.5 ± 0.4 _A	9.6 ± 1.2 _B	8.1 ± 0.9 _{AB}	4.8 ± 0.6 _{AB}	4.7 ± 0.6 _A	6.2 ± 0.7 _{AB}	0.003
	Grazed	6.5 ± 0.4 _{AB}	7.3 ± 0.5 _A	8.9 ± 1.0 _B	5.2 ± 0.6 _{AB}	5.0 ± 0.7 _A	7.3 ± 0.5 _{AB}	
Grassland	Nongrazed	2.4 ± 0.2 _A	1.3 ± 0.6 _A	2.3 ± 0.5 _A	2.8 ± 0.3 _A	2.7 ± 0.4 _A	3.9 ± 0.6 _A	0.054
	Grazed	2.4 ± 0.2 _A	1.6 ± 0.3 _A	1.4 ± 0.6 _A	4.8 ± 0.6 _A	4.8 ± 0.7 _A	1.6 ± 0.3 _A	

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Only avian residency and breeding categories and guilds with richness or abundance significantly affected by a treatment * grazing interaction presented ($P \leq 0.1$).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

Table 5-10. Habitat characteristics that best predict total, category, and guild-specific avian abundance and richness in Florida flatwoods, 2007–2008.

Dependant Variable ^a	Independent Model Variables ^b	Standardized Regression Coefficient	R ²	P	
Richness (no. of species)	Total				
		Mean soil moisture (%)	0.233	0.242	≤0.001
		Shrub density (no./m ²)	0.187		
		Mean forb richness (no. of species)	-0.175		
		Mean forb height (cm)	0.158		
		Shrub cover (%)	0.158		
		Mean soil density (g/cm ³)	0.133		
		Mean graminoid cover (%)	-0.125		
		Mean litter depth (cm)	-0.118		
		Arthropod family richness (no. of families)	0.103		
		Shrub richness (no. of species)	0.096		
		Variance of soil density (g/cm ³)	0.084		
		Variance of forb cover (%)	-0.077		
		Variance of soil pH	-0.062		
	Variance of graminoid richness (no. of species)	-0.052			
Permanent resident		Mean soil moisture (%)	0.212	0.246	≤0.001
		Shrub cover (%)	0.175		
		Mean forb height (cm)	0.166		
		Mean forb richness (no. of species)	-0.160		
		Shrub density (no./m ²)	0.155		
		Mean soil density (g/cm ³)	0.139		
		Mean graminoid cover (%)	-0.136		
		Mean litter depth (cm)	-0.109		
		Shrub richness (no. of species)	0.106		
		Arthropod family richness (no. of families)	0.104		
		Variance of forb cover (%)	-0.101		
		Mean forb cover (%)	0.097		
		Variance of soil pH	-0.077		
		Variance of soil density g/cm ³)	0.076		
Breeding		Mean soil moisture (%)	0.219	0.264	≤0.001
		Shrub density (no./m ²)	0.172		
		Mean forb height (cm)	0.168		
		Shrub cover (%)	0.163		
		Mean graminoid cover (%)	-0.158		
		Mean forb richness (no. of species)	-0.153		

Table 5-10. Continued

Dependant Variable ^a	Independent Model Variables ^b	Standardized Regression		
		Coefficient	R ²	P
Woodland	Mean soil density (g/cm ³)	0.147		
	Arthropod family richness (no. of families)	0.112		
	Shrub richness (no. of species)	0.107		
	Mean litter depth (cm)	-0.106		
	Variance of soil density (g/m ³)	0.097		
	Variance of forb cover (no. of species)	0.090		
	Variance of pH	0.090		
	Mean forb cover (%)	0.084		
	Mean soil moisture (%)	0.311	0.288	≤0.001
	Shrub richness (no. of species)	0.198		
	Mean litter depth (cm)	-0.197		
	Mean forb richness (no. of species)	-0.195		
	Shrub cover (%)	0.182		
	Mean graminoid cover (%)	-0.157		
	Mean forb cover (%)	0.150		
	Overstory basal area (cm ² /ha)	0.136		
	Mean soil density (g/cm ³)	0.126		
	Variance of soil density (g/cm ³)	0.101		
	Variance of litter depth (cm)	0.093		
	Variance of litter cover (%)	-0.079		
Shrub density (no./m ²)	0.078			
Variance of pH	-0.069			
Successional scrub	Mean forb height (cm)	0.196	0.241	≤0.001
	Shrub density (no./m ²)	0.190		
	Mean forb richness (no. of species)	-0.181		
	Shrub cover (%)	0.150		
	Variance of graminoid height (cm)	-0.124		
	Mean litter cover (%)	-0.108		
	Visual obstruction from (% from 5 m distance)	0.101		
	Mean graminoid height (cm)	0.086		
	Variance of graminoid cover (%)	-0.085		
	Mean graminoid cover (%)	-0.059		
	Variance of forb height (cm)	0.050		
	Variance of pH	-0.037		
	Arthropod family richness (no. of families)	0.026		

Table 5-10. Continued

Dependant Variable ^a	Independent Model Variables ^b	Standardized Regression					
		Coefficient	R ²	P			
Wetland	Variance of forb cover (%)	-0.234	0.208	≤0.001			
	Shrub richness (no. of species)	-0.207					
	Mean forb richness (no. of species)	0.132					
	Understory density (no./ha)	-0.125					
	Shrub density (no./m ²)	0.121					
	Arthropod abundance (no. of individuals)	0.118					
	Shrub height (cm)	-0.104					
	Mean graminoid height (cm)	-0.098					
	Mean litter cover (%)	0.088					
	Shrub cover (%)	-0.083					
	Variance of graminoid height (cm)	0.076					
	Mean forb height (cm)	0.069					
	Mean soil moisture (%)	0.064					
	Variance of soil density	-0.058					
	Mean soil density (g/cm ³)	-0.053					
	Grassland	Mean graminoid richness (no. of species)			0.208	0.221	≤0.001
		Shrub cover (%)			0.165		
		Shrub height (cm)			-0.147		
		Overstory basal area (cm ² /ha)			-0.147		
		Variance of graminoid richness (no. of species)			0.132		
Mean soil density (g/cm ³)		0.125					
Understory density (no./ha)		-0.125					
Mean soil moisture (%)		0.105					
Mean graminoid cover (%)		-0.105					
Arthropod family richness (no. of families)		0.097					
Mean forb cover (%)		0.095					
Variance of forb cover (%)		-0.078					
Variance of graminoid height (cm)		0.072					
Variance of soil density (g/cm ³)		0.065					
Mean pH	0.061						
Abundance (No. of Individuals)							
Woodland	Mean litter depth (cm)	-0.240	0.239	≤0.001			
	Mean forb richness (No. of species)	-0.207					
	Overstory density (No./ha)	0.198					
	Shrub cover (%)	0.190					
	Mean soil moisture (%)	0.176					
	Mean forb cover (%)	0.165					

Table 5-10. Continued

Dependant Variable ^a	Independent Model Variables ^b	Standardized Regression Coefficient	R ²	P
Successional scrub	Shrub richness (no. of species)	0.158	0.252	≤0.001
	Arthropod abundance (no. of individuals)	-0.151		
	Mean graminoid richness (no. of species)	0.130		
	Variance of graminoid richness (no. of species)	-0.110		
	Variance of litter height (cm)	0.096		
	Variance of litter cover (%)	-0.092		
	Shrub density (no./m ²)	0.089		
	Mean graminoid cover (%)	-0.074		
	Variance of forb height (cm)	-0.057		
	Mean forb height (cm)	0.220		
	Shrub cover (%)	0.214		
	Shrub richness (no. of species)	0.149		
	Mean litter cover (%)	-0.145		
	Mean forb richness (no. of species)	-0.138		
	Overstory basal area (cm ³ /ha)	0.106		
	Variance of graminoid height (cm)	-0.105		
	Mean pH	-0.101		
	Variance of soil moisture (%)	-0.090		
	Understory cover (%)	0.074		
	Variance of forb height (cm)	0.057		
Wetland	Mean soil density (g/cm ³)	-0.048	0.224	≤0.001
	Variance of litter depth (cm)	-0.047		
	Variance of graminoid richness (no. of species)	-0.045		
	Variance of graminoid cover (%)	-0.044		
	Mean forb richness (no. of species)	0.190		
	Variance of forb cover (%)	-0.186		
	Shrub richness (no. of species)	-0.142		
	Mean litter cover (%)	0.130		
	Understory density (no./ha)	-0.105		
	Visual obstruction (% from 10 m distance)	-0.104		
	Mean soil density (g/cm ³)	-0.098		
	Shrub height (cm)	-0.097		
	Variance of soil density (g/cm ³)	-0.096		
	Mean graminoid height (cm)	-0.077		
	Mean forb height (cm)	0.076		
Variance of pH	0.072			
Variance of graminoid height (cm)	0.068			

Table 5-10. Continued

Dependant Variable ^a	Independent Model Variables ^b	Standardized Regression		
		Coefficient	R ²	P
Grassland	Overstory cover (%)	-0.060	0.214	≤0.001
	Arthropod abundance (no. of individuals)	0.052		
	Mean forb richness (no. of species)	0.208		
	Understory density (no./ha)	-0.146		
	Overstory basal area (cm ³ /ha)	-0.138		
	Shrub height (cm)	-0.123		
	Mean soil density (g/cm ³)	0.117		
	Variance of graminoid richness (no. of species)	0.099		
	Shrub cover (%)	0.086		
	Variance of forb cover (%)	-0.083		
	Variance of forb height (cm)	0.064		
	Mean pH	0.060		
	Mean graminoid richness (no. of species)	0.057		
	Variance of graminoid height (cm)	0.057		
	Visual obstruction (% from 10 m distance)	-0.055		
	Mean litter depth (cm)	0.028		
Variance of soil density (g/cm ³)	-0.023			

^a Migrant status categories and guilds: resident category, migrant category (short distance migrant and neo-tropical migrant guilds). Breeding status categories and guilds: non-breeding category, breeding category (woodland, urban, successional scrub, wetland and grassland guilds).

^b Vegetation characteristics selected using backward stepwise multiple linear regression.

CHAPTER 6 DIURNAL LEPIDOPTERAN RESPONSE TO PRESCRIBED BURNING AND ROLLER CHOPPING IN FLORIDA FLATWOODS

Introduction

Lepidoptera (butterflies and moths) play an essential role in natural systems as herbivores and pollinators (Scott 1986, Hendrix and Kyhl 2000), and have an ability to foster public sympathy, something that is unusual among insects. They are often considered a flagship or umbrella taxa for the conservation of other wildlife and have the potential to act as indicators of habitat type, quality, and/or condition (Erhardt 1985, Kremen 1992, Launer and Murphy 1994, Nelson and Anderson 1994, New 1997) and the presence of certain bird species (Swengel and Swengel 1999). As a result, they are frequently the target of invertebrate management, research, and conservation efforts (New 1997).

Many species of Lepidoptera inhabit Florida's pine flatwoods (Gerberg and Arnett 1989, Covell 1984, Opler 1998), including 2 species of butterfly, the arogos skipper (*Atrytone arogos arogos*) and southern dusted skipper (*Atrytonopsis hianna loammi* Whitney), listed by the Florida Fish and Wildlife Conservation Commission (FWC) as species having declining populations and of conservation need (FWC 2005). Unfortunately, the pine flatwoods that provide habitat for these Lepidoptera are also deteriorating and in recent years have exhibited considerable declines in quantity and quality (Abrahamson and Hartnett 1990, United States Fish and Wildlife Service [USFWS] 1999, FWC 2005). Seventy-five percent of Florida's pine flatwoods (Cox et al. 1997) are privately-owned (FWC 2005) and used primarily for livestock production. Changes to management practices in these flatwoods, and pinelands across the southeastern U.S., particularly the modification of historic fire regimes (i.e., deviations in

fire intensity, return frequency and seasonality), have resulted in excessive shrub growth and the loss of herbaceous vegetation (Wade et al. 1980, Robbins and Myers 1992, Glitzenstein et al. 1995, Abrahamson and Abrahamson 1996, Platt 1998). These vegetative changes have altered the structure and composition of flatwoods, contributing to their degradation and reducing their suitability for associated wildlife, including grassland- and open country- associated Lepidopteran species (FWC 2005).

In an attempt to maintain and restore remaining areas of privately owned pine flatwoods, FWC and the United States Department of Agriculture are utilizing assistance-based programs, such as the Farm Bill's Environmental Quality Incentives Program and Wildlife Habitat Incentives Program, to encourage landowners to implement appropriate management activities on their lands. Management activities promoted under these programs include the use of prescribed fire and roller chopping during dormant (November–March) and growing (April–October) seasons. Prescribed fire and roller-chopping are management techniques that have been shown to reduce shrub cover and encourage the growth and flowering of grasses and forbs (Chapter 2; Platt et al. 1988, Tanner et al. 1988, Fitzgerald and Tanner 1992, Glitzenstein et al. 1995, Watts and Tanner 2003, Watts et al. 2006), helping to improve the quality of degraded pineland and prairie habitats where changes in fire use have permitted the proliferation of shrubby vegetation to the detriment of herbaceous groundcover species.

The effects management activities have on Lepidopteran communities occupying Florida's pine flatwoods have not been studied. Many rare butterflies require vegetation management in some form to maintain their populations (New 1991, Oates 1995, Robertson et al. 1995). In Florida's flatwoods, it might be suspected that prescribed

burning and roller-chopping may benefit grassland-associated butterfly species through a reduction in shrub cover and an increase in herbaceous plant species (Chapter 2; Platt 1988, Tanner et al. 1988, Fitzgerald and Tanner 1992, Glitzenstein et al. 1995, Watts and Tanner 2003, Watts et al. 2006), potentially providing supplementary food sources in the form of nectar producing plants and additional sites for egg-laying and caterpillar development. However, in other areas of North America, declines in Lepidopteran species richness and abundance have been observed following fire, at least in the short-term, suggesting that this management practice may threaten populations of locally endangered species (Dunwiddie 1991, Siemann 1997). Swengel (1996, 1998) also observed declines in many Lepidopteran species following fire, and proposed that non-fire vegetation management strategies (e.g., mowing) might be more favorable for maintenance of specialist butterflies than prescribed burning. With assistance-based management programs currently encouraging the use of fire and roller chopping on Florida rangelands, we need to investigate further the impacts these practices have on pine flatwoods-associated Lepidoptera. Such research will determine whether the use of these practices to manage pine flatwoods vegetation is also appropriate for the conservation of this insect group. The objectives of my study were to 1) compare diurnal Lepidopteran species richness and abundance on treated (management activities implemented) and untreated (no management activities implemented) pine flatwoods sites, 2) contrast the species richness and abundance of nectar producing plants on treated and untreated sites, and 3) investigate the relative importance of local pine flatwoods habitat characteristics (e.g., flowering forb and shrub abundance, shrub

density, forb cover, graminoid height) may play in determining Lepidopteran species richness and abundance.

Methods

Study Sites

I conducted research on 50 privately- and publicly-owned, paired treatment and control sites across 6 counties (Desoto, Highlands, Lee, Manatee, Osceola, and Sarasota) in central and south Florida. Study sites consisted of pine flatwoods habitats with varying management histories and grazing regimes that were being prescribed burned and roller chopped by local landowners and land managers using varying, individual protocols. Florida's pine flatwoods are characterized as having an overstory of scattered slash (*Pinus elliotti* Engelm.) and longleaf (*P. palustris* Mill.) pine, either in pure stands or in combination. The understory and shrub layer includes saw palmetto (*Serenoa repens* [Bartram] Small), wax myrtle (*Morella cerifera* [L.] Small), gallberry (*Ilex glabra* [Pursh] Chapm.), fetterbush (*Lyonia lucida* [Lam.] K. Koch), staggerbush (*Lyonia fruticosa* [Michx.] G. S. Torr), dwarf huckleberry (*Gaylussacia dumosa* [Andrews] Torr. & A. Gray), dwarf live oak (*Quercus mimima* [Sarg.] Small), and tarflower (*Bejaria racemosa* Vent.). An appreciable herbaceous layer exists when the shrub layer is relatively open. This layer contains a wide variety of grasses (e.g., *Agrostis*, *Andropogon*, *Aristida*, *Eragrostis*, *Panicum*, and *Paspalum* spp.). Common forbs include legumes (e.g., *Cassia*, *Crotalaria*, *Galactia*, *Tephrosia* spp.), milkweeds (*Asclepias* spp.), milkworts (*Polygala* spp.), and a wide variety of composites (e.g., *Aster*, *Chrysopsis*, *Eupatorium*, *Liatris*, and *Solidago* spp.; Abrahamson and Hartnett 1990, U.S. Fish and Wildlife Service 1999).

Treatment Types

Treatment types included dormant season (November–March) burn, growing season (April–October) burn, dormant season roller chop, growing season roller chop, and a roller chop/burn combination treatment. The roller chop/burn combination treatment (hereafter referred to as roller chop/burn) involved roller chopping in the dormant season followed by burning within 6 months. I established a total of 11 dormant season burn, 9 growing season burn, 9 dormant season roller chop, 12 growing season roller chop, and 9 roller chop/burn site pairs.

Lepidopteran Surveys

I used a paired sampling approach to examine the effects of treatment type (i.e., prescribed burning, roller chopping, and roller chopping/burning) on Lepidopteran species richness and abundance. Richness and abundance were compared between sampling points randomly located in paired treated (e.g., dormant season roller chopped) and untreated (control) flatwoods sites. Paired treatment and control sampling points were adjacent, being located in the same pasture or management unit. In addition, they were of similar current and past management (e.g., grazing intensity), surrounding land-use, plant community (e.g., overstory cover), and soil conditions. I established 1 randomly selected sampling point within each treatment and control site. To minimize edge effects, I rejected and randomly relocated sampling points that fell within 50 m of the edge of a treatment or control site. Sites within which treatment and control sampling points were located ranged from 2–20 ha in size.

I conducted Lepidopteran surveys in spring (March–April) of 2008 following the application of dormant and growing season burning and roller chopping treatments. Lepidoptera were surveyed using line transect techniques along 2 100-m perpendicular

transects centered on the sampling point (Swengel 1998, Giuliano et al. 2004, Shepherd and Dubinsky 2005). I walked each transect at a steady pace of 10 m/min for a total survey time of 20 min (Shepherd and Debinski 2005). During this period, I recorded all Lepidoptera observed through binoculars or captured in a sweep net (Opler and Maluku 1998, Glassberg 1999). The 20-min sampling period did not include capture, processing, or recording of individuals. I only conducted surveys on calm (winds <17km/h), sunny (cloud cover < 60%), and warm (temperature > 18°C) days between 1000 and 1500 hours (Shepherd and Debinski 2005).

Nectar Producing Plant Sampling

I also used a paired sampling approach to examine the effects of treatment type on nectar producing plant species richness and abundance, which were compared at the same sampling points used for Lepidopteran surveys. Nectar producing plant sampling was conducted on the same day as Lepidopteran surveys and involved counting the number of nectar producing forbs and shrubs exhibiting inflorescence within a 0.03-ha nested plot centered on the sampling point (Dueser and Shugart 1978, Higgins et al. 2005).

Habitat Sampling

I conducted habitat sampling in spring (April–May) 2008 at the same points used for Lepidopteran surveys. At each sampling point, plant community composition and structure, litter and soil variables, and vertical obstruction were examined in several strata (i.e., ground, herbaceous, and shrub) within the 0.03-ha nested circular plot (Dueser and Shugart 1978, Higgins et al. 2005).

Ground layer. I assessed litter cover (%; ocular estimate) within 4 1-m² sub-sample plots, 1 randomly located in each quadrant of the 0.03-ha circular plots, along

with soil density (g/cm^3), moisture (%), and pH. Litter cover was recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Donhaue et al. 1971, Hays et al. 1981, Higgins et al. 2005). I measured soil density as the dry weight density (g/cm^3) of a 5-cm diameter, 10-cm deep soil core sample after oven drying at 45°C for 48 hours. Soil pH and moisture were measured using a Kelway soil tester (Rodewald and Yahner 2001).

Herbaceous layer. I determined species richness, cover (%; ocular estimate), and maximum height (cm) of forbs and graminoids within the 1- m^2 subsample plots. Forb and graminoid cover were recorded on a scale: 0 = 0%, 1 = 1–5%, 2 = 5–25%, 3 = 25–50%, 4 = 50–75%, 5 = 75–95%, 6 = 95–99%, and 7 = 100% (Hays et al. 1981, Krebs 1999, Higgins et al. 2005).

Shrub layer. I counted and measured the height of all shrubs (woody vegetation <2.0 m in height) in 2 perpendicular 20- m^2 quadrats centered on the 0.03-ha plot to estimate species richness (no. of species), density (no./ m^2), and maximum height (cm) for individual species and all combined (Hays et al. 1981, Krebs 1999, Higgins et al. 2005). Shrub cover (%) was assessed along 2 perpendicular 20-m transects centered on the 0.03-ha circular plot using the line intercept method (Hays et al. 1981, Higgins et al. 2005).

Analyses

I used mixed model regressions to examine differences in Lepidopteran and nectar producing plant species richness and abundance between untreated (control) and treated sites, both within (e.g., dormant season burn) and among (i.e., dormant season burn, growing season burn, dormant season roller chop, growing season roller chop, and roller chop/burn) treatment types. Study site pair was included in analyses as

a blocking factor. Significant comparisons among treatment type were followed by Fisher's Protected LSD tests.

Multiple linear regressions were used to examine which combination of habitat characteristics best described changes in Lepidopteran species richness and abundance. I subjected all predictor variables involved in pairwise correlations with $r \geq 0.7$ to a univariate, one-way analysis of variance (ANOVA) with each dependent variable to reduce issues with multicollinearity. For each pair of highly correlated predictor variables, I retained the one with the greatest F value (Noon 1981, McGarigal et al. 2000). All regression models were fit using a backward stepwise procedure with Tolerance = 0.001, F-to-enter = 0.15, and F-to-remove = 0.15. These values are considered appropriate for predictor variables that are relatively independent (SYSTAT 2007). I considered regression models statistically and biologically significant at $P \leq 0.1$ and $R^2 \geq 0.2$. Only models considered significant are presented. I assessed the relative importance of each variable in the best model by examining standardized regression coefficients (SC; i.e., variables with higher coefficients made greater individual contributions to the explanatory power of the model).

I rank transformed all data sets prior to analyses due to violations of normality and homogeneity of variance assumptions (Conover 1998, Zar 1999, SYSTAT 2007). Statistical significance was concluded at $P \leq 0.1$ for all tests. I used this value, rather than the more common $P \leq 0.05$, to minimize the probability of making a Type II error (Mapstone 1995, Zar 1999). All statistical tests were performed using SYSTAT (2007) statistical software.

Results

Lepidopteran Species Richness and Abundance

I identified 20 Lepidopteran species during the study (Table 6-1). Lepidopteran species richness was unaffected by dormant and growing season burning and roller chopping and roller chopping/burning ($P \geq 0.129$). Lepidopteran abundance was affected by dormant season burning, decreasing by 64% from 3.3 ± 0.6 ($\bar{x} \pm SE$) individuals on burn to 1.2 ± 0.5 individuals on control sites ($P = 0.026$). No other treatment had an effect on Lepidopteran abundance ($P \geq 0.424$).

Flowering Forb and Shrub Species Richness and Abundance

Flowering forb species richness was affected by growing season roller chopping, increasing by 67% from 4.2 ± 0.8 species on control to 7.0 ± 0.8 species on roller chop sites ($P = 0.011$). Dormant and growing season burning, dormant season roller chopping, and roller chopping/burning had no effect on flowering forb species richness ($P \geq 0.146$).

Dormant season roller chopping affected flowering forb abundance, which increased by 682% from 38.7 ± 18.1 individuals on control to 302.7 ± 159.1 individuals on roller chop sites ($P = 0.017$). Flowering forb species richness was unaffected by all other treatments ($P \geq 0.106$).

Growing season roller chopping affected flowering shrub species richness, which decreased by 39% from 2.8 ± 0.5 species on control to 1.7 ± 0.6 species on roller chop sites ($P = 0.084$). Flowering shrub species richness was also affected by roller chopping/burning, decreasing by 33% from 2.4 ± 0.3 species on control to 1.6 ± 0.3 species on roller chop/burn sites ($P = 0.099$). Comparisons among treatments revealed the effects of growing season roller chopping and roller chopping/burning on flowering

shrub species richness were similar (Table 6-2). Dormant and growing season burning and dormant season roller chopping had no effect on flowering shrub species richness ($P \geq 0.249$).

Growing season roller chopping affected flowering shrub abundance, which decreased by 22% from 72.3 ± 16.3 individuals on control to 55.8 ± 51.3 individuals on roller chop sites ($P = 0.086$). Roller chopping/burning also affected flowering shrub abundance, which decreased by 53% from 21.3 ± 8.0 individuals on control to 10.0 ± 5.0 individuals on burn/roller chop sites ($P = 0.037$). Again, comparisons among treatments revealed the effects of growing season roller chopping and roller chopping/burning on flowering shrub abundance were similar (Table 6-2). Dormant and growing season burning and dormant season roller chopping had no effect on flowering shrub abundance ($P \geq 0.366$).

Lepidoptera-Habitat Relationships

Relationships among Lepidopteran species richness and abundance and habitat characteristics were generally weak and best models contained large numbers of variables. Habitat characteristics that best predicted Lepidopteran species richness were variance of litter depth (SC = 0.393), variance of graminoid cover (SC = -0.320), variance of litter cover (SC = -0.293), shrub height (SC = -0.254), mean graminoid height (SC = 0.251), flowering shrub abundance (SC = -0.249), variance of graminoid height (SC = -0.235), mean graminoid richness (SC = -0.215), variance of soil density (SC = -0.202), mean pH (SC = 0.196), mean forb height (SC = -0.193), and mean litter cover (SC = -0.184; $P \leq 0.001$, $R^2 = 0.358$). Mean forb height (SC = -0.334), variance of graminoid cover (SC = -0.299), shrub height (SC = -0.226), mean litter cover (SC = -0.214), variance of litter cover (SC = -0.210), flowering shrub abundance (SC = -0.207),

variance of soil density (SC = -0.203), variance of litter height (SC = 0.199), shrub cover (SC = 0.192), variance of forb cover (SC = 0.192), mean soil moisture (SC = 0.184), and mean forb cover (SC = 0.171; $P = 0.026$, $R^2 = 0.273$) were the combination of variables that best predicted Lepidopteran abundance.

Discussion

Lepidopteran species richness and abundance were largely unaffected by treatment type, the exception being dormant season burn sites where declines in abundance were observed. However, results should be interpreted cautiously as the high mobility of adult butterflies may mask management effects on this insect group (Swengel 1998). In prairie habitats, while leaving areas entirely unmanaged rarely benefits Lepidoptera, regular prescribed burning may result in low numbers, particularly for more specialized species (Swengel 1998). Wildfires or less frequent burns that resemble wildfires appear more appropriate in maintaining and increasing Lepidopteran abundance in prairie habitats (Swengel 1998). Infrequent burns that create new habitat patches to be occupied by Lepidoptera during fire-free intervals, rather than repeated fires that maintain existing habitat already occupied, may be most favorable for maintenance of species richness and abundance (New 1991, 1993). In contrast, Lepidopteran abundance is higher in regularly burned compared to control Texas pine forests, with regularly burned sites having a more open mid- and understory (Rudolph and Ely 2000). The effect of season of burn on Lepidoptera has not been widely examined. However, differences in the responses of members of this insect group may depend on whether individuals are in a stage of activity or diapause (Swengel 2001). Burns that occur when Lepidoptera are in diapause may result in greater mortality of immatures and reduced abundance of adults later.

Prescribed burning alone had no effect on flowering forb or shrub species richness and abundance. In Texas forests, increases in flowering plants were observed following regular prescribed burning, providing additional valuable nectar sources and corresponding to greater Lepidoptera abundance (Rudolph and Ely 2000). Platt et al. (1988) and Streng et al. (1993) demonstrated that flowering in the herbaceous plant community of southeastern pine forests increased following burning. The pattern of flowering depended on season of burn, with many forb species including *Liatris*, *Pityopsis*, and *Solidago* spp. flowering more profusely following growing season burning. Similar increases in flowering forb abundance might have been expected on pine flatwoods sites subject to growing season burning. If this had been the case, concurrent increases in Lepidopteran species richness or abundance may have been observed.

Growing season roller chopping resulted in increases in flowering forb species richness and dormant season roller chopping to increases in flowering forb abundance. These practices result in shrub cover, height, and occasionally density reductions (Chapter 2). The opening up of the ground layer and reduced shading of herbaceous vegetation that result may allow for increased flowering. However, there were no corresponding increases in Lepidopteran species richness and abundance on these sites. Flowering shrub species richness and abundance decreased on growing season roller chop and roller chop/burn sites. Lepidopteran species richness and abundance were negatively affected by these habitat characteristics. Therefore, despite there being no increase in Lepidopteran species richness and abundance on roller chopped sites, there is the potential for Lepidoptera to be positively affected by these treatments.

Lepidopteran abundance and species richness were also negatively affected by graminoid and shrub height and cover. These habitat characteristics are often reduced on prescribed burn and roller chopped sites (Chapter 2 and 3), potentially improving habitat for Lepidoptera. However, regression models also suggest that species richness and abundance of these insects is often positively related to forb cover and height and litter depth. These habitat characteristics also decline on prescribed burn and roller chop sites (Chapter 2 and 3), potentially hindering Lepidopteran management efforts.

Management Implications

Dormant season burning may negatively affect Lepidopteran abundance in pine flatwoods. However, until further research examining the response of both immature and adult Lepidoptera to prescribed burning is conducted, the application of all treatments over large areas, in situations where the management of these insects is a priority, should be carefully considered. Examination of only mobile adults may result in treatment effects being masked. Until further research is conducted, application of prescribed burning and roller chopping practices in pine flatwoods where active Lepidopteran management is occurring, should be done on smaller areas in a mosaic arrangement. This will ensure a variety of pine flatwoods habitats are available for occupation by a range of Lepidopteran species.

Table 6-1. Lepidopteran abundance (no. of individuals) in prescribed burned and roller chopped Florida flatwoods, 2008.

Common Name	Scientific Name	Abundance (no. of individuals)									
		Dormant Burn		Growing Burn		Dormant Roller Chop		Growing Roller Chop		Roller Chop/Burn	
		C ^a	T ^b	C	T	C	T	C	T	C	T
Black swallowtail	<i>Papilio polyxenes</i>	3	0	7	4	2	0	3	4	1	3
Cabbage white	<i>Pieris rapae</i>	2	2	1	1	0	3	2	2	3	1
Checkered white	<i>Pontia protodice</i>	0	1	4	4	4	4	0	3	9	6
Cloudless sulphur	<i>Phoebis sennae</i>	2	0	1	0	0	0	1	0	1	1
Common buckeye	<i>Junonia coenia</i>	0	2	0	2	2	1	4	1	4	4
Eastern tiger swallowtail	<i>Papilio glaucus</i>	1	0	0	0	0	0	0	0	0	0
Giant swallowtail	<i>Papilio cresphontes</i>	0	0	0	0	0	0	0	0	3	0
Gray hairstreak	<i>Strymon melinus</i>	0	0	2	1	0	0	0	0	0	0
Gulf fritillary	<i>Agraulis vanillae</i>	0	0	0	1	0	0	0	1	0	0
Least skipper	<i>Ancyloxypha numitor</i>	0	0	2	0	0	1	2	0	0	1
Little metalmark	<i>Calephelis virginiensis</i>	3	0	0	1	0	1	2	4	0	2
Monarch	<i>Danaus plexippus</i>	0	0	0	0	0	0	1	0	0	0
Obscure sphinx	<i>Errinyis obscura</i>	0	0	0	5	0	0	0	0	0	0
Orange sulphur	<i>Colias eurytheme</i>	0	0	0	0	0	0	0	0	1	0
Palmetto skipper	<i>Euphyes arpa</i>	0	0	0	0	0	1	0	0	0	0
Red admiral	<i>Vannesa atalanta</i>	0	0	2	1	0	0	0	0	1	0
Short-lined chocolate	<i>Agnomonia anilis</i>	0	0	0	0	0	0	0	0	1	0
Southern broken dash	<i>Wallengrenia otho</i>	0	0	0	0	0	0	1	0	0	0
Southern skipperling	<i>Copaeodes minimus</i>	2	0	2	1	0	2	0	2	0	1
Zebra swallowtail	<i>Eurytides marcellus</i>	13	7	10	19	5	2	3	3	12	7

^a C = Control.

^b T = Treated.

Table 6-2. Comparison of the effects of treatment on lepidopteran, flowering forb, and flowering shrub species richness and abundance in Florida flatwoods, 2007–2008

Lepidopteran, Flowering Forb, and Flowering Shrub Abundance and Species Richness ^a	Treatment Type ($\bar{x} \pm SE$) ^b						<i>P</i>
	Control	Dormant Burn	Growing Burn	Dormant Roller Chop	Growing Roller Chop	Roller Chop/ Burn	
Abundance (no. of individuals)							
Lepidoptera	2.7 ± 0.4 _A	1.2 ± 0.5 _B	4.3 ± 1.7 _A	1.7 ± 0.3 _A	2.0 ± 0.7 _A	3.0 ± 0.9 _A	0.065
Flowering forb	61.3 ± 14.7 _A	61.1 ± 29.7 _A	15.2 ± 6.2 _A	302.7 ± 159.1 _{AB}	238.4 ± 115.6 _B	322.4 ± 163.9 _B	0.015
Flowering shrub	72.3 ± 16.3 _A	118.3 ± 30.5 _A	71.8 ± 16.2 _A	30.0 ± 13.2 _A	55.8 ± 51.3 _B	10.0 ± 5.0 _B	0.002
Richness (no. of species)							
Flowering forb	3.9 ± 0.4 _A	3.2 ± 0.4 _{AB}	2.7 ± 0.4 _A	6.0 ± 1.2 _{AB}	7.0 ± 0.8 _B	5.8 ± 1.1 _B	0.035
Flowering shrub	3.0 ± 0.2 _A	4.4 ± 0.5 _A	2.9 ± 0.4 _A	3.1 ± 0.5 _A	1.7 ± 0.6 _B	1.6 ± 0.3 _B	0.002

^a Only Lepidopteran, flowering forb, and flowering shrub species richness or abundance significantly affected by treatment presented ($P \leq 0.1$).

^b Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

CHAPTER 7 AVIAN COMMUNITY RESPONSE TO GRAZING INTENSITY ON MONOCULTURE AND MIXED FLORIDA PASTURES

Introduction

There are approximately 1.2 million ha of monoculture and mixed pasture in Florida, 78% of which occurs on private lands and is used primarily for cattle grazing (Florida Fish and Wildlife Conservation Commission [FWC] 2005). Monoculture pastures, also known as non-native or improved pastures, are dominated by non-native forage species and usually comprised of former native pastures that have been cleared, tilled, and reseeded with improved forage types, so that few native vegetative species remain. Mixed pastures, also known as semi-native or semi-improved pastures, are comprised of a mixture of non-native improved forage species interspersed with substantial quantities of native grasses and forbs. Mixed pasture conversion is less intense than for monoculture pastures and management inputs (e.g., fertilizer, weed control) are lower (FWC 2005).

Monoculture and mixed pastures are not a native part of Florida's landscape but, if grazed appropriately, have the potential to provide significant habitat for and be utilized by a diversity of wildlife species including resident and migratory birds (Alsop 2002, Engstrom et al. 2005, FWC 2005). Many of these species, some of which are federally and/or state listed as endangered or threatened, have decreasing populations and are of conservation concern (FWC 2005).

Grazing livestock most typically affect avian communities indirectly through changes in vegetation composition, structure, and biomass, and can cause decreases in spatial heterogeneity of the vegetative community (Brennan and Kuvlesky 2005, Coppedge et al. 2008, Derner et al. 2009). Reductions in spatial heterogeneity caused

by grazing imply the loss of habitat diversity (Adler et al. 2001) and changes in food abundance, foraging conditions, and breeding, thermal, and escape cover (Bock and Webb 1984, Vallentine 1990, Milchunas and Lauenroth 1993, Saab et al. 1995). Such changes have the potential to cause declines in avian species richness of grazed pastures, although avian abundance may be little affected, as species adapted to the grazed conditions often become highly abundant (Kantrud and Kologiski 1982).

Therefore, avian community structure has the potential to be strongly influenced by the degree of structural heterogeneity in associated plant communities (Wiens 1974), with certain birds being attracted to habitats with specific vegetative attributes (Cody 1985).

The majority of research on the effects of grazing on plant communities and bird abundance and species richness has been conducted in the central and western United States (Bock and Webb 1984, Vallentine 1990, Adler et al. 2001, Brennan and Kuvlesky 2005, Coppedge et al. 2008, Derner et al. 2009). However, the transfer of results among environmentally divergent lands should be done cautiously. If management activities of benefit to birds associated with monoculture and mixed pastures are to be promoted in Florida, the impact grazing of these lands has on avian communities needs further investigation. The objectives of this study were: 1) to compare avian species richness and abundance on monoculture and mixed pastures subject to 4 grazing intensities (nongrazed, low, medium, and high) and 2) to explore the role structural habitat attributes play in determining avian species richness and abundance on monoculture and mixed grazed pastures.

Methods

Study Sites

Research was conducted at the MacArthur Agro-Ecology Research Station (MAERC), a 4,170 ha cattle ranch operated by Archbold Biological Station, located in Highlands County, south Florida (27° 09'N, 81° 12'W). One monoculture pasture and 1 mixed pasture study area were selected at MAERC. During 1996–1998, the 162-ha monoculture pasture study area was subdivided using fences into eight, approximately 20-ha experimental pasture units used for summer grazing (May–October). These monoculture pasture units were comprised almost entirely of bahiagrass (*Paspalum notatum* Flügge), but included scattered wetlands, the majority of which were ditched and consisted of grasses, sedges, and miscellaneous wetland species (Werner et al. 1998). Wetland dominants included carpetgrass (*Axonopus furcatus* Flügge Hitchc.), maidencane (*Panicum hemitomon* Schult.), soft rush (*Juncus effuses* L.), yellow-eyed grass (*Xyris* sp.), pickerelweed (*Pontederia cordata* L.), and sawgrass (*Cladium jamaicense* Crantz). A number of the monoculture pastures also contained small cabbage palm (*Sabal palmetto* [Walt.] Lodd. ex J.A. and J.H. Schultes) hammocks (Werner et al. 1998). During 1996–1998, the 260 ha mixed pasture study area was subdivided using fences into eight, approximately 32-ha experimental pasture units used for winter grazing (November–April). These mixed pasture units were comprised of bahiagrass and a variety of native species such as broomsedge (*Andropogon virginicus* L.) and bushy bluestem (*A. glomeratus* [Walt] B.S.P). The mixed pasture units were interspersed with seasonal wetlands, nearly all within 30 m of existing ditches, and comprised of grasses, sedges, and miscellaneous wetland species. Dominants in these wetlands included carpetgrass, maidencane, red top panicum (*P.*

rigidulum Bosc ex Nees), hat pins (*Eriocaulum* sp.), yellow-eyed grass, and some pickerelweed and soft rush. Cabbage palm hammocks occurred in the western third of this mixed pasture unit array. Mixed pastures were not as intensively drained as monoculture pastures and were frequently flooded or saturated during the June to October rainy season (Werner et al. 1998). Swain et al. (2007) provide additional details on pastures. Forage production was higher on summer grazed monoculture pastures than on winter grazed mixed pastures. Therefore, to provide similar amounts of forage in both seasons and accommodate consistent grazing intensities, it was necessary for mixed pasture units to be larger than monoculture pasture units (Capece et al. 2007, McSorley and Tanner 2007).

In 1998, a grazing study was initiated on the monoculture and mixed pasture units. Pasture units were subject to 4 cattle grazing intensities: 0 = nongrazed (control), 15 = low, 20 = medium, or 35 = high animal units (AU) per pasture unit (no cattle, 1.3, 1.0, and 0.6 ha • AU⁻¹ on monoculture pastures and no cattle, 2.1, 1.6, and 0.9 ha • AU⁻¹ on mixed pastures). Stocking densities were selected based on input from the Florida Cattleman's Association and the University of Florida Institute of Food and Agricultural Sciences to reflect typical regional stocking densities, which average 1.42 ha • AU⁻¹ (Gornak and Zhang 1999). Each grazing intensity was replicated twice for each pasture type and randomly assigned to the pasture units. The monoculture pasture units were summer grazed from May–October and mixed pasture units were winter grazed from November–April each year. Cattle were introduced and grazing intensities initiated on mixed pasture units in October 1998 and on monoculture pasture units in May 1999. Monoculture pasture units were not grazed from October 1998 through April 1999. Prior

to the initiation of study grazing treatments, cattle grazed the 2 pasture types during these same seasons at an average stocking density of approximately 1 ha • AU⁻¹ and 1.6 ha • AU⁻¹ on monoculture and mixed pastures, respectively.

Prescribed burning was conducted in all mixed pasture units during November–December 1998 and in all monoculture pastures units during February 1999 with similar affects observed across the study areas. All mixed pasture units were prescribed burned again in February 2002 and monoculture pastures in April 2002. All monoculture pasture units were mowed at a height of 35 cm for general weed control between September and November in 1998, 2000, and 2002. Dog fennel (*Eupatorium capillifolium* Lam.) in the monoculture pasture units was treated with a combination of dimethylamine salts of dicamba and 2,4-D (WEEDMASTER®) at 4.6 L • ha⁻¹ plus 7.5 mL • L⁻¹ of nonionic surfactant from May–July of 2001, 2002, and 2003. More details on pasture management are provided in Swain et al. (2007).

Vegetation Sampling

Vegetation sampling was conducted quarterly during spring (March 15–April 15), summer (June 1–July 1), fall (October 1–November 1), and winter (January 1–February 1) for 4 years (1999–2003). Vegetation sampling methods were similar to those described by Wiens (1969) and utilized a transect system. One 800-m line transect was established in each pasture and divided into 4 sampling units of equal length. Within each sampling unit, I randomly located 4 vegetation sampling sub-points on either side of the transect. Sampling was repeated in each of the transect sampling units for a total of 32 sub-points per transect.

At each subpoint, I visually estimated percent canopy coverage of grasses, forbs, litter, and bare ground to the nearest 5% within a 2.4-m² circular plot (Wiens 1969,

Higgins et al. 2005). I measured vertical stem density in the center of each plot by recording the number of vegetation contacts with a pole at 10 cm height intervals (Wiens 1969). This data was used to calculate vertical stem density 0–30, 30–60, 60–90, and 90–120 cm above ground (stems/30 cm). Litter depth to the nearest 10 cm was measured on the same pole.

Bird Surveys

The avian communities within pastures were sampled using strip transect methods. Two parallel, 50 x 800-m strips were positioned centrally in each of the 16 pasture units by marking the start and end points, as well as each 200-m interval, with 3-m-tall PVC pipe. Strip transects were separated by a 50-m buffer and ≥ 50 m from an adjacent pasture (Wiens and Rotenberry 1981, Eberhardt 1978, Gibbons et al. 1996, Bibby et al. 2000). Vegetation sampling and avian surveys were conducted concurrently. I sampled avian transects quarterly between 1999 and 2003, corresponding to presumed seasonal differences in avian habitat utilization: spring (March 15–April 15) and fall (October 1–November 1) migrations, and breeding (June 1–July 1) and wintering (January 1–February 1) seasons. The June 1–July 1 breeding season selected for this study is the same as that used by the North American Breeding Bird Survey (Peterjohn and Sauer 1993). Each transect was sampled twice per season beginning at sunrise and ending no later than 3.5 h after sunrise. Sampling time per transect was 25–30 min. Sampling was not conducted during excessive wind, rain, fog, or periods of other unusual weather conditions (Gibbons et al. 1996, Bibby et al. 2000). Transects within the same pasture unit were not sampled on the same day to reduce the chance of counting the same bird twice (Wiens and Rotenberry 1981). I alternated sampling order each sampling period to reduce bias of counting the same transect at

the same time of the morning. Only birds recorded within the 50-m strip boundaries were counted (Bibby et al. 2000). I counted flushed birds at the point they were first observed. I used careful observation, including recording of the location of flushed birds, to reduce the likelihood of double counting (Gregory et al. 2004). Average height of the herbaceous vegetation was approximately 75 cm and 150 cm in the summer and winter pasture arrays, respectively, allowing observers consistent sighting ability within the pasture types.

I divided counts of avian abundance and species richness into guilds prior to analyses. Eight guilds were utilized, each falling into 1 of 2 major categories based on breeding habitat or migrant status (Peterjohn and Sauer 1993). Grassland, wetland and open water, successional-scrub, woodland, and urban species guilds comprised the breeding habitat category and short-distance migrant, neo-tropical migrant, and permanent resident species guilds the migrant status category. Not all species fell within a breeding habitat and migrant status guild. Within each category, guilds were independent and did not overlap in species composition (Table 7-1).

Analyses

I performed repeated measures analyses using mixed model regressions, with season and time since introduction of grazing as repeated measures, followed by Fisher's Protected LSD tests, to examine differences in vegetation attributes (mean, variance, and maximum percent coverage of grasses, forbs, bare ground, and litter, litter depth, and vertical stem density), total avian abundance and species richness, and avian abundance and species richness by guild among grazing intensities for monoculture and mixed pastures. I focused on grazing intensity effects rather than repeated measures effects. Two- and three-way grazing intensity interactions were

noted in the results, if they occurred. Because three-way interactions are difficult to reliably interpret, they were not discussed further (Zar 1999, SYSTAT 2007).

On both monoculture and mixed pastures, multiple linear regression was used to examine which combination of vegetation attributes best described changes in avian abundance and species richness, both overall and by guild. To reduce multicollinearity problems, all predictor variables involved in pairwise correlations with $r \geq 0.7$, were subjected to a univariate, one-way analysis of variance (ANOVA) with each dependent variable. For each pair of highly correlated predictor variables, the variable retained was the one with the greatest F value (Noon 1981, McGarigal et al. 2000). All regression models were fit using a forward stepwise procedure with Tolerance = 0.001, F-to-enter = 0.15, and F-to-remove = 0.15. These values are considered appropriate for predictor variables that are relatively independent (SYSTAT 2007). Regression models were considered statistically and biologically significant at $P \leq 0.1$ and $R^2 \geq 0.3$. Only models considered significant are presented. The relative importance of each variable in the best model was assessed by examining standardized regression coefficients (SC; i.e., variables with higher coefficients made greater individual contributions to the explanatory power of the model).

All data sets were rank transformed prior to analyses due to violations of normality and homogeneity of variance assumptions (Zar 1999, Conover 1998, SYSTAT 2007). Statistical significance was concluded at $P \leq 0.1$ for all tests. This value was used rather than the more conservative $P \leq 0.05$, to minimize the probability of making a Type II error (1998). All statistical tests were performed using SYSTAT (2007) statistical software.

Results

Vegetation

Monoculture pasture. Variance of grass cover and litter depth were affected by grazing, decreasing as grazing intensity increased. Grazing also affected mean, variance, and maximum forb cover, mean and variance of litter cover, maximum vegetation height, mean litter depth, and mean, variance, and maximum stem density at 60–90 cm and 90–120 cm. All generally decreased in the presence of grazing compared to control pasture units, but effects at low, medium, and high grazing intensities were similar. Mean grass cover was also affected by grazing, increasing as grazing intensity increased (Table 7-2).

A grazing intensity * season interaction affected mean vegetation height, maximum litter depth, maximum litter cover, mean bare ground cover, variance in stem density at 0–30 cm, and mean, variance, and maximum stem density at 30–60 cm (Table 7-3). During the spring, maximum stem density at 30–60 cm and maximum litter depth decreased in the presence of grazing compared to control pasture units. Mean vegetation height and mean and variance of stem density at 30–60 cm typically decreased in the presence of grazing in winter and spring. Maximum litter cover decreased in the presence of grazing in all seasons. For all of these vegetation attributes, effects at low, medium, and high grazing intensities were similar. In spring, variance of stem density at 0–30 cm decreased in the presence of grazing compared to control pasture units, with reductions similar at low and high and greatest at medium grazing intensities. Mean bare ground cover decreased at high grazing intensity compared to control pasture units in fall, winter, and summer. Depending on season, reductions in this variable were also observed at medium grazing intensity (Table 7-3).

Mean, variance, and maximum stem density at 0–30 cm were affected by a grazing intensity * time interaction (Table 7-4). Decreases in mean stem density at 0–30 cm were typically observed at medium and high grazing intensities compared to control pasture units 1–4 years after introduction of grazing. Similar decreases were seen in variance and maximum stem density at 0–30 cm, but generally only immediately following and up to 1 year after introduction of grazing (Table 7-4). Variance in vegetation height and bare ground cover were affected by a grazing intensity * season * time interaction ($P = 0.019$ and $P = 0.098$, respectively). Grazing alone and grazing intensity * time, grazing intensity * season, and grazing intensity * season * time interactions had no impact on maximum grass or bare ground cover ($P \geq 0.449$).

Mixed pasture. Grazing affected mean and variance of forb cover, which increased at low grazing intensity compared to controls. In contrast, mean grass cover decreased at low grazing intensity. No clear trend in mean and variance of forb cover or mean grass cover was observed with further increases in grazing intensity. Mean litter depth and vegetation height were affected by grazing. Mean litter depth decreased at high grazing intensity and mean vegetation height decreased at medium and high grazing intensities when compared to control pasture units. Grazing affected maximum forb cover, but no clear trend was observed with increasing grazing intensity (Table 7-2).

Mean stem density at 0–30 cm was affected by a grazing intensity * season interaction, decreasing at high grazing intensity in fall and summer (Table 7-3). This vegetation attribute was also affected by a grazing intensity * time interaction, with decreases observed 2, 3, and 5 years after introduction of grazing at high, and

sometimes medium and low, grazing intensities compared to control pasture units (Table 7-4). Grazing intensity alone and grazing intensity * time, grazing intensity * season, and grazing intensity * season * time interactions had no impact on variance and maximum grass cover, variance and maximum vegetation height, variance and maximum litter depth, mean, variance, and maximum litter cover, mean, variance, and maximum bare ground cover, variance and maximum stem density at 0–30 cm, or mean, variance, and maximum stem density at 30–60 cm, 60–90 cm, and 90–120 cm ($P \geq 0.127$).

Avian Abundance and Species Richness

Monoculture pasture. Sixty-nine bird species were observed on monoculture pasture units (Table 7-1). Grazing affected wetland guild abundance ($P = 0.075$), which decreased at low and medium grazing intensities and increased at high grazing intensity compared to control pasture units. Short-distance migrant ($P = 0.028$) and permanent resident ($P \leq 0.001$) guilds were also affected by grazing, exhibiting decreases in abundance at low and high grazing intensities. However, at medium grazing intensity, abundance within these guilds was similar to that observed on control pasture units. Grazing affected neo-tropical migrant guild abundance ($P \leq 0.001$), which decreased at low, medium, and high grazing intensities. Declines were similar at low and medium grazing intensities and greatest at high grazing intensity (Figure 7-1).

Total species richness was affected by grazing ($P \leq 0.001$), decreasing in the presence of grazing compared to control pasture units. Reductions were similar at low and high grazing intensities and greatest at medium grazing intensity. Grazing also affected short distance migrant ($P = 0.001$) and permanent resident ($P \leq 0.001$) guild species richness, which decreased in the presence of grazing. Declines were similar at

low and medium grazing intensities and greatest at high grazing intensity. Species richness within successional scrub ($P \leq 0.001$) and neo-tropical migrant ($P \leq 0.001$) guilds was affected by grazing. Both decreased in the presence of grazing compared to control pasture units, but reductions at low, medium, and high grazing intensities were similar (Figure 7-2).

A grazing intensity * season interaction affected grassland ($P = 0.002$) and woodland ($P = 0.055$) guild abundance and woodland species richness ($P = 0.086$). In the fall, grassland guild abundance increased in the presence of grazing compared to control pasture units. However, increases at low, medium, and high grazing intensities were similar. Woodland guild abundance and species richness decreased in the fall at low grazing intensity, but no differences were observed between control pasture units and those subject to medium and high grazing intensities (Figure 7-3).

Abundance within successional-scrub ($P = 0.053$) and woodland ($P = 0.074$) guilds was affected by a grazing intensity * time interaction. Successional-scrub guild abundance decreased in the presence of grazing compared to control pasture units 3-4 years after the introduction of grazing. However, reductions at low, medium, and high grazing intensities were similar. Woodland guild abundance decreased at low grazing intensity 2 years after the introduction of grazing. However, abundance at medium and high grazing intensities was similar to that observed on control pasture units (Figure 7-4). Grazing intensity alone and grazing intensity * time, grazing intensity * season, and grazing intensity * season * time interactions had no effect on total avian abundance, urban guild abundance, and wetland, grassland, and urban guild species richness ($P \geq 0.209$).

Mixed pasture. Seventy-eight bird species were observed on mixed pasture units (Table 7-1). Grazing affected total avian abundance ($P = 0.017$), which decreased at medium and high grazing intensities. At low grazing intensity, total avian abundance was similar to that observed on control pasture units. Within the grassland guild, abundance was also affected by grazing ($P = 0.045$), increasing at low and medium grazing intensities ($P = 0.045$). However, at high grazing intensity abundance was similar to that observed on control pastures units. Grazing affected urban ($P = 0.046$) and neo-tropical migrant ($P = 0.002$) guild abundance. For both guilds, abundance decreased at high grazing intensity, but no differences were observed between control pastures and those subject to low and medium grazing intensities. Successional scrub abundance was affected by grazing ($P = 0.013$). However, no clear trend in abundance was observed with increasing grazing intensity. Short distance migrant guild abundance was affected by grazing ($P = 0.071$), but was similar at low, medium, and high grazing intensities to that observed on control pasture units (Figure 7-5).

Grazing affected successional-scrub guild richness ($P = 0.009$), which decreased at high grazing intensity. However, no differences in richness were observed between control pastures and those subject to low and medium grazing intensities. Neo-tropical migrant guild abundance was affected by grazing ($P = 0.070$), decreasing at medium and high grazing intensities. However, at low grazing intensity abundance was similar to that observed on control pasture units. Grazing affected urban species richness ($P = 0.027$), which increased at high and low grazing intensities. No differences in urban species richness were observed between control pasture units and those subject to medium grazing intensity. Species richness within the short-distance migrant guild was

affected by grazing intensity ($P = 0.015$). However, no clear trend was observed as grazing increased (Figure 7-6).

A grazing intensity * season interaction affected woodland ($P = 0.048$) and permanent resident ($P = 0.055$) guild abundance. In fall, within the woodland guild, abundance increased at medium grazing intensity, but was similar to control pasture units at low and high grazing intensities. In all seasons, abundance within the permanent resident guild was similar on control pasture units and those subject to low, medium, and high grazing intensities. Total avian species richness ($P = 0.075$) and species richness within the woodland guild ($P = 0.028$) were also affected by a grazing intensity * season interaction. However, in all seasons, total species richness was similar on control pasture units and those subject to low, medium, and high grazing intensities. In fall, species richness within the woodland guild increased at medium grazing intensity, but was similar to control pastures at low and high grazing intensities. Woodland guild richness also increased in spring at high grazing intensity, but was similar on control pasture units and those subject to low and medium grazing intensities (Figure 7-7).

Total species richness was affected by a grazing intensity * time interaction ($P = 0.092$). However, richness was similar at low, medium, and high grazing intensities compared to controls at all times following introduction of grazing (Figure 7-8).

Grassland guild species richness was affected by a grazing intensity * season * time interaction ($P = 0.087$). Grazing intensity alone and grazing intensity * time, grazing intensity * season, and grazing intensity * season * time interactions had no effect on

wetland guild abundance and wetland and permanent resident guild abundance and species richness ($P \geq 0.124$).

Avian-Habitat Relationships

Monoculture pasture. Vegetation attributes that best predicted wetland guild abundance were mean vegetation height (SC = 0.425), minimum bare ground cover (SC = 0.262), variance of litter depth (SC = -0.182), and variance of stem density at 0–30 cm (SC = 0.161; $P \leq 0.001$, R^2 0.387). Mean vegetation height (SC = 0.358), variance of stem density at 0–30 cm (SC = 0.231), variance of litter depth (SC = 0.153), and mean forb cover (SC = 0.143; $P \leq 0.001$, R^2 = 0.368) were the vegetation attributes that best explained neo-tropical migrant guild abundance.

Vegetation attributes that best predicted species richness within the wetland guild were mean stem density 0–30 cm (SC = 0.631), maximum stem density at 60–90 cm (SC = -0.330), mean bare ground cover (SC = 0.282), mean litter cover (SC = -0.201), maximum stem density at 0–30 cm (SC = -0.197), and mean stem density at 90–120 cm (SC = 0.166, $P \leq 0.001$, R^2 = 0.358). The combined effects of variance of stem density at 0–30 cm (SC = 0.641), maximum stem density at 0–30 cm (SC = -0.558), mean vegetation height (SC = -0.558), mean litter depth (SC = 0.476), mean stem density at 0–30 cm (SC = 0.329), mean litter cover (SC = -0.327), and variance of grass cover (SC = 0.166; $P \leq 0.001$, R^2 = 0.359) best predicted grassland guild species richness. Variance of litter depth (SC = 0.354), variance of vegetation height (SC = 0.333), maximum litter cover (SC = -0.218), and mean forb cover (SC = 0.175; $P \leq 0.001$, R^2 = 0.308) were the vegetation attributes that best explained successional-scrub guild species richness. Vegetation attributes that best predicted short distance migrant guild species richness were mean litter depth (SC = 0.422), variance of stem density at

0–30 cm (SC = 0.404), mean vegetation height (SC = -0.366), mean litter cover (SC = -0.334), maximum forb cover (SC = 0.278), and maximum vegetation height (SC = -0.268; $P \leq 0.001$, $R^2 = 0.326$). Mean vegetation height (SC = 0.458), maximum litter depth (SC = 0.185), and mean forb cover (SC = 0.138; $P \leq 0.001$, $R^2 = 0.324$) were the vegetation attributes that best explained neo-tropical migrant guild species richness.

Mixed pasture. Vegetation attributes that best explained successional-scrub guild abundance were maximum stem density at 90–120 cm (SC = 0.413), mean grass cover (SC = -0.403), variance of litter depth (SC = 0.373), mean litter cover (SC = -0.365), variance of grass cover (SC = -0.223), and maximum grass cover (SC = 0.118; $P \leq 0.001$, $R^2 = 0.452$). Vegetation attributes that best predicted neo-tropical migrant guild abundance were maximum litter depth (SC = 0.385), variance of vegetation density at 60–90 cm, mean litter cover (SC = -0.275), variance of forb cover (SC = 0.102), and maximum grass cover (SC = 0.097, $P \leq 0.001$, $R^2 = 0.475$). Neo-tropical migrant guild species richness was best explained by maximum litter depth (SC = 0.320), maximum stem density at 90–120 cm (SC = 0.301), variance of grass cover (SC = 0.196), and maximum litter cover (SC = -0.149; $P \leq 0.001$, $R^2 = 0.306$).

Discussion

Vegetation

Monoculture pasture. The only vegetation attribute that increased as grazing intensity increased on monoculture pasture was mean grass cover. The dominant grass in monoculture pastures was the improved species bahiagrass. Persistence of monoculture pastures and improved grasses, when subject to grazing, is a crucial factor in their sustainability. Bahiagrass is capable of forming a highly persistent sward which tolerates severe defoliation (Beaty et al. 1977, Stanley et al. 1977, Hirata 1993, 2000;

Hirata and Pakiding 2001) and, when grown in regions with warm summers and cool winters, often shows large seasonal variations in herbage mass under grazing (Pakiding and Hirata 2002).

Eighteen of the other vegetation attributes examined exhibited some degree of decline in the presence of grazing. A decrease in the variance of a considerable number of vegetation attributes in the presence of grazing suggests a loss of spatial heterogeneity. Grazing can result in decreased spatial heterogeneity through reductions in plant biomass and cover, and changes in structural conditions (e.g., plant density and height, and litter cover and depth; Vallentine 1990, Milchunas and Lauenroth 1993, Fuhlendorf and Engle 2001, Derner et al. 2009).

Current grazing practices often neglect to recognize the importance of maintaining spatial heterogeneity in plant structure and composition to biodiversity conservation. Livestock have typically been managed for uniform use of vegetation or “management to the middle” with extremes in vegetations structure (e.g., low sparse and high dense) absent (Derner et al. 2009). However, if used appropriately, grazing offers a potentially important tool for conservation management because of its influence on habitat structure and composition (Collins et al. 1998, Adler et al. 2001). Livestock have the potential to be used as ecosystem engineers, altering the heterogeneity of vegetation (Derner et al. 2009). Herbivores naturally exhibit preference for the consumption of certain plants over others (Van Soest 1996). If stocking rates are appropriate and pastures of a sufficient size, this results in differential patterns of use of individual plant species across a pasture (Launchbaugh and Howery 2005). Typically, declines in heterogeneity are only observed at very low or very high intensities of grazing as

respectively, livestock remove almost none or all of the vegetation. At medium grazing intensity, heterogeneity is maintained or increased as livestock selectively alter and remove a greater proportion of the vegetation in certain areas compared to others (Ausden 2007, Derner et al. 2009). I did not observe such a trend on monoculture pasture during this study and further investigation is needed to understand grazing intensity, pasture sizes, and other livestock management activities that might permit maintenance of spatial heterogeneity in vegetation structure and composition on monoculture pastures in Florida. Possible methods proposed for enhancing spatial heterogeneity at the pasture scale include the strategic placement of supplemental feed, implementation of patch burns, and manipulation of water sources to alter vegetation structure in certain locations across the pasture area (Derner et al. 2009).

Mixed pasture. Far fewer vegetation attributes were affected by grazing on mixed than monoculture pastures, and then often only at low or high grazing intensities. Only 3 vegetation attributes (mean grass cover, mean litter depth, and mean vegetation height) exhibited decreases based on grazing intensity alone on this pasture type. Mean and variance of forb cover increased at low grazing intensities. Other studies have shown moderate livestock grazing can result in increased forb cover, abundance, and species richness. These changes, as in this study, are often concomitant with decreases in vegetation height and litter depth (Talbot et al. 1939, Fensham et al. 1991, McNaughton 1993, Hayes and Holl 2003).

No vegetation attributes exhibited a decline in variance on mixed pasture because of grazing, suggesting that spatial heterogeneity of plant structure and composition may have been better maintained than on monoculture pastures. Grazing of native pasture

systems tends to reduce their heterogeneity by favoring the most productive and palatable forage species for domestic cattle (Fuhlendorf and Engle 2001). On mixed pasture, such changes may have been observed at higher grazing intensities and over a longer time period. However, they may not have occurred during the relatively short duration of this study as a result of interannual and seasonal fluctuations in vegetation composition and quality and spatial and temporal patterns in diet selection observed in more complex vegetation (Ash and Smith 1996). Certainly, many of the native bunch grasses present on mixed pasture, such as broomsedge bluestem, chalky bluestem (*Andropogon cappilipes* Nash), and little bluestem (*Schizachyrium scoparium* [Michx.] Nash var. *stoloniferum* [Nash] Wipff), can grow to considerable heights compared to bahiagrass. During the winter, when this pasture type was grazed, these grasses become largely dormant, leaving dry, rank vegetation above ground. This vegetation is largely unpalatable to and not grazed by livestock. These taller grasses in combination with shrubs and lower growing and newly sprouting grasses and forbs may help maintain structural variability in this habitat (Willcox personal observation).

Avian Abundance and Species Richness

Monoculture pasture. Although, grazing had no impact on total avian abundance on monoculture pastures, total species richness decreased as grazing intensity increased. Heavy grazing can reduce overall species richness in grassland ecosystems (Kantrud 1981, Kantrud and Kologiski 1982) as spatial heterogeneity in the plant community is reduced (Derner et al. 2009). Reductions in spatial heterogeneity caused by grazing imply the loss of habitat diversity (Adler et al. 2001), and can influence the suitability and availability of food and cover resources for a variety of avian species (Saab et al. 1995, Brennan and Kuvaesky 2005, Coppedge et al. 2008, Derner et. al.

2009). It has been proposed that declines in grassland birds may, in part, be associated with grazing-driven reductions in vegetation heterogeneity that minimize the heavily disturbed and undisturbed plant communities different species require (Brennan and Kuvlesky 2005). Despite declines in species richness, total avian abundance is often little affected. Although some species are negatively affected by grazing, others respond positively (Saab et al. 1995). Species adapted to the grazed conditions become highly abundant resulting in little change in the total number of birds present (Kantrud and Kologiski 1982).

On monoculture pastures, grazing proved detrimental to many of the avian guilds examined, negatively affecting species richness and abundance. Most notably affected were short distance migrant, neo-tropical migrant, and permanent resident guilds, all of which exhibited a decrease in species richness with increasing grazing intensity. In addition, these guilds also exhibited decreases in abundance as grazing intensity increased. "Management to the middle" places emphasis on the homogenous use of vegetation by grazers. The results of this study indicated grazing of monoculture pasture led to a trend of decreasing heterogeneity for a variety of habitat attributes. Loss of heterogeneity typically results in a lack of suitable habitat for birds that occupy the extremes of the vegetation structure gradient, e.g., low-sparse and high-dense vegetation, many of which are in these guilds (Kantrud and Kologiski 1982, Bollinger and Gavin 1992, Wilkins and Swank 1992, Saab et al. 1995, Guzy and Ritchison 1999, Derner et al. 2009). This results in loss of species richness and, if remaining guild members do not increase in number, decreases in abundance. The use of livestock as ecosystem engineers at the pasture scale has the potential to alter and maintain

vegetation structure, particularly at the extremes of the structure gradient. Grazing management of this type would permit the creation and maintenance of a variety of habitat types and bird species (Derner et al. 2009).

Wetland guild abundance increased with increasing grazing intensity. At high grazing intensity, abundance within this guild was higher than in control pastures. Studies have shown that waterfowl are tolerant of light to medium grazing, although optimal habitat conditions probably occur in the absence of grazing (Kirsch 1969, Kruse and Bowen 1996).

On monoculture pasture, the successional scrub guild did not exhibit declines in abundance until 3-4 years and the woodland guild until 2 years after the introduction of grazing. This suggests that the birds within these guilds were only sensitive to the vegetation effects and decreased heterogeneity of monoculture pasture habitats after prolonged, high intensity grazing. High adult breeding-site fidelity is typical for many migratory birds. These species will often return yearly to the same areas to nest despite declining habitat conditions. After failing to reproduce successfully for a number of years, they may make the decision to move to new breeding habitats (Hass 1998, Hoover 2003) or die, resulting in declines in avian abundance (Greenwood 1980, Greenwood and Harvey 1982, Beheler et al. 2003, Ortega et al. 2006). However, it should be noted that pasture management strives to reduce woody plant dominance, so declines in shrub-associated species are to be expected.

Attributes that were most often identified as positively related to avian abundance and richness within guilds on monoculture pasture were mean vegetation height and cover of forbs and variance of litter depth and stem density 0–30 cm above ground. All

of these attributes exhibited declines of some degree in the presence of grazing, with some declines being seasonal and others affected by time since introduction of grazing. All of these attributes may be important to a variety of bird species due to their role in providing food and cover resources. Cover of forbs, litter, and bare ground are likely to affect seed and invertebrate food availability. Vegetation height and stem density may influence the type and availability of cover present (Saab et al. 1995).

Mixed pasture. Total avian abundance decreased at medium and high grazing intensities but total species richness was not affected until 3 years after introduction of grazing at which point declines were observed at high grazing intensity. Grazing of mixed pastures had a detrimental effect on species richness and abundance within some avian guilds. However, in this pasture type, the number of guilds negatively affected by grazing was fewer than in monoculture pastures. In addition, in mixed pastures, I only observed negative impacts on species richness and abundance at high grazing intensity, compared to monoculture pastures where detrimental effects were frequently observed at low and medium grazing intensities. Species richness and abundance within successional-scrub and neo-tropical migrant guilds only decreased at high grazing intensity. Few vegetation attributes on mixed pasture were affected by grazing and typically only at high grazing intensity. Short-distance migrant and urban guild species richness increased over that of control pastures at medium and high grazing intensities respectively. Grassland guild abundance increased at low and medium grazing intensities. Members of the grassland guild are of particular concern due to recent population declines (Brennan and Kuvaesky 2005). Their declines may be associated with grazing-driven reductions in vegetation heterogeneity and the

suitability and availability of food and cover resources (Saab et al. 1995, Brennan and Kuvalesky 2005, Coppedge et al. 2008, Derner et al. 2009). However, spatial heterogeneity in plant structure and composition was largely maintained on mixed pastures throughout the study. This likely resulted in a diversity of food and cover resources, which helped maintain and increase avian abundance within the grassland guild and species richness within the short-distance migrant and urban guilds (Saab et al. 1995, Coppedge et al. 2008, Derner et al. 2009). This study suggests that, on mixed pastures, management and conservation of species within the grassland guild may be compatible with low to medium grazing intensities, and that livestock have the potential to serve as ecosystem engineers for members of this and other guilds (Derner et al. 2009).

Attributes that were most often identified as positively related to avian abundance and richness on mixed pasture were mean and maximum grass cover and maximum litter depth and vegetation density 90–120 cm above ground. Mean grass cover was the only one of these attributes to decrease in the presence of grazing and may be important to birds as a food and cover resource. Within many guilds, abundance declined as maximum litter depth increased. Therefore, methods that reduce litter present on the ground may benefit many species.

Management Implications

On monoculture and mixed pasture, increasing grazing intensity resulted in changes in a variety of vegetation attributes. There was a trend toward increasing homogeneity of plant structure and composition as grazing intensity increased, particularly on monoculture pasture, and, depending on guild, this resulted in increases or decreases in abundance and richness within particular avian guilds. If the

management and conservation of certain avian guilds is a priority, grazing intensity should be tailored to fit their needs. On monoculture and mixed pasture, the minimization of grazing intensity would be advantageous and likely result in increased abundance and species richness of many guilds. Based on the results of this study, a grazing intensity of 1.3 and 2.1 ha • AU⁻¹ on monoculture and mixed pasture, respectively is recommended. However, some decline in species richness may still be expected. Ultimately, if habitat diversity is to be maximized and a range of avian species supported on monoculture and mixed pastures, the goal should be to maintain spatial heterogeneity in plant structure and composition, potentially using livestock as ecosystem engineers.

Table 7-1. Avian guild composition and seasonal abundance on monoculture and mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003.

Guild ^{a,b}	Common Name	Scientific Name	Abundance (no. of individuals)							
			Fall		Winter		Spring		Summer	
			Mono ^c	Mixed ^d	Mono	Mixed	Mono	Mixed	Mono	Mixed
WT	American bittern	<i>Botaurus lentiginosus</i>	6	9	1	1	0	0	0	0
WD, SD	American crow	<i>Corvus brachyrhynchos</i>	50	32	63	53	23	32	22	13
SS, SD	American goldfinch	<i>Carduelis tristis</i>	0	0	31	15	57	28	0	0
SD	American kestrel	<i>Falco sparverius</i>	16	11	14	20	21	12	0	0
UB, SD	American robin	<i>Turdus migratorius</i>	0	0	96	122	28	26	0	0
WT	Anhinga	<i>Anhinga anhinga</i>	0	1	2	0	0	0	0	0
WT, SD	Bald eagle	<i>Haliaeetus leucocapillus</i>	0	0	0	1	0	0	0	0
GR, SD	Barn owl	<i>Tyto alba</i>	0	1	0	0	0	0	0	0
NM	Barn swallow	<i>Hirundo rustica</i>	83	52	0	0	4	0	0	0
WD, RE	Barred owl	<i>Strix varia</i>	0	2	0	2	0	0	0	0
WT	Belted kingfisher	<i>Ceryle alcyon</i>	1	0	5	0	0	0	0	0
RE	Black vulture	<i>Coragyps atratus</i>	4	7	26	19	8	22	0	0
WT	Black-crowned nt.-heron	<i>Nycticorax nycticorax</i>	5	1	0	0	0	0	0	0
WT	Black-winged teal	<i>Anas discors</i>	1	0	0	0	0	0	0	0
UB, SD	Blue jay	<i>Cyanocitta cristata</i>	0	3	0	0	0	1	0	0
WD, NM	Blue-gray gnatcatcher	<i>Poliophtila caerulea</i>	12	20	6	8	0	0	0	0
WT, RE	Boat-tailed grackle	<i>Quiscalus major</i>	130	191	28	30	36	12	30	36
GR, NM	Bobolink	<i>Dolichonyx garrulus</i>	64	40	0	0	0	0	0	0
SD	Brown-headed cowbird	<i>Molothrus ater</i>	0	0	2	0	0	0	0	1
GR, NM	Burrowing owl	<i>Athene cunicularia</i>	0	0	0	0	1	0	1	0
SS, RE	Carolina wren	<i>Thryothorus ludovicianus</i>	4	3	0	0	3	0	0	0
WT	Cattle egret	<i>Bubulcus ibis</i>	1063	249	23	6	25	3	237	25
UB, SD	Common grackle	<i>Quiscalus quiscula</i>	5	287	0	4	14	7	34	19
SS, RE	Common ground dove	<i>Columbina passerine</i>	1	2	1	0	0	2	7	11
WT	Common moorhen	<i>Gallinula chloropus</i>	0	1	0	0	0	0	1	0
WT	Common snipe	<i>Gallinago gallinago</i>	146	27	147	88	26	20	0	26
SS, NM	Common yellowthroat	<i>Geothlypis trichas</i>	226	499	46	54	27	71	18	72

Table 7-1. Continued

Guild ^{a,b}	Common Name	Scientific Name	Abundance (no. of individuals)							
			Fall		Winter		Spring		Summer	
			Mono ^c	Mixed ^d	Mono	Mixed	Mono	Mixed	Mono	Mixed
RE	Crested caracara	<i>Caracara cheriway</i>	7	5	5	5	5	15	1	4
WT	Double-crest. cormorant	<i>Phalacrocorax auritus</i>	0	1	1	0	2	0	0	2
NM	Eastern kingbird	<i>Tyrannus tyrannus</i>	0	0	0	0	0	0	0	7
GR, SD	Eastern meadowlark	<i>Sturnella magna</i>	7141	568	394	265	723	465	648	606
SD	Eastern phoebe	<i>Sayornis phoebe</i>	17	15	27	11	1	2	0	0
SS	Eastern towhee	<i>Pipilo erythrophthalmus</i>	0	1	0	0	0	0	0	0
UB, SD	European starling	<i>Sturnus vulgaris</i>	0	4	8	0	0	8	0	0
WT,	Glossy ibis	<i>Plegadis falcinellus</i>	3	1	37	8	0	0	0	0
GR, NM	Grasshopper sparrow	<i>Ammodramus savannarum</i>	0	1	0	5	2	19	0	0
SS, NM	Gray catbird	<i>Dumetella carolinensis</i>	1	4	0	0	0	0	0	0
WT	Great blue heron	<i>Ardea herodias</i>	3	2	3	7	3	1	0	3
WT	Great egret	<i>Ardea alba</i>	43	22	28	7	11	0	33	11
WT, NM	Greater yellowlegs	<i>Tringa melanoleuca</i>	3	0	3	3	19	18	0	19
WT	Green heron	<i>Butorides virescens</i>	4	0	0	0	1	0	3	1
SS, NM	House wren	<i>Troglodytes aedon</i>	3	28	5	37	5	18	0	0
SS, NM	Indigo bunting	<i>Passerina cyanea</i>	0	1	0	0	0	0	0	0
SD	Killdeer	<i>Charadrius vociferous</i>	6	0	16	18	12	9	0	4
WT	King rail	<i>Rallus elegans</i>	10	5	0	0	0	0	0	0
GR, SD	Le Conte's sparrow	<i>Ammodramus leconteii</i>	0	0	0	11	1	0	0	0
WT, NM	Least sandpiper	<i>Calidris minutilla</i>	3	0	0	0	0	3	0	0
WT, NM	Lesser yellowlegs	<i>Tringa flavipes</i>	2	0	0	0	0	0	0	0
WT	Little blue heron	<i>Egretta garzetta</i>	38	10	5	2	2	1	9	2
SD	Loggerhead shrike	<i>Lanius ludovicianus</i>	9	19	3	7	0	4	3	6
WT, SD	Marsh wren	<i>Cistothorus palustris</i>	9	25	1	1	0	0	0	0
WD, NM	Merlin	<i>Falco columbarius</i>	2	0	0	0	0	0	0	0
WT	Mottled duck	<i>Anas fulvigula</i>	58	63	28	18	26	70	19	26
UB, SD	Mourning dove	<i>Zenaida macroura</i>	7	12	2	14	18	25	27	32
SS, RE	Northern bobwhite	<i>Colinus virginianus</i>	158	17	60	6	142	29	145	35
SS, RE	Northern cardinal	<i>Cardinalis cardinalis</i>	5	7	2	5	1	5	3	5

Table 7-1. Continued

Guild ^{a,b}	Common Name	Scientific Name	Abundance (no. of individuals)							
			Fall		Winter		Spring		Summer	
			Mono ^c	Mixed ^d	Mono	Mixed	Mono	Mixed	Mono	Mixed
GR, SD	Northern harrier	<i>Circus cyaneus</i>	0	3	7	20	4	11	0	0
UB, RE	Northern mockingbird	<i>Mimus polyglottos</i>	4	18	10	3	2	16	6	8
WT, SD	Osprey	<i>Pandion haliaetus</i>	0	0	0	0	1	0	0	1
SS	Palm warbler	<i>Dendroica palmarum</i>	264	310	341	152	94	72	0	0
WD, RE	Pileated woodpecker	<i>Dryocopus pileatus</i>	0	1	0	1	0	1	0	0
WD, RE	Red-bellied woodpecker	<i>Melanerpes carolinus</i>	8	10	2	18	0	15	12	7
WD, SD	Red-shouldered hawk	<i>Buteo lineatus</i>	24	12	9	9	11	16	19	10
WT, SD	Red-winged blackbird	<i>Agelaius phoeniceus</i>	2156	1672	671	213	490	523	711	490
NM	Rough-winged swallow	<i>Stelgidopteryx serripennis</i>	0	0	0	0	0	0	2	0
WT	Sandhill crane	<i>Grus canadensis</i>	1	6	44	19	15	13	2	15
GR, SD	Savannah sparrow	<i>Passerculus sandwichensis</i>	126	142	359	654	423	393	1	0
GR, SD	Sedge wren	<i>Cistothorus plantensis</i>	32	232	152	301	93	192	0	0
WT,	Snowy egret	<i>Egretta thula</i>	3	1	10	1	1	0	0	0
NM	Solitary sandpiper	<i>Tringa solitaria</i>	0	0	2	0	0	0	0	0
SS, SD	Song sparrow	<i>Melospiza melodia</i>	0	0	0	0	2	0	0	0
WT	Sora	<i>Porzana carolina</i>	9	2	3	0	0	0	0	0
SD	Swamp sparrow	<i>Melospiza georgiana</i>	1	12	70	727	14	124	0	0
WD	Swallow-tailed kite	<i>Elanoides forficatus</i>	0	0	0	0	0	1	0	0
SD	Tree swallow	<i>Tachycineta bicolor</i>	68	144	631	379	349	198	0	0
WT	Tricolored heron	<i>Egretta tricolor</i>	14	1	4	0	0	0	0	0
SD	Turkey vulture	<i>Cathartes aura</i>	2	40	12	29	2	9	0	5
WT	Virginia rail	<i>Rallus limicola</i>	3	2	0	0	0	0	0	0
WT	White ibis	<i>Endocimus albus</i>	958	119	61	2	0	0	110	0
RE	White-tailed kite	<i>Elanus leucurus</i>	0	0	0	4	0	0	0	0
WT	Wood stork	<i>Mycteria americana</i>	71	23	19	11	3	2	0	3
WD	Yellow-rumped warbler	<i>Dendroica coronate</i>	0	0	2	15	0	2	0	0
WD	Yellow-throated warbler	<i>Dendroica dominica</i>	0	0	0	1	0	0	0	0

^a Breeding habitat guilds: WT = wetland, GR = grassland, SS = successional-scrub, WD = woodland, and UB = urban.

^b Migrant Status guilds: RE = resident, SM = short-distance migrant, and NM = neo-tropical migrant.

^c Monoculture pasture.

^d Mixed pasture.

Table 7-2. Effects of grazing intensity on vegetation attributes of monoculture and mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003.

Pasture Type ^a	Vegetation Attributes ^b	Cattle Grazing Intensity ($\bar{x} \pm SE$) ^c				P
		Control ^d	Low ^e	Medium ^f	High ^g	
Mono	Mean grass cover (%)	78.9 ± 1.7 _A	90 ± 0.8 _B	86.9 ± 1.1 _B	90.0 ± 0.9 _B	≤0.001
	Variance of grass cover (%)	734.7 ± 62.9 _A	400.0 ± 42.1 _B	533.8 ± 1.0 _{AC}	482.0 ± 57.2 _{BC}	0.003
	Mean forb cover (%)	38.0 ± 20.7 _A	6.5 ± 0.6 _B	9.0 ± 1.0 _C	6.6 ± 0.9 _B	≤0.001
	Variance of forb cover (%)	552.3 ± 63.3 _A	226.2 ± 28.0 _B	312.3 ± 36.2 _B	285.7 ± 46.1 _B	0.001
	Maximum forb cover (%)	85.9 ± 3.4 _A	65.9 ± 4.4 _B	74.0 ± 3.8 _B	70.3 ± 4.0 _B	0.003
	Maximum vegetation height (cm)	53.4 ± 6.4 _A	30.4 ± 3.0 _B	34.6 ± 5.6 _B	30.8 ± 4.8 _B	≤0.001
	Mean litter depth (cm)	3.3 ± 0.6 _A	2.3 ± 0.5 _{BC}	2.1 ± 0.4 _B	2.3 ± 0.6 _C	≤0.001
	Variance of litter depth (cm)	13.2 ± 4.3 _A	4.4 ± 1.5 _B	8.2 ± 5.8 _C	3.2 ± 1.2 _C	≤0.001
	Mean litter cover (%)	34.5 ± 5.2 _A	31.2 ± 5.4 _B	29.6 ± 5.3 _B	24.8 ± 5.4 _B	0.001
	Variance of litter cover (%)	343.0 ± 60.9 _A	255.2 ± 48.6 _{BC}	300.0 ± 69.4 _B	177.0 ± 48.1 _C	0.001
	Mean stem density 60–90 cm (stems/30 cm)	0.2 ± 0.1 _A	0.03 ± 0.01 _B	0.04 ± 0.2 _B	0.03 ± 0.01 _{AB}	0.006
	Variance of stem density 60–90 cm (stems/30 cm)	0.8 ± 0.3 _A	0.1 ± 0.03 _B	0.2 ± 0.1 _B	0.1 ± 0.1 _{AB}	0.007
	Maximum stem density 60–90 cm (stems/30 cm)	3.4 ± 0.7 _A	0.9 ± 0.3 _B	1.0 ± 0.4 _B	2.3 ± 1.2 _{AB}	0.009
	Mean stem density 90–120 cm (stems/30 cm)	0.1 ± 0.0 _A	0.0 ± 0.0 _B	0.0 ± 0.0 _B	0.0 ± 0.0 _B	0.001
	Variance of stem density 90–120 cm (stems/30 cm)	0.3 ± 0.1 _A	0.0 ± 0.0 _B	0.1 ± 0.0 _B	0.1 ± 0.1 _B	0.001
Maximum stem density 90–120 cm (stems/ 30 cm)	1.8 ± 0.5 _A	0.4 ± 0.2 _B	0.4 ± 0.2 _B	0.7 ± 0.4 _B	0.001	
Mixed	Mean grass cover (%)	90.8 ± 10.4 _A	85.7 ± 1.9 _B	88.2 ± 2.4 _{AB}	90.7 ± 2.2 _{AB}	0.033
	Mean forb cover (%)	4.16 ± 0.5 _A	7.8 ± 1.0 _B	5.5 ± 0.7 _{ABC}	3.8 ± 0.6 _{AC}	0.011
	Variance of forb cover (%)	111.5 ± 16.2 _A	248.8 ± 38.1 _B	157.9 ± 24.3 _{ABC}	99.2 ± 0.023 _{AC}	0.023
	Maximum forb cover (%)	50.3 ± 4.7 _{AB}	62.3 ± 4.9 _A	57.9 ± 4.9 _A	41.1 ± 5.2 _B	0.043
	Mean vegetation height (cm)	23.5 ± 2.9 _A	21.6 ± 2.8 _{AB}	20.7 ± 2.5 _B	19.3 ± 2.1 _B	0.029
Mean litter depth (cm)	4.0 ± 0.1 _A	3.2 ± 0.7 _{AB}	2.7 ± 0.5 _{AB}	2.3 ± 0.4 _B	0.020	

^a Pasture type: Mono = monoculture pasture, Mixed = mixed pasture.

^b Only vegetation attributes significantly affected by grazing presented ($P \leq 0.1$)

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

^d Nongrazed.

^e 1.3 ha • AU⁻¹ on monoculture pasture and 2.1 ha • AU⁻¹ on mixed pasture.

^f 1.0 ha • AU⁻¹ on monoculture pasture and 1.6 ha • AU⁻¹ on mixed pasture.

^g 0.6 ha • AU⁻¹ on monoculture pasture and 0.9 ha • AU⁻¹ on mixed pasture.

Table 7-3. Effects of a grazing intensity * season interaction on vegetation attributes of monoculture and mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003.

Pasture Type ^a	Vegetation Attributes ^b	Season	Cattle Grazing Intensity($\bar{x} \pm SE$) ^c				P
			Control ^d	Low ^e	Medium ^f	High ^g	
Mono	Mean vegetation height (cm)	Fall	26.3 ± 4.4 _A	19.6 ± 3.2 _A	19.1 ± 3.9 _A	17.6 ± 3.2 _A	0.052
		Winter	14.0 ± 0.8 _A	11.6 ± 0.3 _A	10.4 ± 0.3 _B	10.2 ± 0.2 _B	
		Spring	15.0 ± 1.4 _A	10.8 ± 0.6 _B	10.9 ± 0.6 _B	10.5 ± 0.4 _B	
		Summer	24.4 ± 5.4 _A	16.4 ± 3.3 _A	14.7 ± 3.3 _A	11.5 ± 2.2 _A	
	Maximum litter depth (%)	Fall	12.9 ± 3.4 _A	59.5 ± 10.7 _A	18.6 ± 9.8 _A	8.4 ± 3.4 _A	0.084
		Winter	11.2 ± 4.0 _A	11.3 ± 5.0 _A	5.7 ± 1.4 _A	5.7 ± 1.3 _A	
		Spring	13.3 ± 3.7 _A	4.8 ± 1.3 _B	6.0 ± 1.4 _B	4.3 ± 1.3 _A	
		Summer	15.0 ± 8.1 _A	5.4 ± 1.9 _A	4.5 ± 1.8 _A	6.3 ± 3.8 _A	
	Maximum litter cover (%)	Fall	80.0 ± 6.7 _A	59.5 ± 10.7 _B	56.5 ± 11.5 _A	46.0 ± 10.5 _B	0.067
		Winter	84.2 ± 11.0 _A	75.0 ± 13.1 _B	71.7 ± 13.5 _A	80.0 ± 11.3 _B	
		Spring	73.6 ± 11.0 _A	56.9 ± 12.3 _B	61.9 ± 12.3 _A	53.1 ± 9.5 _B	
		Summer	72.0 ± 9.8 _A	70.5 ± 8.0 _B	69.0 ± 9.1 _A	54.0 ± 8.7 _B	
	Mean bare ground cover (%)	Fall	5.6 ± 1.4 _A	3.12 ± 1.0 _A	6.4 ± 1.3 _A	5.3 ± 1.2 _B	0.064
		Winter	2.7 ± 0.8 _A	2.9 ± 1.0 _A	3.1 ± 0.8 _{AB}	3.4 ± 1.0 _B	
		Spring	2.6 ± 0.9 _A	2.6 ± 0.7 _A	1.5 ± 0.3 _A	2.0 ± 0.8 _A	
		Summer	3.4 ± 2.1 _A	4.4 ± 1.7 _A	3.8 ± 1.0 _A	2.5 ± 0.8 _B	
	Variance of stem density 0–30 cm (stems/30 cm)	Fall	102.7 ± 34.3 _A	46.6 ± 6.9 _A	51.7 ± 7.0 _B	62.5 ± 22.6 _{AB}	0.005
		Winter	68.1 ± 22.3 _A	43.6 ± 9.0 _A	29.0 ± 6.2 _A	25.6 ± 7.3 _A	
		Spring	71.6 ± 23.6 _A	19.5 ± 3.1 _B	18.6 ± 3.8 _C	20.0 ± 2.7 _{BC}	
		Summer	40.1 ± 11.5 _A	23.1 ± 8.4 _A	25.6 ± 8.4 _A	19.5 ± 10.1 _A	
	Mean stem density 30–60 cm (stems/30 cm)	Fall	2.2 ± 0.6 _A	1.2 ± 0.5 _A	1.0 ± 0.4 _A	1.0 ± 0.4 _A	0.015
		Winter	0.2 ± 0.1 _A	0.0 ± 0.0 _{AB}	0.0 ± 0.0 _B	0.1 ± 0.1 _{AB}	
		Spring	0.5 ± 0.2 _A	0.0 ± 0.0 _B	0.0 ± 0.0 _B	0.1 ± 0.0 _B	
		Summer	1.4 ± 0.1 _A	1.0 ± 0.7 _A	0.7 ± 0.4 _A	0.4 ± 0.3 _A	
	Variance of stem density 30–60 cm (stems/30 cm)	Fall	15.2 ± 6.0 _A	4.3 ± 1.8 _A	4.4 ± 2.1 _A	4.1 ± 1.8 _A	0.031
		Winter	0.7 ± 0.2 _A	0.1 ± 0.1 _{AB}	0.0 ± 0.0 _B	0.7 ± 0.6 _{AB}	
		Spring	2.5 ± 1.3 _A	0.0 ± 0.0 _B	0.2 ± 0.1 _B	0.2 ± 0.1 _B	
		Summer	4.5 ± 2.3 _A	3.9 ± 3.1 _A	3.0 ± 2.0 _A	1.5 ± 1.1 _A	

Table 7-3. Continued

Pasture Type ^a	Vegetation Attributes ^b	Season	Cattle Grazing Intensity ($\bar{x} \pm SE$) ^c				<i>P</i>
			Control ^d	Low ^e	Medium ^f	High ^g	
Mono	Maximum stem density 30–60 cm (stems/30 cm)	Fall	15.6 ± 4.2 _A	8.2 ± 2.0 _A	8.6 ± 2.4 _A	8.1 ± 1.6 _A	0.052
		Winter	4.5 ± 1.0 _A	2.0 ± 1.0 _A	0.3 ± 0.2 _A	3.7 ± 2.2 _A	
		Spring	7.6 ± 2.6 _A	0.4 ± 0.3 _{AB}	1.6 ± 1.0 _B	2.4 ± 0.9 _{AB}	
		Summer	26.6 ± 19.4 _A	4.0 ± 2.2 _A	4.4 ± 2.4 _A	2.8 ± 1.5 _B	
Mixed	Mean vegetation density 0–30 cm (stems/30 cm)	Fall	18.9 ± 3.4 _A	15.7 ± 2.1 _A	18.7 ± 2.8 _A	18.9 ± 2.7 _B	0.012
		Winter	15.0 ± 2.5 _A	13.5 ± 2.4 _A	10.1 ± 2.1 _A	9.9 ± 2.2 _A	
		Spring	13.5 ± 2.1 _A	8.0 ± 1.5 _A	9.9 ± 2.0 _A	8.1 ± 1.5 _A	
		Summer	9.6 ± 1.3 _A	8.0 ± 1.0 _A	8.8 ± 1.0 _A	8.4 ± 1.5 _B	

^a Pasture type: Mono = monoculture pasture, Mixed = mixed pasture.

^b Only vegetation attributes significantly affected by a grazing * season interaction presented ($P \leq 0.1$).

^c Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

^d Nongrazed.

^e 1.3 ha • AU⁻¹ on monoculture pasture and 2.1 ha • AU⁻¹ on mixed pasture.

^f 1.0 ha • AU⁻¹ on monoculture pasture and 1.6 ha • AU⁻¹ on mixed pasture.

^g 0.6 ha • AU⁻¹ on monoculture pasture and 0.9 ha • AU⁻¹ on mixed pasture.

Table 7-4. Effects of a grazing intensity * time interaction on vegetation attributes of monoculture and mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003.

Pasture Type ^a	Vegetation Attributes ^b	Time ^c	Cattle Grazing Intensity($\bar{x} \pm SE$) ^d				P
			Control ^e	Low ^f	Medium ^g	High ^h	
Mono	Mean stem density 0–30 cm (stems/30 cm)	0	22.3 ± 5.4 _A	10.8 ± 3.4 _A	10.3 ± 4.0 _A	9.9 ± 4.6 _A	≤0.001
		1	13.5 ± 1.6 _A	8.9 ± 1.0 _{ABC}	7.2 ± 0.7 _B	40.3 ± 7.2 _C	
		2	7.4 ± 1.8 _A	5.9 ± 1.6 _A	5.0 ± 1.4 _A	4.4 ± 1.3 _B	
		3	11.0 ± 1.9 _A	8.9 ± 1.4 _{AB}	8.0 ± 1.2 _B	7.0 ± 1.3 _B	
		4	11.6 ± 0.4 _A	12.0 ± 1.0 _A	14.5 ± 1.5 _B	11.9 ± 1.0 _A	
	Variance of stem density 0–30 cm (stems/30 cm)	0	179.5 ± 74.2 _A	47.1 ± 22.7 _B	43.1 ± 23.9 _B	84.0 ± 61.4 _B	0.005
		1	97.7 ± 20.7 _A	49.1 ± 8.5 _{AB}	40.3 ± 7.2 _A	40.3 ± 7.2 _B	
		2	28.4 ± 7.7 _A	20.6 ± 6.3 _A	15.5 ± 5.1 _A	13.9 ± 4.6 _A	
		3	54.6 ± 15.0 _A	27.1 ± 3.2 _A	28.5 ± 4.6 _A	20.7 ± 4.0 _A	
		4	40.8 ± 9.6 _A	25.3 ± 5.1 _A	41.4 ± 8.8 _A	33.5 ± 5.9 _A	
	Maximum stem density 0–30 cm (stems/30 cm)	0	45.5 ± 5.1 _A	26.3 ± 7.7 _B	23.5 ± 9.5 _B	26.5 ± 12.5 _B	0.013
		1	38.5 ± 3.7 _{AB}	27.4 ± 2.5 _B	40.3 ± 7.2 _A	26.0 ± 3.0 _B	
2		19.3 ± 4.4 _A	16.9 ± 4.2 _A	28.1 ± 14.0 _A	12.8 ± 3.3 _A		
3		35.4 ± 8.4 _A	24.0 ± 1.6 _A	24.3 ± 2.5 _A	20.8 ± 2.4 _A		
4		32.8 ± 6.0 _A	23.5 ± 2.4 _A	33.5 ± 3.1 _B	27.7 ± 2.5 _A		
Mixed	Mean stem density 0–30 cm (stems/30 cm)	0	45.0 ± 0.0 _A	4.9 ± 0.0 _B	38.8 ± 0.0 _A	42.4 ± 0.0 _A	≤0.001
		1	15.5 ± 3.7 _A	15.2 ± 3.3 _A	14.7 ± 4.0 _A	13.1 ± 3.3 _A	
		2	15.3 ± 2.8 _A	12.8 ± 3.1 _{AB}	14.6 ± 2.3 _B	14.4 ± 1.9 _B	
		3	13.6 ± 0.9 _A	9.3 ± 2.4 _B	9.4 ± 1.0 _B	8.2 ± 1.0 _B	
		4	11.9 ± 2.4 _A	9.3 ± 2.4 _A	9.4 ± 2.2 _A	8.1 ± 2.5 _A	
		5	11.2 ± 2.8 _A	10.3 ± 1.2 _A	10.1 ± 1.0 _A	11.4 ± 1.3 _B	

^a Pasture type: Mono = monoculture pasture, Mixed = mixed pasture.

^b Only vegetation attributes significantly affected by a grazing * time interaction presented ($P \leq 0.1$).

^c Time since introduction of grazing (years).

^d Means in a row followed by the same uppercase letter not significantly different ($P > 0.1$).

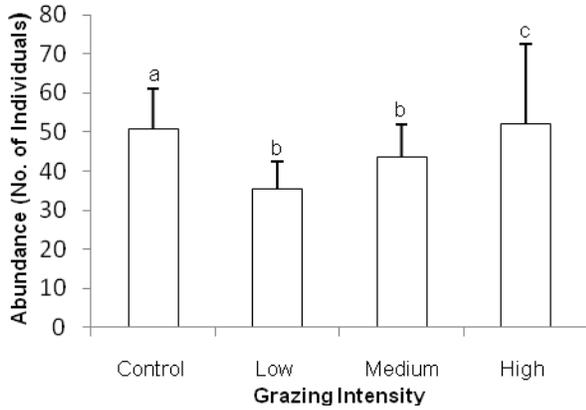
^e Nongrazed.

^f 1.3 ha • AU⁻¹ on monoculture pasture and 2.1 ha • AU⁻¹ on mixed pasture.

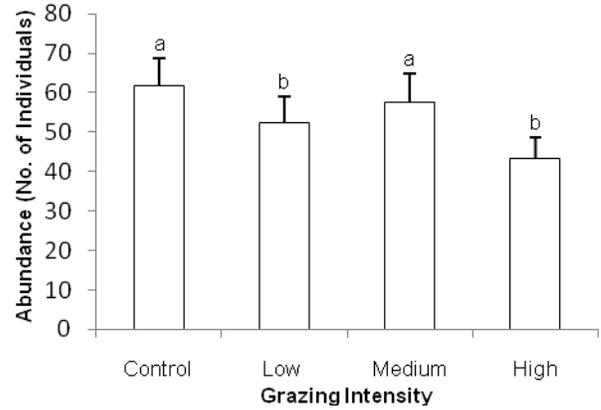
^g 1.0 ha • AU⁻¹ on monoculture pasture and 1.6 ha • AU⁻¹ on mixed pasture.

^h 0.6 ha • AU⁻¹ on monoculture pasture and 0.9 ha • AU⁻¹ on mixed pasture.

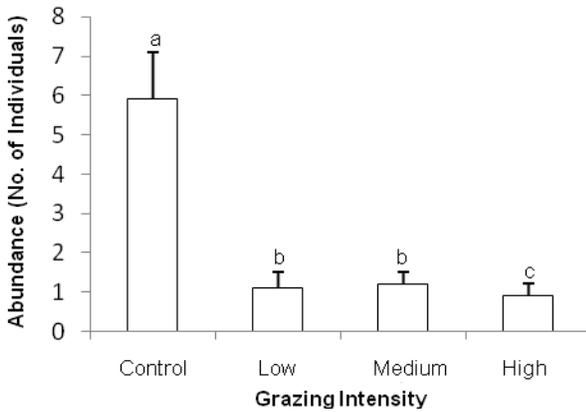
Wetland Guild



Short-Distance Migrant Guild



Neo-Tropical Migrant Guild



Permanent Resident Guild

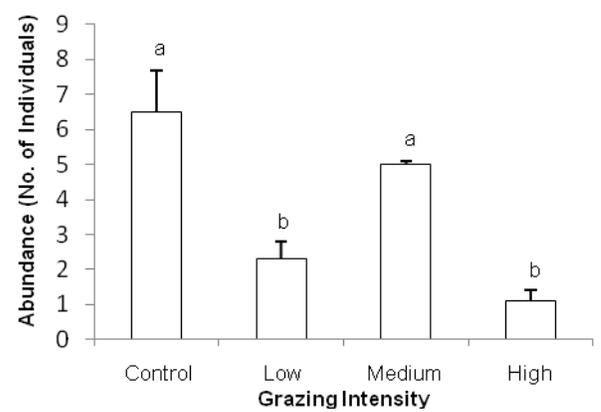
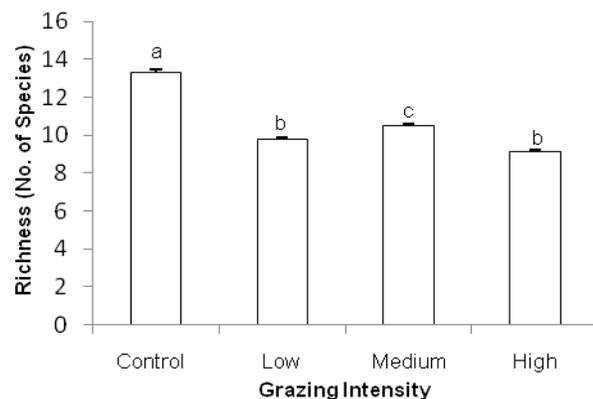
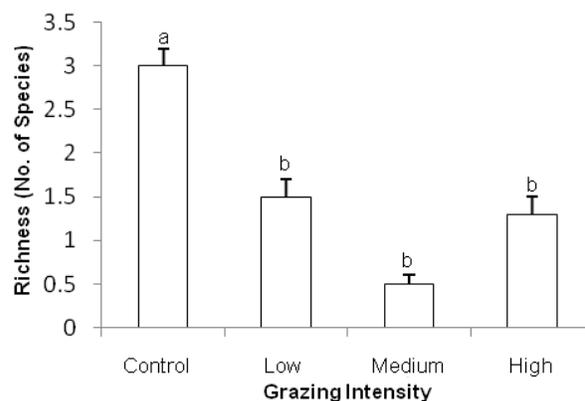


Figure 7-1. Effects of grazing intensity on avian abundance by guild in monoculture pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present $\bar{x} \pm SE$. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $1.3 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.0 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.6 \text{ ha} \cdot \text{AU}^{-1}$.

Total Avian



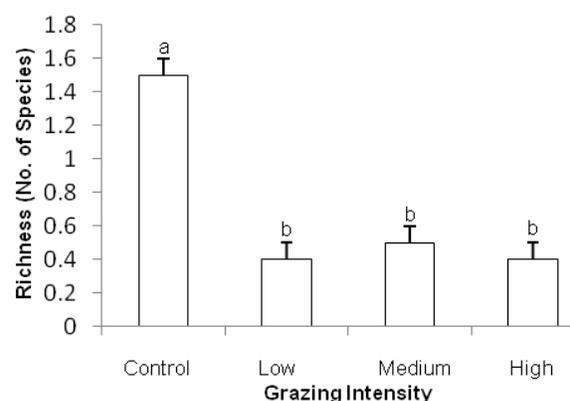
Successional-Scrub Guild



Short-Distance Migrant Guild



Neo-Tropical Migrant Guild



Permanent Resident Guild

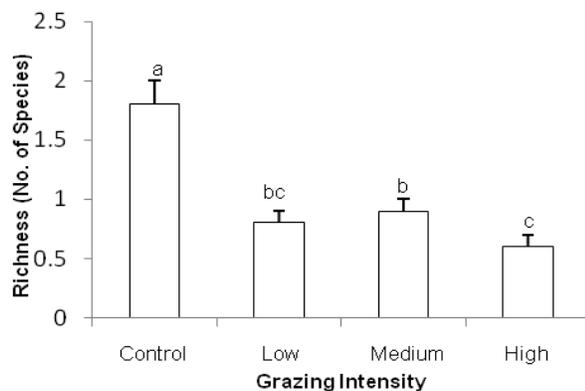


Figure 7-2. Effects of grazing intensity on total avian species richness and avian species richness by guild in monoculture pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present $\bar{x} + SE$. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $1.3 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.0 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.6 \text{ ha} \cdot \text{AU}^{-1}$.

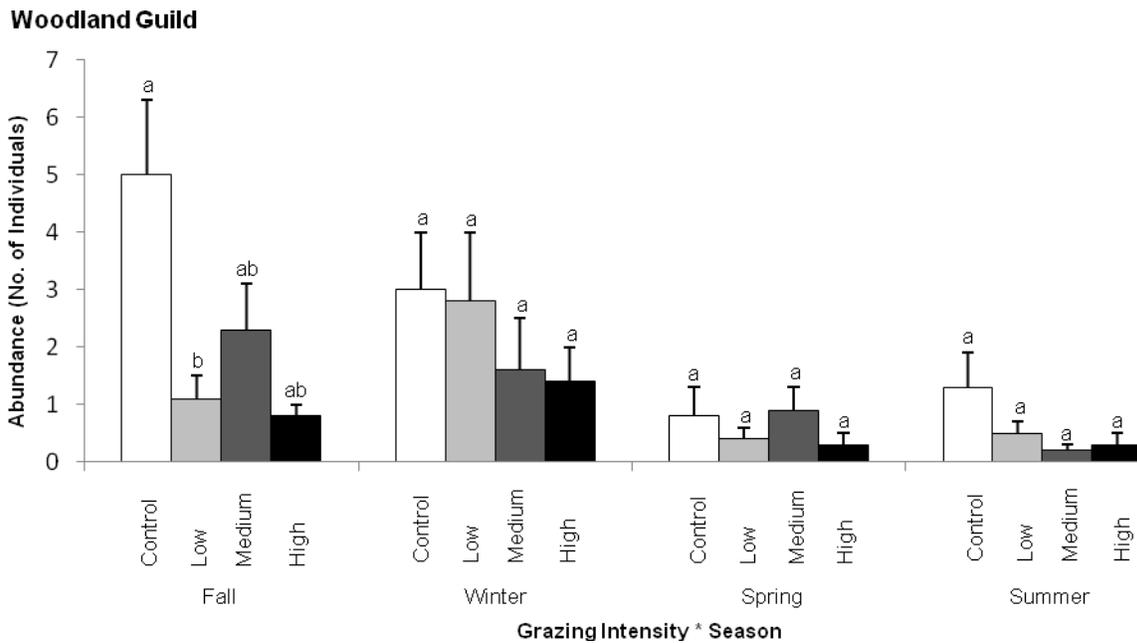
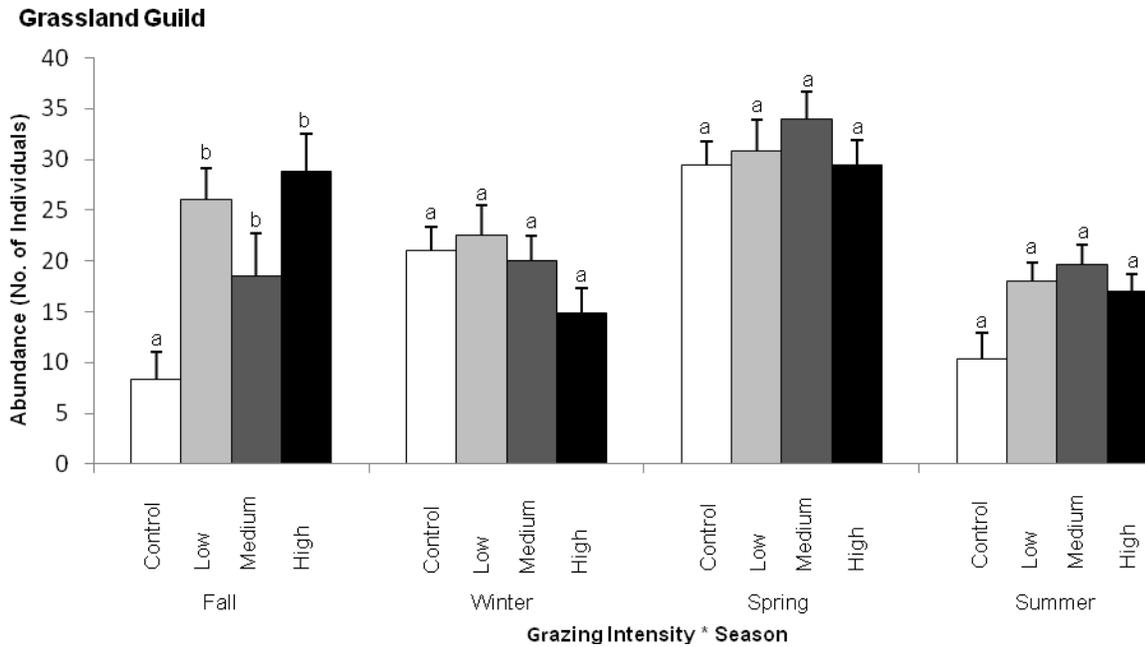


Figure 7-3. Effects of a grazing intensity * season interaction on avian abundance and species richness by guild in monoculture pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present $\bar{x} + SE$. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $1.3 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.0 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.6 \text{ ha} \cdot \text{AU}^{-1}$.

Woodland Guild

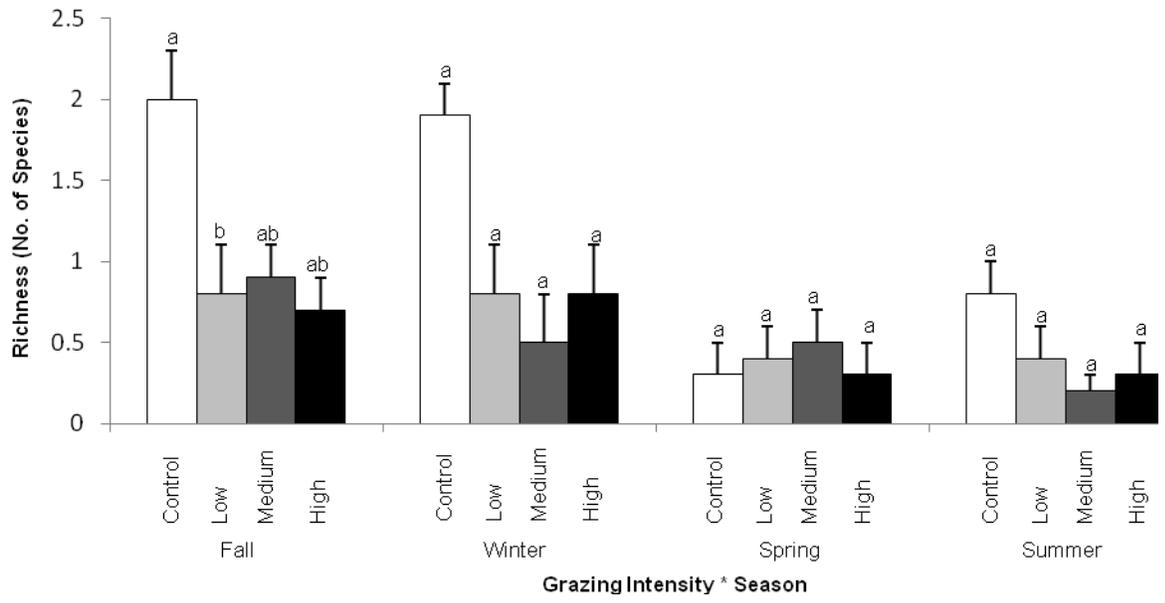


Figure 7-3. Continued

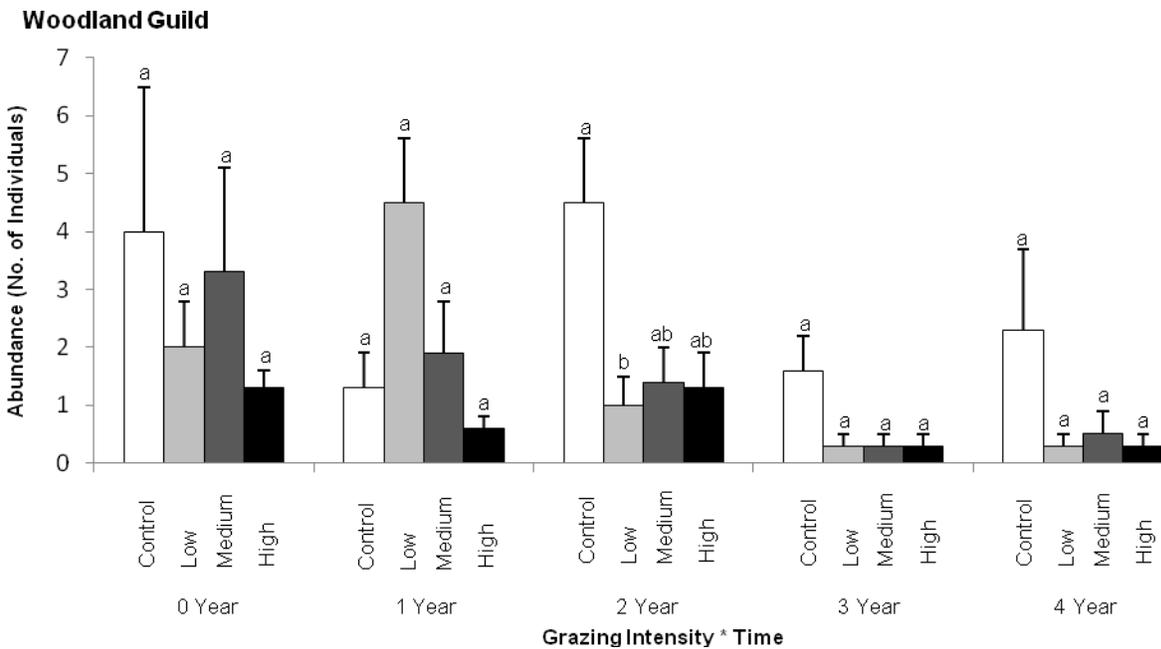
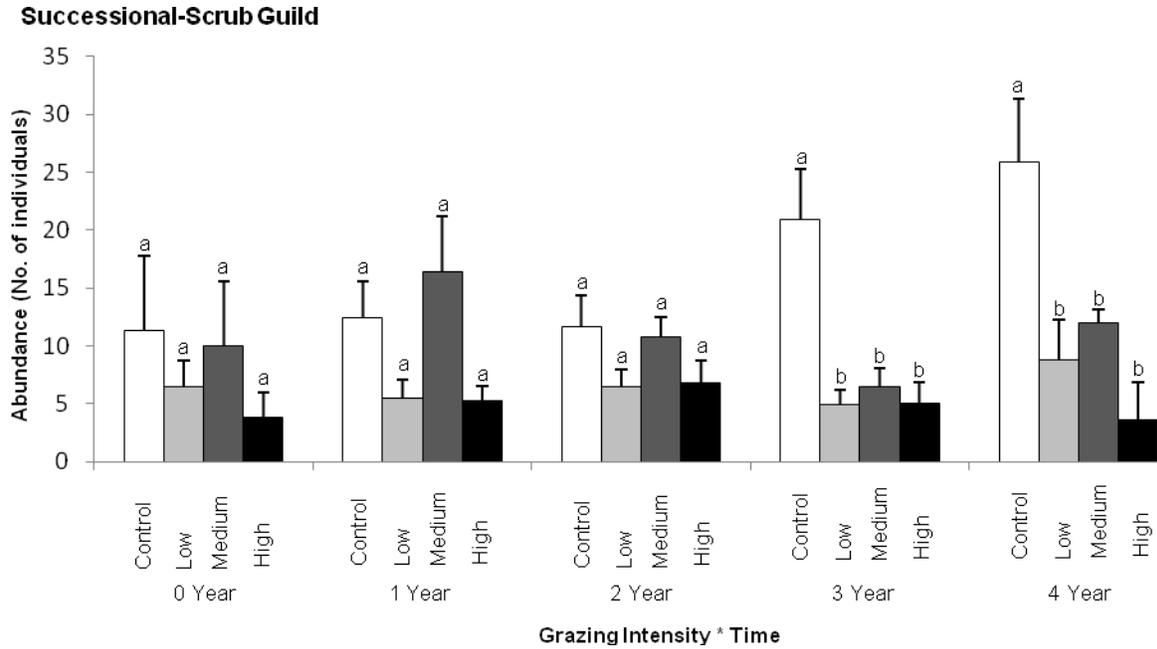
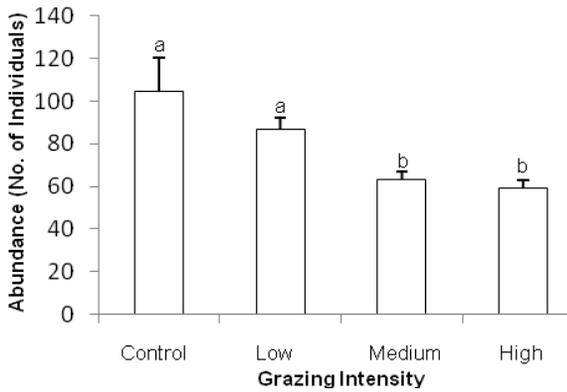
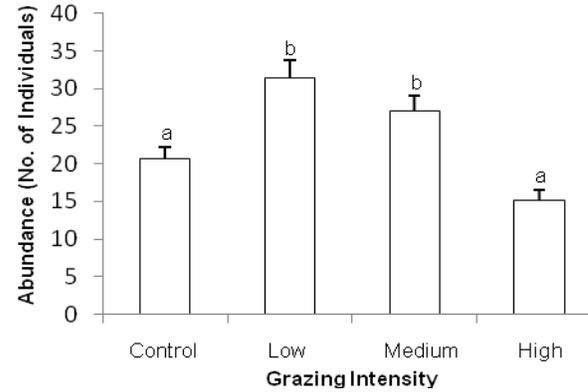


Figure 7-4. Effects of a grazing intensity * time interaction on avian abundance by guild in monoculture pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present $\bar{x} + SE$. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $1.3 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.0 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.6 \text{ ha} \cdot \text{AU}^{-1}$.

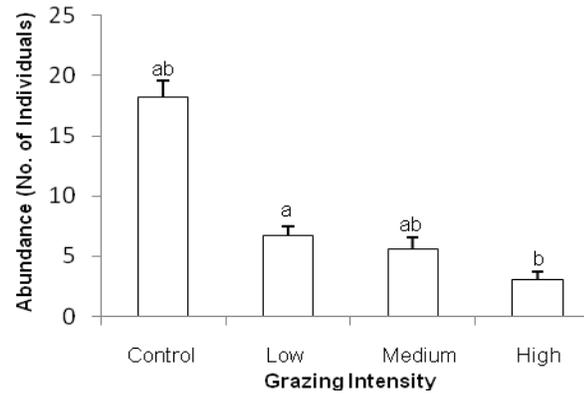
Total Avian



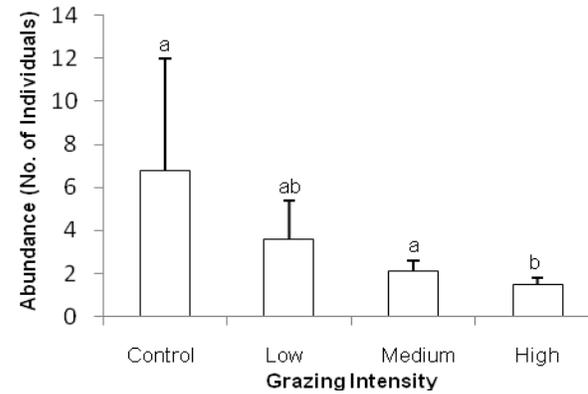
Grassland Guild



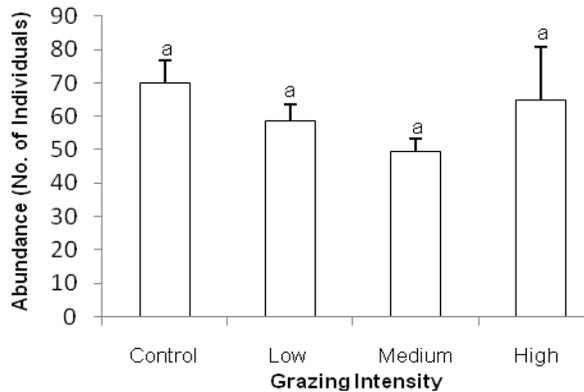
Successional-Scrub Guild



Urban Guild



Short-Distance Migrant Guild

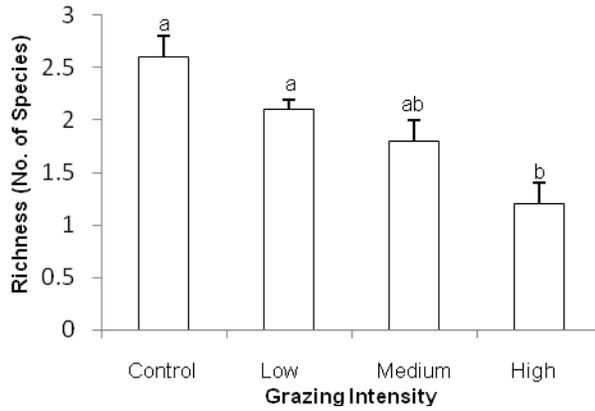


Neo-Tropical Migrant Guild

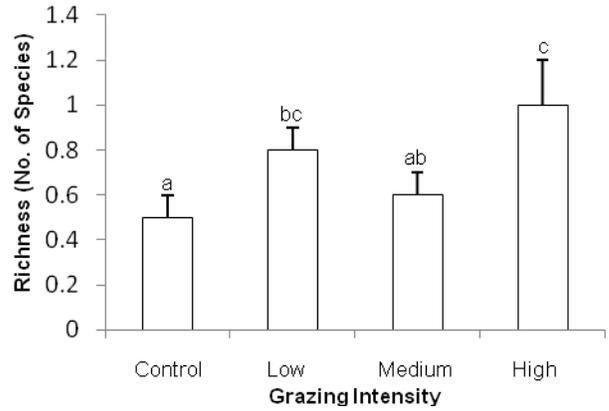


Figure 7-5. Effects of grazing intensity on total avian abundance and avian abundance by guild in mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present \bar{x} + SE. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $2.1 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.6 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.9 \text{ ha} \cdot \text{AU}^{-1}$.

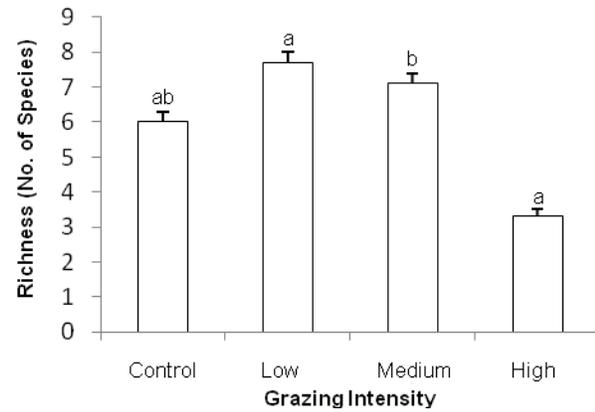
Successional-Scrub Guild



Urban Guild



Short-Distance Migrant Guild



Neo-Tropical Migrant Guild

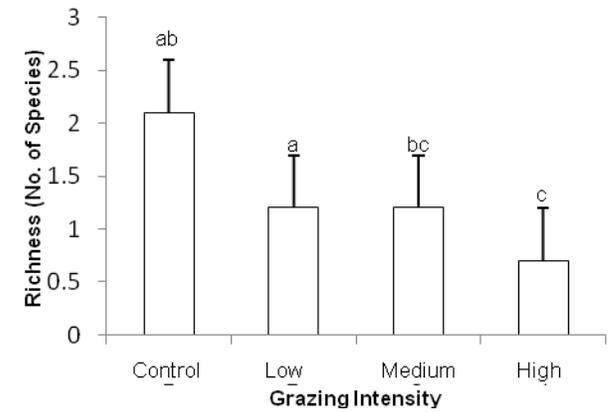


Figure 7-6. Effects of grazing intensity on avian species richness by guild in mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present $\bar{x} + SE$. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $2.1 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.6 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.9 \text{ ha} \cdot \text{AU}^{-1}$.

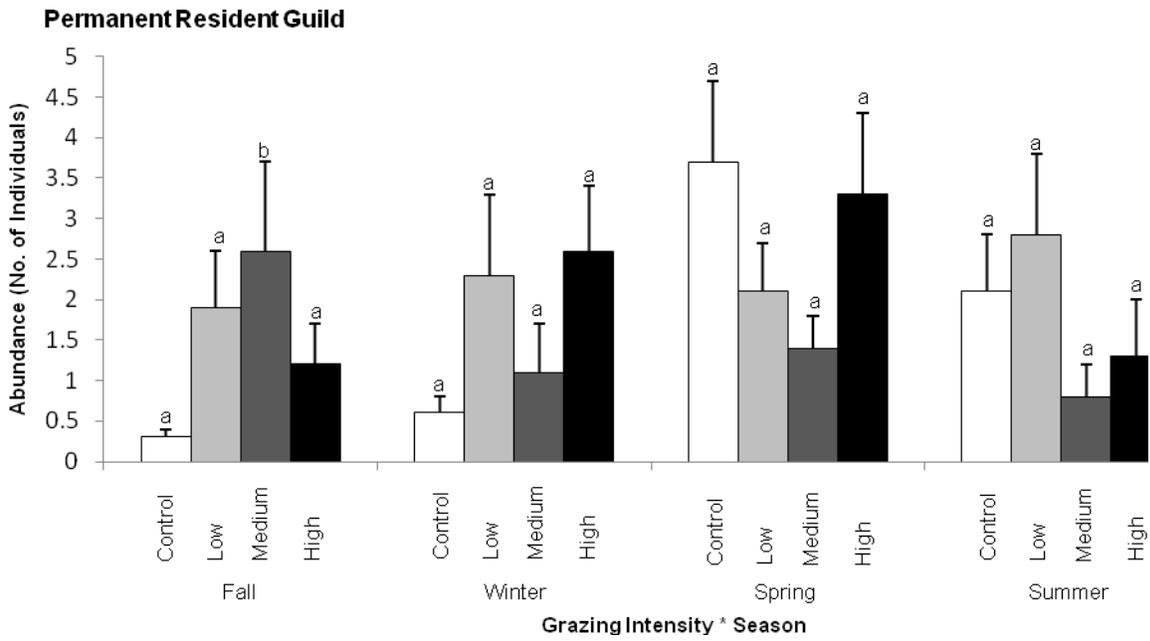
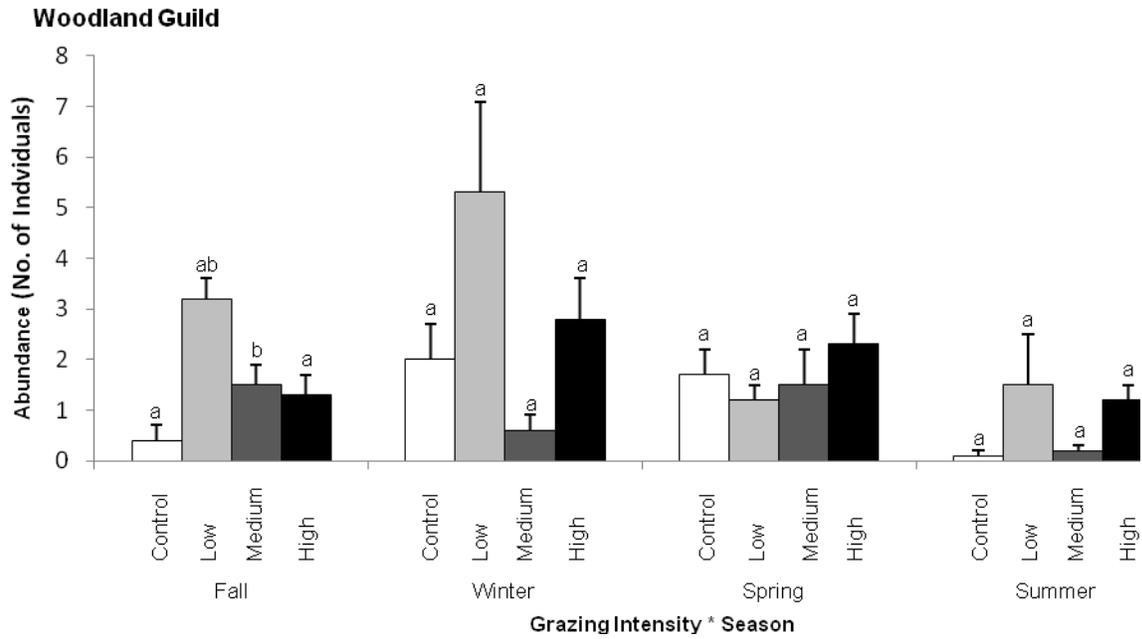


Figure 7-7. Effects of a grazing intensity * season interaction on total avian species richness and avian abundance and species richness by guild in mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present $\bar{x} + SE$. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $2.1 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.6 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.9 \text{ ha} \cdot \text{AU}^{-1}$.

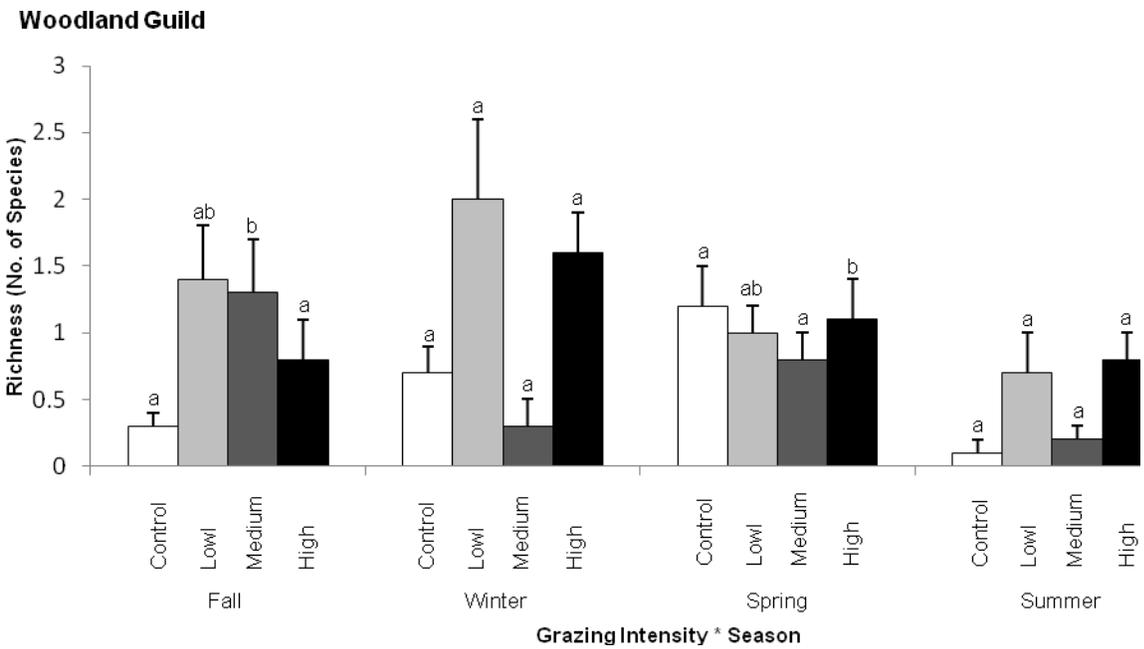
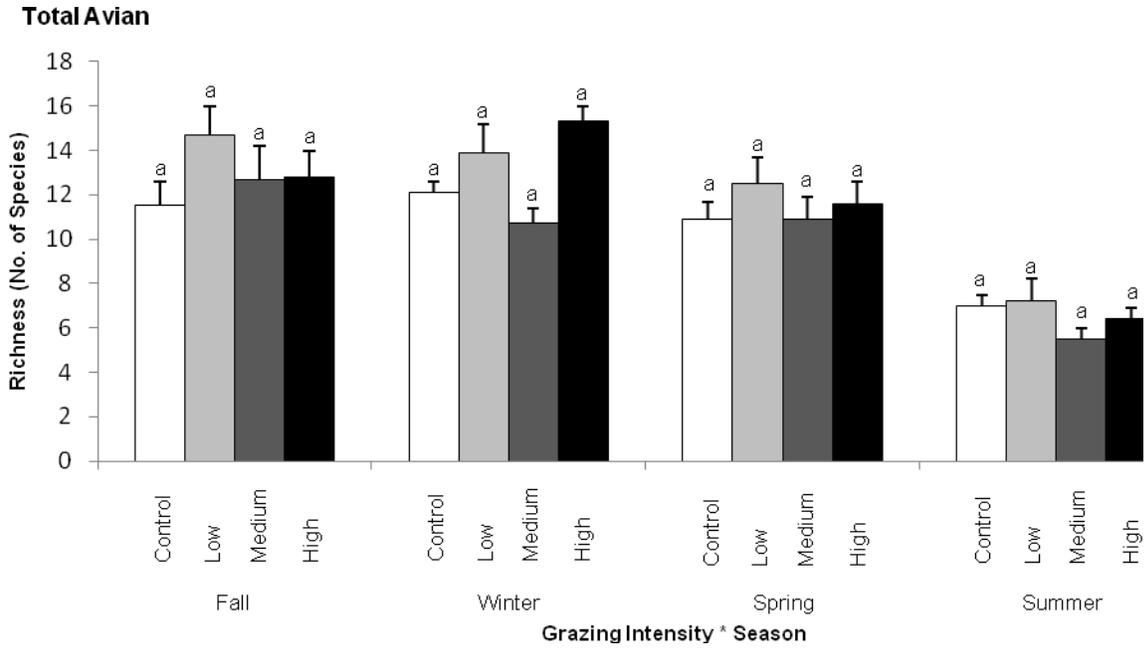


Figure 7-7. Continued

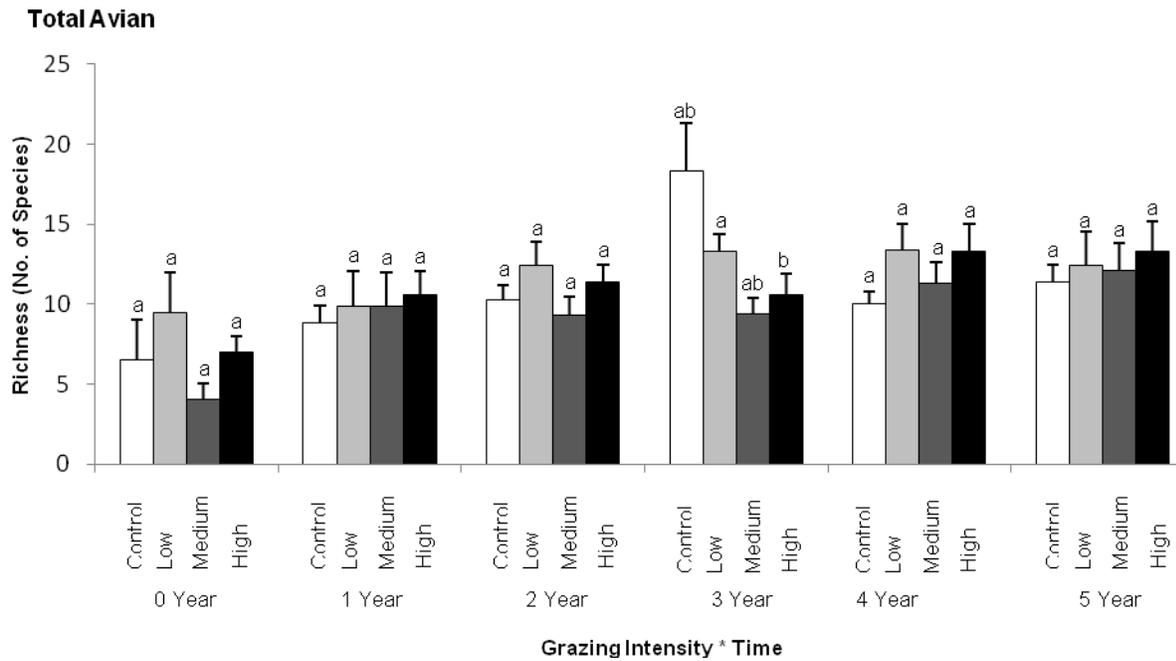


Figure 7-8. Effects of a grazing intensity * time interaction on avian species richness by guild on mixed pastures at MacArthur Agro-Ecology Research Station, Highlands County, Florida, 1999–2003. Bars present $\bar{x} + SE$. Bars topped by different letters are significantly different ($P \leq 0.1$). Grazing intensities: control = nongrazed, low = $2.1 \text{ ha} \cdot \text{AU}^{-1}$, medium = $1.6 \text{ ha} \cdot \text{AU}^{-1}$, and high = $0.9 \text{ ha} \cdot \text{AU}^{-1}$.

CHAPTER 8 CONCLUSIONS

The results of this study suggest prescribed burning and roller chopping, if appropriately applied, have an important role to play in maintaining and enhancing pine flatwoods habitat and associated wildlife communities. In areas of pine flatwoods, where active management has been lacking and shrub proliferation has become a concern, roller chopping provides a means of quickly reducing shrub cover, height, and density to achieve conditions that are more favorable for herbaceous plant growth. In addition, prescribed burning and roller chopping practices, depending on season of application, can shift the structure and composition of pine flatwoods vegetative communities towards conditions preferred by overwintering, permanent resident, migrant, and breeding avian guilds. The study also highlights the need to consider the effect of prescribed burning and roller chopping on arthropod communities, which are frequently negatively affected by these practices. Many arthropods are important pollinators or pests and the practices examined can alter their abundance, potentially hindering or benefiting conservation and control efforts.

Given the lack of knowledge, many questions surrounding vegetation, avian, and arthropod response to prescribed burning and roller chopping still need to be addressed to advance wildlife management. There is a need to investigate the effect frequency of roller chopping has on the structure and composition of the pine flatwoods plant community and, in turn, wildlife. A greater understanding of the interaction of prescribed burning and roller chopping would also be useful in determining the likely impacts of this combination treatment. This study focused on the examination of changes in species richness and abundance of various taxa following prescribed burning and roller

chopping and examined their habitat preferences to enable better tailoring of management activities to priority species or groups. As such, the study was primarily exploratory in nature, providing a strong foundation on which to build future research efforts. To allow practice effects to be more fully understood, future research should consider not only measures of species richness and abundance but demographic variables such as survival and fecundity. Inclusion of these variables would allow a more definitive assessment of habitat quality and help deepen the understanding of habitat preferences.

Grazing had considerable impacts on avian communities occupying monoculture and mixed pastures. Monoculture pastures, in particular, exhibited decreasing spatial homogeneity of the vegetative community as grazing level increased. Loss of spatial heterogeneity typically results in a lack of suitable habitat for birds that occupy the extremes of the vegetation structure gradient. This can lead to a loss of species richness and abundance. For the majority of avian guilds, a low grazing intensity of 1.3 and 2.1 ha • AU⁻¹ on monoculture and mixed pasture, respectively, is recommended to maintain abundance. However, these grazing intensities may result in declines in species richness. Ultimately, if a range of avian species are to be supported on monoculture and mixed pastures, spatial heterogeneity of plant structure and composition must be maintained. These findings suggest that further investigation of the role of livestock as ecosystem engineers could benefit certain avian groups.

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BIOGRAPHICAL SKETCH

Emma graduated with a B.S. in zoology from the University of Cardiff, Wales, United Kingdom. Her M.S. in conservation biology is from the University of Kent, Canterbury, United Kingdom. Emma is an applied ecologist with a strong interest in wildlife ecology, management, and conservation on private, primarily agricultural, lands. Her primary research focus is in the area of wildlife habitat management. In addition, she has a strong interest in undergraduate teaching and wildlife extension activities.