

EFFECTS OF FIRE ON HABITAT ASSOCIATIONS,
ABUNDANCE, AND SURVIVAL OF WINTERING
HENSLOW'S SPARROWS (*AMMODRAMUS HENSLOWII*) IN
SOUTHEASTERN LOUISIANA LONGLEAF PINE SAVANNAS

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DEDICATION

To my parents, Ivan and Delia Johnson, and my grandmother, “Mamuschka” Marianne von Randow, for their support and belief in me to pursue my aspirations in graduate school.

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ABSTRACT

The Henslow's Sparrow (*Ammodramus henslowii*) is the fastest declining short-distance migrant and fastest declining grassland bird in North America. Declines in longleaf pine (*Pinus palustris*) savannas, its primary wintering habitat, have exceeded 97%. Other than basic habitat requirements, very little is known about its wintering ecology. I examined habitat associations of wintering Henslow's Sparrows resulting from fire in longleaf pine savannas in southeastern Louisiana during two winters. Because it is important to understand the relative importance of habitat parameters, which include structure, species composition, and food availability, I measured these to determine their relative importance to Henslow's Sparrow densities. I aged birds to understand the effects of fire on age-specific distributions and body condition, employed a mark-recapture analysis to determine over-winter survival, and determined over-wintering home ranges starting when Henslow's Sparrows first arrived in October until they departed in April. Bird density was higher in savannas burned during the previous growing season ("burn-year") than in savannas burned two growing seasons before ("non-burn-year") in a two-year fire rotation scenario. Burning caused plant species compositions to shift from a *Rhychospora* spp. dominated habitat to an *Andropogon/Schizachyrium* spp. dominated habitat, however, total seed availability was not significantly different between burn-year and non-burn-year savannas. Because habitat-mediated age distributions, body condition, and home range size were also not significantly different between fire treatments, these findings are consistent with the idea that wintering Henslow's Sparrows are generalist foragers. Bird density was best predicted by habitat structure, specifically reduced ground-level herbaceous vegetation and fewer shrubs. A higher survival probability was detected in burn-year savannas and may be responsible for habitat selection in this species. It is not clear how Henslow's Sparrows locate apparently high

quality habitat due to unpredictable changes from one year to the next. I therefore determined the extent of post-migration movements and between-year site fidelity. There was evidence of post-migration movements, while nine of 154 birds exhibited between-year site fidelity, suggesting that a variety of spatial use strategies occur in this population. Land managers can manipulate savanna structure through frequent growing-season prescribed fires, which are critical to Henslow's Sparrow conservation.

CHAPTER 1. THE EFFECTS OF FIRE ON HENSLOW'S SPARROW DENSITIES, BODY CONDITION, HOME RANGE SIZE, AND SURVIVAL PROBABILITIES

INTRODUCTION

Concern over migratory birds in North America is largely because of habitat loss on breeding and wintering grounds and along migratory routes (Sauer et al. 2003, Newton 2004, Rich et al. 2004). It is important to understand how this threat affects population dynamics and to determine the relative importance on each phase of their life cycle (Sherry and Holmes 1996, Sandercock and Jaramillo 2002, Newton 2004, Winter et al. 2005). Not only do breeding, migratory, and wintering periods each have direct effects on avian survival and fitness, carried-over effects have been documented as well (Bearhop et al. 2004, Newton 2004, Saino et al. 2004, Hobson 2005, Norris 2005), as originally predicted by Fretwell (1972). For example, in a population of American Redstarts (*Setophaga ruticilla*) wintering in Jamaica, birds that occupy high quality habitats maintain relatively high body condition. This enables them to depart wintering grounds first, thus affording them first access to high quality breeding territories compared to individuals that occupy low-quality wintering habitat (Marra et al. 1998, Norris et al. 2004).

The importance of the non-breeding season in population dynamics is essential to understand to develop comprehensive management plans for migratory species. Neotropical migrants have received a disproportionate amount of attention compared to short-distance migrants. For example, in a recent review by Newton (2004) describing limits to populations of migratory birds, attention is given to long-distance migrants of the New and Old Worlds. He contrasts that group to resident birds, with little discussion of short-distance migrants. Because limits to populations may differ between Neotropical and short-distance migrants, these groups should be addressed separately for several reasons. First, processes that affect survival

throughout the life cycle are not inherently the same for short-distance migrants and Neotropical migrants. Although breeding ranges overlap, timing of migration (Carlisle et al. 2005) and corresponding energetic requirements differ between these two groups (Moore and Kerlinger 1992, Long and Stouffer 2003). More importantly, wintering latitudes differ in climate (Sauer et al. 1996), habitat (Sillett and Holmes 2002), and predation pressures (Lank et al. 2003, Nebel and Ydenberg 2005). Second, because short-distance migrants adjust their timing of migration to local conditions, which correlate with conditions on breeding grounds, short-distance migrants show a greater response to global climate change than do long-distance migrants (Butler 2003, Marra et al. 2005). This can decrease breeding fitness because the phenology of other organisms (such as plants and insects) often does not respond as rapidly to climate change. Third, because most short-distance migrants spend their entire life cycle within the United States and Canada, conservation strategies are relatively easy to address from political and practical perspectives. It is therefore critical that the wintering ecology and its role in population dynamics are understood to develop appropriate management decisions for short-distance migrants.

Of all short-distance migrants in North America, Henslow's Sparrow (*Ammodramus henslowii*) populations are declining at the fastest rate (Sauer et al. 2005). At an annual decline of 8.7%, it is also the fastest declining grassland bird in North America (Sauer et al. 2005) and therefore has been given the highest grassland bird conservation priority (Herkert et al. 1996). It is listed as endangered, threatened, or a species of concern in sixteen states and has been extirpated from New England (Burhans 2002).

Henslow's Sparrows breed from the northeastern United States and extreme southern Canada to the eastern plains states (Hyde 1939, Herkert et al. 2002). It occupies grasslands that have been disturbed by fire, mowing, or hardwood reduction within the last 2-4 years, which

creates a habitat with dense ground-level herbaceous vegetation and sparse woody vegetation (Zimmerman 1988, Swengel 1996, Herkert and Glass 1999, Cully and Michaels 2000, Burhans 2002).

Henslow's Sparrows winter from eastern Texas across the Gulf States and north along the southern Atlantic seaboard to North Carolina (Hyde 1939, Herkert et al. 2002). Much of this wintering range was historically dominated by longleaf pine (*Pinus palustris*) habitats (Figure 1.1), including flatwood savannas, seepage bogs, and upland savannas (Chapman 1932, Penfound and Watkins 1937, Wahlenberg 1946, Bridges and Orzell 1989, Peet and Allard 1993, Outcalt and Sheffield 1996). This ecosystem was historically maintained in a non-equilibrium condition generated by fire resulting from frequent growing season lightning strikes (Olson and Platt 1995, Platt 1999). Fire suppression, logging practices, and urbanization have reduced the longleaf pine ecosystem to about 3% of its former area (Figure 1.2). Only 0.01% of the overstory remains as old-growth (Frost 1993, Varner and Kush 2004), making it one of the most critically endangered ecosystems in North America (Noss et al. 1995).

Henslow's Sparrow winter ecology is poorly understood because of its secretive winter behavior (Burhans 2002, Herkert et al. 2002). Even as recently as 1991, the Henslow's Sparrow was not widely known to be a common winter inhabitant of the longleaf pine ecosystem (Engstrom 1993). What we have recently learned about the wintering ecology of Henslow's Sparrows comes from a few studies addressing basic habitat requirements. These studies found high concentrations of Henslow's Sparrows in grassy bogs in silvicultural lands (Plentovich et al. 1999) and throughout the longleaf pine ecosystem including pitcher plant bogs (Tucker and Robinson 2003) and savannas (Chandler and Woodrey 1995, McNair 1998, Carrie et al. 2002, Thatcher 2003, Bechtoldt and Stouffer 2005).

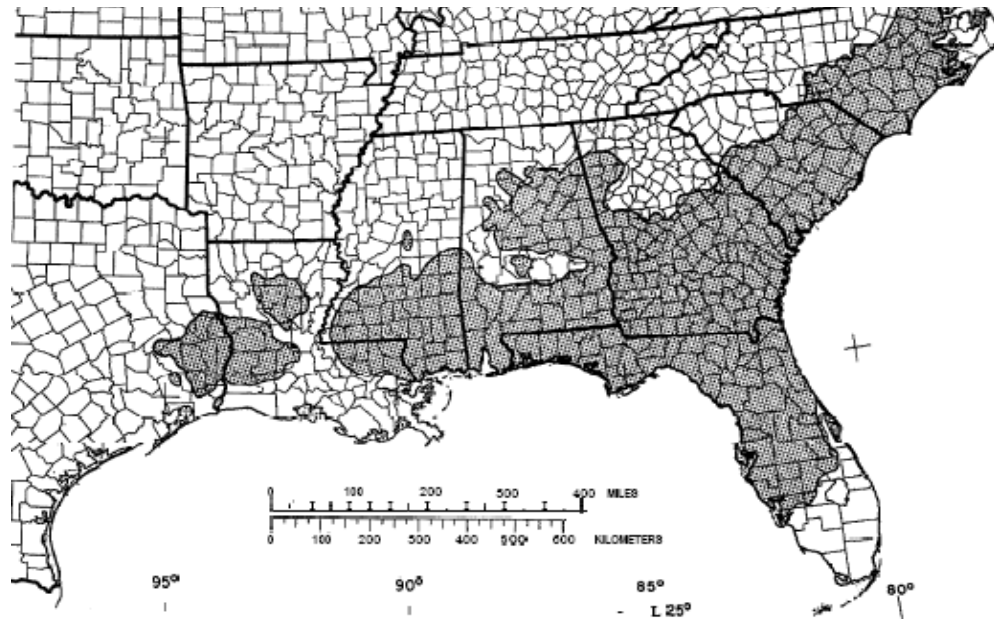


Figure 1.1. Historic range of longleaf pine ecosystems (Outcalt and Sheffield 1996).

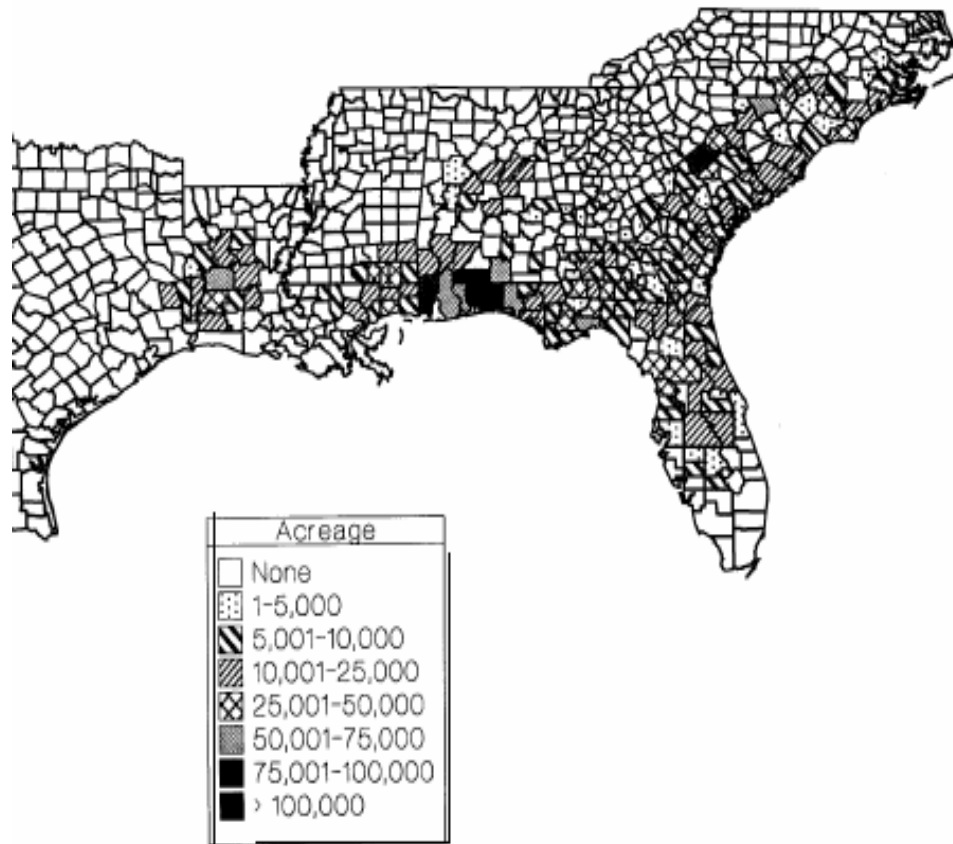


Figure 1.2. Estimated current acreage of longleaf pine ecosystems from Texas to North Carolina (Outcalt and Sheffield 1996).

Henslow's Sparrow winter densities are highest during the winter following an early growing-season fire (Carrie et al. 2002, Tucker and Robinson 2003, Bechtoldt and Stouffer 2005). As years since fire pass, bird density dramatically decreases and by three years post-fire, density approaches zero, suggesting that recently burned savannas are higher in quality (Bechtoldt and Stouffer 2005). Assuming that bird density reflects habitat quality, birds occupying the preferred habitat would be expected to show higher survival or be in better body condition than their counterparts occupying avoided habitats (Marra 2000).

Body condition is considered a practical estimate of fitness on the wintering grounds because it is thought to be directly related to survival (McNamara and Houston 1990, Newton 1993). Multiple measures of body condition are useful because climate, habitat, food availability, and predation risk affect various measures of body condition differently (Gosler and Carruthers 1999, Strong and Sherry 2000, Rogers and Reed 2003). In small wintering passerines, subcutaneous fat content and mass corrected for body size have been common and practical measures of body condition (Newton 1969, Gosler et al. 1995). Wintering passerines often carry less fat and are leaner than expected because it increases their risk to predation. Therefore, in habitats with high food availability, fat storage is kept to a minimum (Rogers 1987, Rogers and Smith 1993, Gosler et al. 1995). Higher amounts of fat have been interpreted as reflecting deficiencies in habitat quality because fat must be stored to survive during times of food shortages (Rogers and Smith 1993, Gosler et al. 1995, Bearhop et al. 2004).

Ptilochronology, which is a body condition measure that utilizes the rate of feather regrowth, has been another useful technique (White et al. 1991, Grubb and Yosef 1994, Carbonell and Telleria 1999, Jenkins et al. 2001, Brown et al. 2002). During times of stress, birds allocate energetic resources to survival and reduce rates of feather growth, which can be

determined by measuring distances between growth bars on induced feathers (Grubb 1989). This technique also has potential to reveal differences in condition related to demographics and environmental conditions (Grubb et al. 1991).

In addition to body condition, home range size can reflect differences in habitat quality. A relatively large home range can be indicative of poor habitat quality with reduced food resources (Brown et al. 2000). This can influence body condition because it provides the area in which food resources are acquired. A bird with a larger home range theoretically has access to more resources and can increase condition, assuming resources are distributed evenly and the entire area is uniformly accessible. Henslow's Sparrows are known to show strong within-winter site fidelity (Plentovich et al. 1998, Thatcher 2003, Bechtoldt and Stouffer 2005) and radiotelemetry estimates of home range for wintering Henslow's Sparrows generally are less than 0.5 ha. Surprisingly, home range size estimates do not differ between savannas set fire during the previous growing season and savannas set fire two or more growing seasons ago, despite differences in bird density (Thatcher 2003, Bechtoldt and Stouffer 2005). Telemetry studies are limited in that data can only be collected for about three weeks on a bird of this size (Bechtoldt 2002, Thatcher 2003, Bechtoldt and Stouffer 2005), giving an incomplete picture of over-winter home range.

Few estimates are available for winter survival and body condition in Henslow's Sparrows. Bechtoldt (2002) found no difference in fat scores between savannas set fire during the previous growing season and savannas set fire two or more growing seasons before. This suggests that birds inhabiting recently burned savannas may not benefit increased fitness, as might be expected based on bird density alone. Fat scores were analyzed without consideration of age, sex, time of day, or time of year effects (Bechtoldt 2002), which are known to be

important in regulating fat storage for many passerine species in winter (e.g. Newton 1969, Rogers 1987, Lilliendahl et al. 1996, Lilliendahl 2002). In Alabama, Thatcher (2003) used radiotelemetry to provide the first winter survival estimates for Henslow's Sparrows. He found that survival was higher in savannas set fire during the previous growing season than in savannas set fire two growing seasons before. Survival also fluctuated between years because of draught conditions one year. A more complete assessment of body condition and over-winter survival is needed to better understand the effects of time since fire on survival and fitness.

In this study I further examine habitat associations of wintering Henslow's Sparrows in a two-year prescribed fire-rotation scenario in southeastern Louisiana longleaf pine savannas, a project initiated by Bechtoldt and Stouffer (2005). I predict that distributions of birds will resemble those of previous studies examining the effect of fire interval on bird abundance. I aged birds to understand the effects of fire on age-specific distributions, employed a mark-recapture analysis to determine over-winter survival, and determined over-wintering home ranges starting when Henslow's Sparrows arrived in October until they departed in April. I collected three measures of body condition from captured birds to determine the effects of apparent habitat quality on wintering populations.

It is not clear how migrating Henslow's Sparrows locate apparently high quality habitat (i.e. savannas that were recently set fire). Shifts in habitat structure are generally dramatic and unpredictable from one year to the next in a two-year fire rotation scenario (Bechtoldt 2002). Pressures of locating preferred habitats are likely greater today compared to historical times because of the highly fragmented nature of remaining longleaf pine ecosystems (Noss et al. 1995, Varner and Kush 2004). I therefore used a mark-recapture analysis to determine the extent of post-migration movements.

METHODS

Study Site

This study was conducted in the Florida Parishes of southeastern Louisiana during the winters of 2003-04 and 2004-05. Ten 2.25 ha study plots distributed among four sites in longleaf pine savannas were sampled for Henslow's Sparrows (Figure 1.3, Table 1.1). Eight of these plots at three study sites were established in fall 2000 and described in detail in Bechtoldt and Stouffer (2005) and Bechtoldt (2002). These include one plot at Camp Whispering Pines (Girl Scouts of America), five plots at Lake Ramsay Wildlife Management Area (The Nature Conservancy and LA Department of Wildlife and Fisheries), and two plots at Abita Creek Flatwoods Preserve (The Nature Conservancy). Two additional plots were haphazardly chosen in upland savanna at Sandy Hollow Wildlife Management Area (WMA) based on fire rotation and ease of access. Camp Whispering Pines and Sandy Hollow WMA represent Southern Mesic Longleaf Woodland (upland savanna), while Lake Ramsay and Abita Creek represent Southern Longleaf Savanna (flatwoods) (Peet and Allard 1993).

Study sites have a sparse overstory of longleaf pine and many species of grasses (Poaceae) and sedges (Cyperaceae) in the ground cover. At upland sites, dominant plants include *Andropogon* spp., *Schizachyrium scoparium*, *S. tenerium*, *Muhlenbergia expansa*, and *Ctenium aromaticum*. In flatwoods communities, dominant plants include the same species as upland sites, as well as *Rhynchospora* spp. (see Chapter 2 for details on species composition).

Fire Treatment

The general goal of management staff was to burn savanna vegetation in a two-year fire rotation. These low-intensity fires were conducted during the early growing season (12 March – 1 June) except LR03, which was suspected to be set fire resulting from arson during the dormant

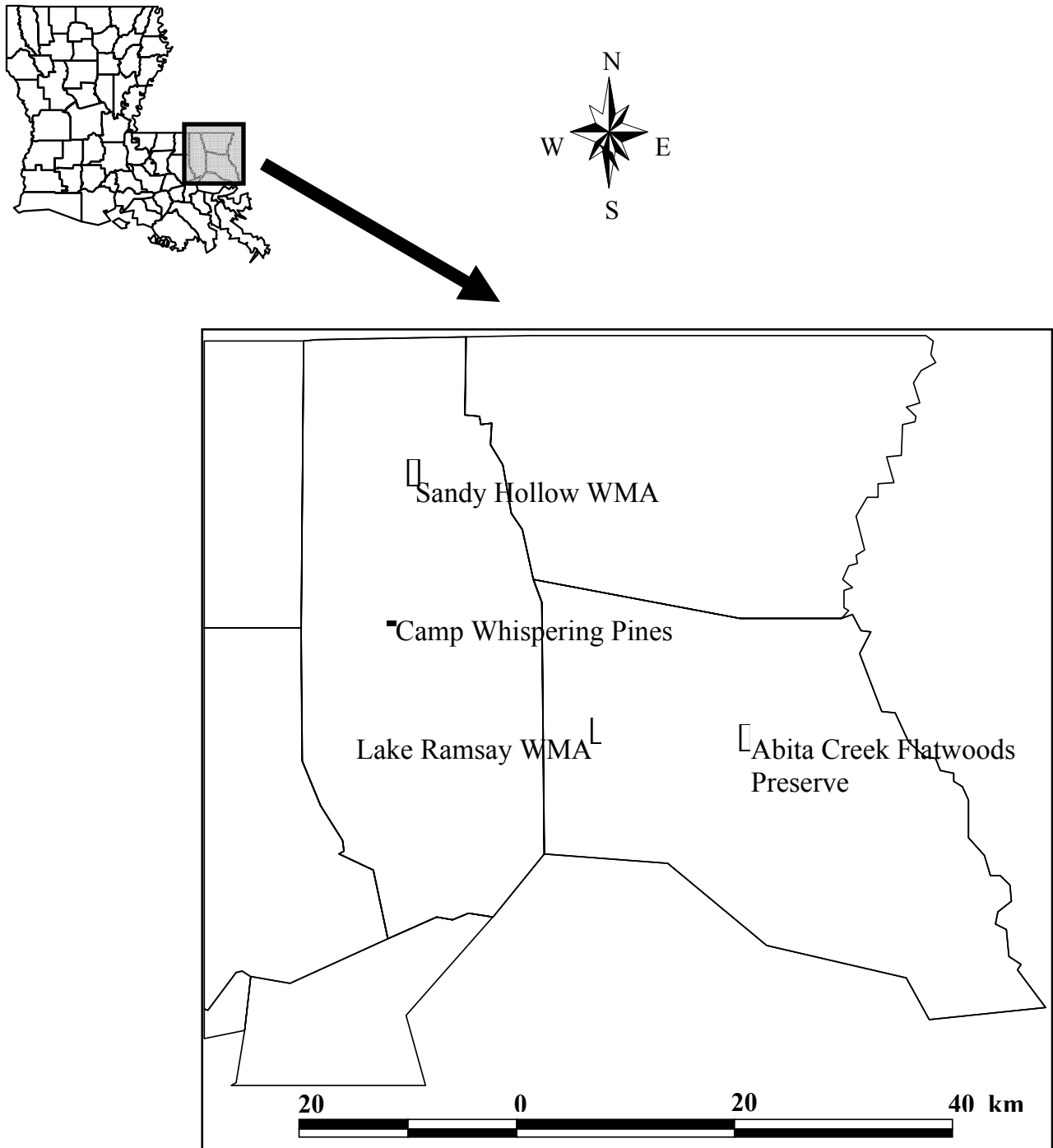


Figure 1.3. Locations of study sites within Louisiana and the Florida Parishes. Sandy Hollow WMA and Camp Whispering Pines are located in Tangipahoa Parish and Lake Ramsay WMA and Abita Creek Flatwoods Preserve are located in St. Tammany Parish.

Table 1.1. Plot locations, names, fire history, and area of management area where each plot is located.

Location	Plot Name (Management Unit)	Most Recent Fire		Size of Management Area (ha)
		2003-04	2004-05	
Lake Ramsay WMA	LR01 (TNC2)	23 May 2003*	23 May 2003 [†]	489.7
Lake Ramsay WMA	LR02 (TNC4)	12 Mar 2003*	12 Mar 2003 [†]	489.7
Lake Ramsay WMA	LR03 (WMA3)	22 Jan 2003*	22 Jan 2003 [†]	489.7
Lake Ramsay WMA	LR04 (WMA1)	29 May 2003*	29 May 2003 [†]	489.7
Lake Ramsay WMA	LR05 (WMA4)	22 Apr 2003*	12 Mar 2004*	489.7
Abita Creek Flatwoods Preserve	AS03 (AC01)	1 Jun 2002 [†]	1 Jun 2002 [†]	321.3
Abita Creek Flatwoods Preserve	AS01 (AC03)	23 Apr 2002 [†]	22 May 2004*	321.3
Camp Whispering Pines	CWP (Area 2A)	spring 2003*	spring 2003 [†]	19.0
Sandy Hollow WMA	SH01	1 Apr 2003*	1 Apr 2003 [†]	1422.5
Sandy Hollow WMA	SH02	1 Apr 2002 [†]	6 Apr 2004*	1422.5

* treatment = burn-year

[†] treatment = non-burn-year

season before the first sampling year (B. Rivers, personal communication). Table 1.1 shows the timing of fire for all ten plots during the study period.

Prior to the 2003-04 sampling period, six of the ten plots were set fire during the growing season and hereafter referred to as “burn-year.” The dormant season arson fire at LR03 is also included in the “burn-year” category. The remaining three plots were considered “non-burn-year” for analyses and were set fire two growing seasons before sampling.

Prior to the 2004-2005 sampling period, three of the ten plots were set fire during the growing season (burn-year). One of these three plots (LR05) was set fire prior to both sampling years. The other seven plots were considered “non-burn-year” and all except one had two growing seasons pass since the last prescribed fire. One (AS03) had three growing seasons pass since the last prescribed fire. The two types of non-burn-year conditions (two and three growing seasons since last fire) were lumped in statistical analyses because Bechtoldt and Stouffer (2005) found no significant difference in habitat characteristics and Henslow’s Sparrow density.

Henslow’s Sparrow Sampling and Data Collection

In 2003-04, plots were sampled for Henslow’s Sparrows seven times between November and March (Table 1.2). In 2004-05, the same plots were sampled five times over the same time period (Table 1.2). One preliminary sample was conducted in the same field as LR01 on 23 October, 2003. Additional sampling was conducted on a bi-weekly basis at CWP, LR01, and LR04 in late March and April of 2004. During October 2004 and April 2005, additional sampling was conducted on a weekly basis in four plots (LR01, LR05, SH01, and SH02). These four plots were chosen to represent every combination of burn-year/non-burn-year and upland/flatwoods, but SH01 could not be sampled after 3 April because it was set fire following this sampling date.

Jennifer DiMiceli and I collected data on birds. We quantified Henslow's Sparrows abundance with a team of 5-10 people walking parallel to each other, systematically covering 2.25 ha plots to target-net all flushed Henslow's Sparrows (Bechtoldt and Stouffer 2005). This involved the team surrounding where the bird landed, setting up a six meter mist net, and re-flushing the bird into the mist net. If the bird was not captured on the first attempt, we would continue to pursue the bird until it was caught or disappeared. Of the Henslow's Sparrows flushed, 64.1% and 73.9% were captured in 2003-04 and 2004-05 respectively, suggesting that this is an effective technique to capture wintering Henslow's Sparrows. Birds that were not captured but positively identified as Henslow's Sparrows were included in density calculations.

Birds were captured between 9:00 and 16:00 and the time of capture for each bird was recorded. Each captured Henslow's Sparrow was fitted with a unique US Fish and Wildlife band. Birds were aged as hatching-year/second-year (HY/SY), after-hatching-year/after-second-year (AHY/ASY), or unknown/after-hatching-year (U/AHY) based on plumage characteristics of the wing and/or skull ossification (Pyle 1997). We measured unflattened wing chord (± 0.5 mm), weight (± 0.25 g), and fat score (0-8) based on the Kaiser (1993) scale. We also removed the outer most right rectrix (R6) to induce feather growth, and the regrown feather was collected if it was nearly or fully regrown upon recapture (Grubb 1989). Collected feathers were stored in a paper envelope until measured in the lab. Following methodology described by Grubb (1989), feathers were placed over an index card and a pin was used to mark the most proximal and distal growth bar. This distance was then measured with digital calipers (± 0.01 mm) and the number of growth bars between these marks was counted, providing an average distance between growth bars. We only analyzed feathers for which we measured at least six growth bars. These data

provide three different measures to approximate body condition: 1) mass corrected for body size, 2) fat score, and 3) ptilochronology.

The location from where the bird first flushed (initial flush location) was marked with a Garmin eVista or eTrex Geographic Positioning System (GPS). Radiotelemetry has revealed that Henslow’s Sparrows do not run ahead of a sparrow flushing team before they flush into view (Bechtoldt 2002). Therefore, these GPS points represent the location a bird was occupying before the team approached the bird. For birds captured in multiple sampling periods, distances between initial flush locations were calculated using MapSource 4.09 (Garmin Corp.). The maximum distance measured was used as a proxy for home range. In comparison to radio telemetry studies, this is an efficient and inexpensive method to sample many birds over a long time.

Statistical Analysis

For all analyses, the 2.25 ha plot was the experimental unit and individual birds caught within these plots were observational units. The residuals of dependent variables from all analyses were tested for normality and homogeneity using a Shapiro-Wilk test of normality and

Table 1.2. Dates of Henslow’s Sparrow sampling by replicate.

Sampling Period	Range of Sampling Dates		Median Date	
	2003-04	2004-05	2003-04	2004-05
1	1 Nov – 13 Nov	6 Nov – 14 Nov	7 Nov	10 Nov
2*	15 Nov – 24 Nov (5 Dec) [†]	N/A	20 Nov	N/A
3	6 Dec – 11 Dec	4 Dec – 19 Dec	8 Dec	11 Dec
4*	14 Dec – 21 Dec	N/A	17 Dec	N/A
5	11 Jan – 24 Jan	8 Jan – 30 Jan	18 Jan	19 Jan
6	31 Jan – 9 Feb	11 Feb – 22 Feb	4 Feb	16 Feb
7	21 Feb – 29 Feb	5 Mar – 13 Mar	25 Feb	9 Mar

* Data collected during these replicates were not used when sampling period was used as a repeated measure in analyses.

[†] The December 5 date refers to a late sampling of LR04 during that replicate. Weather and other logistical situations made it impossible to sample before that date. Only one Henslow’s Sparrow was captured at that date, minimizing bias in the analyses.

Levene's Test of homogeneity; transformations of the dependent variables were made when appropriate. Treatment means are presented as Least Squares Means (LSMEANS, SAS Institute 2003) and back-transformed for ease of interpretation when dependent variables were transformed.

I used a quasi Latin rectangle with repeated-measures mixed-model analysis of variance (ANOVA; Appendix 2) to test the effect of fire treatment (burn-year vs. non-burn-year) on Henslow's Sparrow density. Bird density was expressed as Henslow's Sparrows flushed (captured + not captured) per hectare and log transformed to meet assumptions of parametric statistics. Fixed effects included fire treatment and sampling period (Table 1.1). Random effects included year (winter 2003-04 vs. winter 2004-05), plot, and error terms. Sampling period was used as a repeated measure assuming a compound symmetry covariance structure (PROC MIXED, SAS Institute 2003). Because there was unequal sampling effort between the two years of this study, the 2nd and 4th replicates from the 2003-04 field season were not included in analyses. The sampling periods of the remaining replicates were assumed to be biologically equivalent between years (Table 1.2), but LSMEANS are presented separately for each year. I followed this approach for all analyses using sampling period as a repeated measure.

Two analyses were conducted to test treatment-related differences in body condition. The first test used principal component scores for each captured bird using mass-corrected body size residuals and fat scores in a principal component analysis (Bearhop et al. 2004; PROC PRINCOMP, SAS Institute 2003). PC1 explained 72.5% of the variation in mass-corrected body size residuals and fat score (factor loadings for fat score and mass-corrected body size were both 0.707). Therefore, PC1 was selected as a body condition index. Mass-corrected body size residuals resulted from a regression of mass on body size (PROC REG, SAS Institute 2003). I

used a quasi Latin rectangle with repeated measures mixed model analysis of covariance (ANCOVA) to test treatment related differences on body condition (PROC MIXED, SAS Institute 2003). Fixed effects included fire treatment, age (HY/SY vs. AHY/ASY, with U/AHY deleted from the analysis), and sampling period. Random effects included year, plot, and error terms. Covariates included the capture time (number of minutes after sunrise) and minimum temperature (°C) during the previous night because these variables have been shown to be important in determining daily fluctuations in fat storage (Gosler 1996). These data were collected from the National Weather Service Daily Summary archives from Hammond, LA. Hammond, LA is located within 50 km of all ten study plots.

A separate test of body condition was conducted on a subset of birds using ptilochronology because only recaptured birds could be used. Again a quasi Latin rectangle with repeated-measures mixed-model ANCOVA was used with the same fixed and random effects as above using distance between growth bars of the induced feather as the dependent variable. The average distance between growth bars of the initial feather was used as a covariate to correct for individual variation in feather growth rates. Because of the low sample size, bird age could not be included in the model.

A body size- or sex-related hierarchical social structure may exist in wintering passerine populations that show strong habitat selection (Marra 2000), such as Henslow's Sparrows (Plentovich et al. 1999, Carrie et al. 2002, Tucker and Robinson 2003, Bechtoldt and Stouffer 2005). Because male Henslow's Sparrows are on average slightly larger than females (Pyle 1997), male birds may exclude female birds or larger individuals may exclude smaller individuals from the preferred habitat. Differences in body size between fire treatments were tested using a quasi Latin rectangle mixed-model ANOVA with the same treatments and random

effects as above. Wing chord was used as the dependent variable as an approximation for body size.

Treatment-related differences in home range could only be analyzed using recaptured birds. Again, a quasi Latin rectangle mixed-model ANOVA was used, however, instead of using “sampling period” as a repeated measure, two different intervals between captures (≥ 25 days vs. < 25 days) was used as a categorical variable. These categories were used to allow comparison with results of radio telemetry studies, which can only track birds for about three weeks.

Program MARK (White and Burnham 1999) was used to determine survival probabilities using a Cormack-Jolly-Seber (CJS) closed capture model (Cormack 1964, Jolly 1965, Seber 1965). Because of different sampling effort between the two years of study, years were analyzed separately; however, the global model was the same. Single-state capture histories were categorized into six groups using combinations of bird age (HY/SY and AHY/ASY), fire treatment (burn-year and non-burn-year), and location within plot (core and periphery). A bird was considered “core” if it was ever captured within a one hectare area centered within the plot. This distinction was made based on previous estimates of home range and knowledge of site fidelity (Thatcher 2003, Bechtoldt and Stouffer 2005). It was assumed that birds found along the periphery of the plot would spend much of their time outside the sampling area, thus have a lower capture probability than birds found towards the center of the plot. Disparities in the amount of time between sampling periods (sampling intervals) were corrected for and survival and capture probabilities are reported for an average sampling interval, which approximately corresponds to one month.

Model selection was based upon the lowest ranking of Akaike’s Information Criterion corrected for small sample size and over-dispersion (QAIC_c). Δ QAIC_c is the difference between

the top ranked model (smallest QAIC_c) and a competing model. If ΔQAIC_c is less than 2, then there is “substantial support” for all models within this range (Burnham and Anderson 2002).

The global model of each year, $\phi(. * \text{age} * \text{fire} * \text{time})\rho(\text{location} * .)$, where ϕ = apparent survival probability and ρ = capture probability, assumed fully time dependent survival parameters and accounted for differences in apparent survival between and within age classes and fire treatments, but core and peripheral birds were assumed to have equal survival probabilities. Capture probabilities were set to be time constant and equal among and within groups (Tucker and Robinson 2003), except for core versus peripheral birds. A Chi-Square test revealed that the percentage of birds caught of those flushed was not significantly different between burn-year and non-burn-year savannas ($\chi^2 = 0.464$, $df = 1$, $p = 0.496$; Table 1.3). Although this does not account for birds not flushed, it suggests that our netting technique is consistent between fire treatments and further supports the equal capture probability assumption.

Table 1.3. The number (and % of total) of birds captured and not captured of those flushed in burn-year vs. non-burn-year savannas combined over the two years of sampling.

	Burn-year	Non-burn-year
Caught	290 (68.4%)	123 (65.6%)
Not Caught	134 (31.6%)	64 (34.4%)
TOTAL	424	187

Following Lebreton et al. (1992), the global model was assessed for its goodness-of-fit (GOF) by estimating over-dispersion (\hat{c}), which was used to correct AIC_c calculations. The closer \hat{c} is to 1.0 the better the fit of the model. When $\hat{c} > 1.0$, there is evidence for over-dispersion and when < 1.0 , there is evidence for under-dispersion. Estimates of \hat{c} close to 1.0 (generally < 3.0) can be used to adjust AIC_c values, which are then represented as QAIC_c (Burnham and Anderson 2002). For both years of data, the global model was tested for its

goodness-of-fit using both a parametric bootstrap approach (White 2002) and the median \hat{c} test (White and Burnham 1999). The most conservative estimate of \hat{c} was used to correct for over-dispersion. A set of reduced candidate models was selected *a priori* and ranked according to QAIC_c statistics to find the most parsimonious model.

Candidate models included combinations of differences in survival by bird age, fire treatment, or sampling interval. One set of models was fully time dependent, one set was constant through time, and one set was partially time dependent. In 2003-04, I created three types of partially time dependent models. One type assumed differences in apparent survival between sampling interval 1-2 and all others (t122222, where numbers indicate the parameter index matrix [PIM] values used by program MARK). Another set of partially time dependent models assumed differences between sampling intervals 1-2, 2-3, and all others (t123333). A final set of partially time dependent models assumed equal apparent survival in sampling intervals 1-2 and 2-3, and these differed from all other sampling intervals (t112222). In 2004-05, I created only one partially time dependent model because there was no sampling period 2. This model assumed differences in apparent survival between sampling interval 1-3 and all others (t1222).

Partially time dependent models were created to detect post-migration movements because apparent survival cannot distinguish between actual survival and dispersal. I would not predict actual survival to be lower during these periods for two reasons: 1) migration likely eliminates the weakest birds before they arrive on their wintering grounds and 2) seed availability during this period is at its highest (unpublished data). *A priori* partially-time-dependent models were established to test whether Henslow's Sparrows exhibit post-migration movements (indicated by lower apparent survival during early sampling intervals), as would be

expected given their dramatic tendency to avoid non-burn-year savannas while inhabiting savannas set fire during the previous growing season. It is also possible to detect higher than expected apparent survival during this period, which would be attributed to the peak abundance of seed resources in November (Chapter 2, unpublished data). In either situation, partially time dependent models would account for differences in apparent survival between post-arrival and the remainder of the winter.

RESULTS

During this study, 246 individual Henslow's Sparrows were captured 413 times. Of the birds captured, 219 were successfully aged (89.0 %) (Table 1.4). Excluding unaged birds (U/AHY) from the analysis, a Chi-square test indicates that there is no age-specific occupancy in relation to time since last fire ($\chi^2 = 0.248$, $df = 1$, $p = 0.618$).

Table 1.4. The number (and % of total) of birds captured in burn-year vs. non-burn-year savannas by bird age combined over the two years of sampling.

Bird Age	Burn-year	Non-burn-year
HY/SY	105 (59.7%)	44 (55.7%)
AHY/ASY	47 (26.7%)	23 (29.1%)
U/AHY	24 (13.6%)	12 (15.2%)
TOTAL	176	79

Of the 246 individuals captured, 96 were recaptured at least once within the same year they were banded. The longest time span between recaptures was 144 days. Many of the birds not recaptured were either found in November or along the periphery of the plot (see below). Ninety-three of the 96 recaptured birds were recaptured within the same plot. One bird (HY/SY) was initially captured in LR02 in November 2003, recaptured about 250 meters away in LR01 in December 2003, and again recaptured in March 2004 in the same field as LR01, but about 200 meters further east, outside the plot. Another (AHY/ASY) was initially captured in SH02 in

November 2004 and recaptured almost 800 meters away in SH02 in February 2005. Finally, one (HY/SY) remained in LR05 in November and December 2003, was recaptured in AS01 in January 2004 (about 20 km east), and was subsequently recaptured in LR05 in February 2004. This bird returned to and remained within LR05 during the winter of 2004-05. Eight other birds banded during the first winter were recaptured during the second winter. All of these birds returned to the same plot in which they were initially captured.

A higher density of Henslow’s Sparrows was found in burn-year plots compared to non-burn-year plots during every sampling period, excluding sampling periods 2 and 4 from the analysis (Table 1.5, Figure 1.4). A Tukey-adjusted comparison of least square means showed that bird density in sampling period 3 was higher than all other replicates, except sampling period 5. In burn-year plots, bird density was lower in sampling period 1 than in 3, but not in non-burn-year plots. There was variation in Henslow’s Sparrow density among plots in the same fire treatment, and not all plots, such as SH01, SH02, and LR03, showed the same treatment effect as did the overall model (Figure 1.5).

There was no significant difference in body condition (PC1) between burn-year and non-burn-year savannas (Table 1.6, Figure 1.6). Body condition fluctuated over time, with lowest levels occurring during mid-winter (Figure 1.6). A Tukey-adjusted comparison indicated no significant difference in body condition between the first and last sampling periods.

Differences in body size were not significantly different between burn-year and non-burn-year savannas or between HY and AHY birds and there was no interaction (Table 1.8).

Table 1.5. The effects of fire and sampling period on bird density.

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	1	8	19.01	0.0024
Sampling Period	4	72	4.17	0.0043
Fire * Sampling Period	4	72	0.66	0.6235

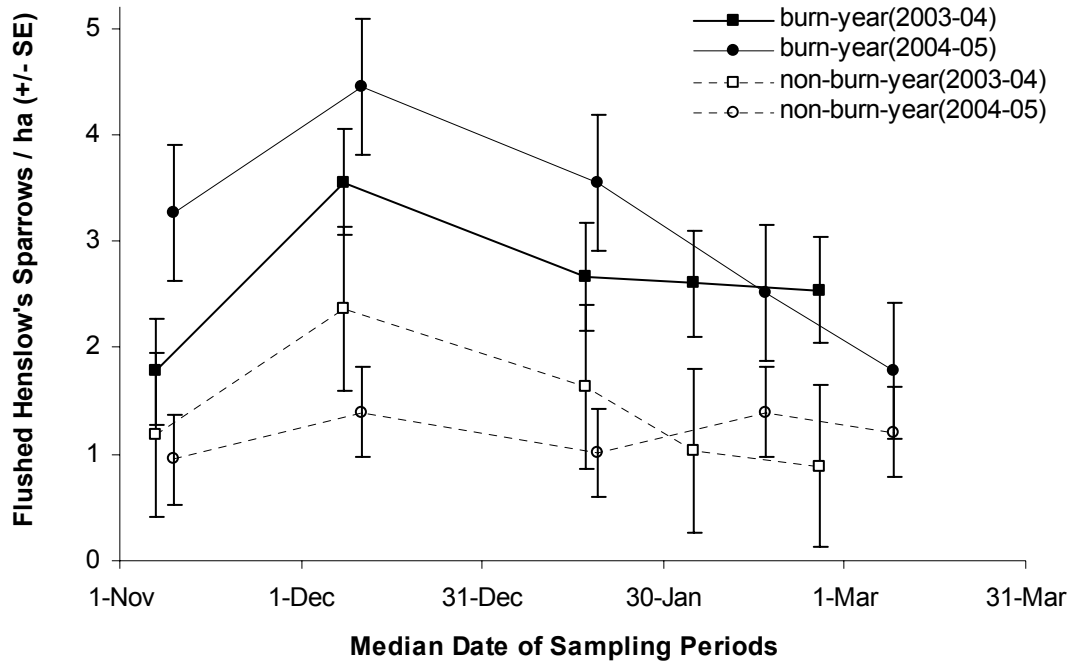


Figure 1.4. The average number of Henslow's Sparrows flushed by fire treatment, sampling period, and year (\pm SE). LSMEANS are presented at the median sampling date of each sampling period.

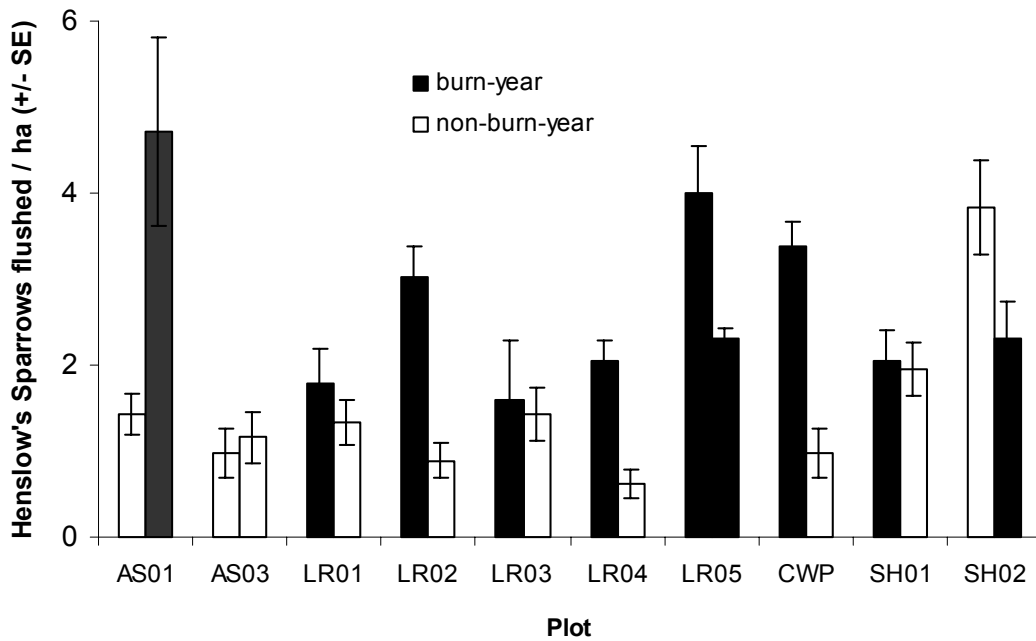


Figure 1.5. The number of Henslow's Sparrows flushed in each plot in each year (left bar: 2003-04, right bar: 2004-05), averaged over the number of sampling periods in each year (\pm SE).

Table 1.6. The effects of fire on PC1 (resulting from fat score and mass-corrected for body size).

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	1	8	0.12	0.7388
Sampling Period	4	57	12.31	< 0.0001
Sampling Period * Fire	4	57	0.46	0.7628
Age	1	175	0.00	0.9742
Age * Fire	1	175	2.34	0.1275
Age * Sampling Period	4	175	0.81	0.5179
Age * Fire * Sampling Period	4	175	1.84	0.1232
Minutes After Sunrise	1	175	7.42	0.0071
Minimum Temperature	1	175	0.38	0.5359

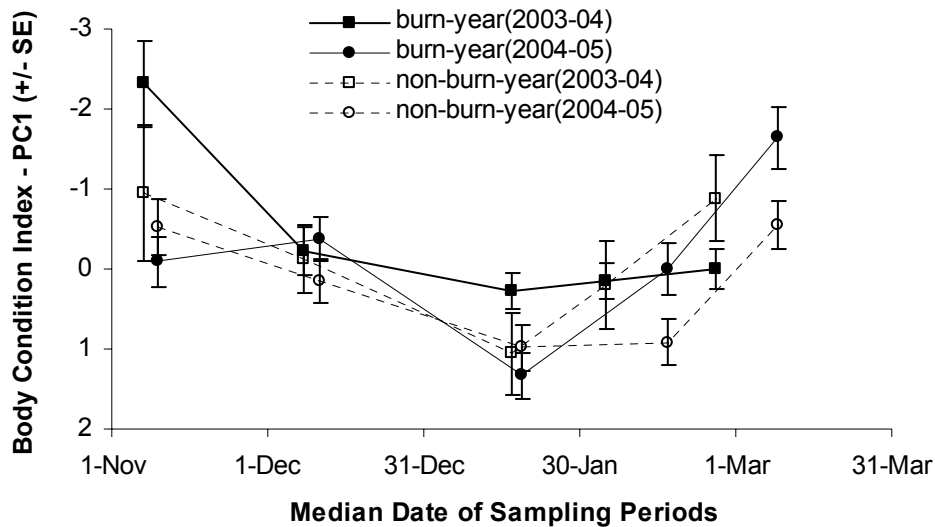


Figure 1.6. Scores from the first principal component (PC1) of a principal component using fat scores and mass-corrected body size by fire treatment and sampling period (\pm SE). Negative PC1 scores represent higher body condition (less fat and leaner) than expected, while positive PC1 scores represent lower body condition (fatter and heavier) than expected. LSMEANS are presented at the median sampling date of each sampling period.

Table 1.7. The effects of fire on induced feather growth (ptilochronology).

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	1	3	0.52	0.5234
Sampling Period	2	15	30.35	< 0.0001
Sampling Period * Fire	2	15	0.51	0.6093
Initial Feather Growth	1	15	31.65	< 0.0001

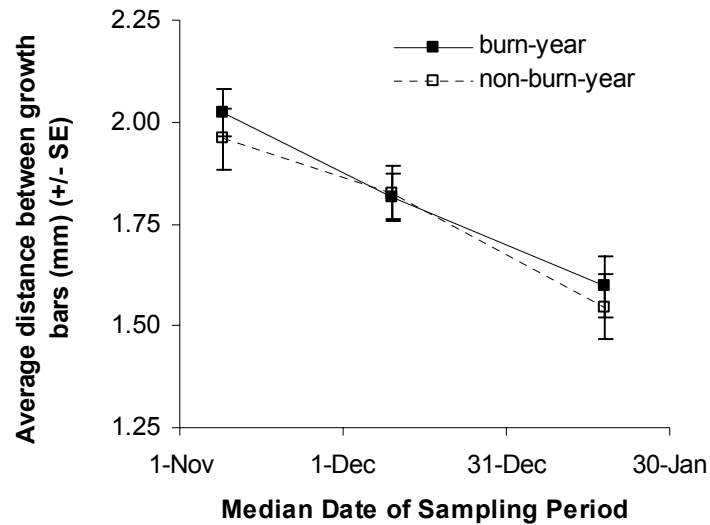


Figure 1.7. Induced feather growth by fire treatment and sampling period, correcting for initial feather growth (\pm SE). Slower regrowth (shorter distance between growth bars) indicates lower body condition than faster regrowth (greater distance between growth bars). LSMEANS are presented at the median sampling date of each sampling period. Because of small sample sizes, LSMEANS cannot be presented for each year separately; however, the median sampling date for each sampling period was less than three days apart between years (Table 1.2).

Table 1.8. Differences in body size by fire treatment and bird age.

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	1	8	0.19	0.6722
Age	1	252	0.01	0.9294
Age * Fire	1	252	1.12	0.2908

The maximum distance between flush locations was determined for each bird that was recaptured within a season as a proxy for home range. These data were log transformed to meet assumptions of parametric statistics. Distances between captures was greater for birds recaptured ≥ 25 days later (Table 1.9, Figure 1.8).

Figure 1.9 illustrates the percentage of birds captured in a given sampling period that were previously or subsequently recaptured, averaged over the two winters of study. Birds captured in the first and second sampling period were less likely to be recaptured than birds

captured during all other sampling periods. These data offer support to the reasoning behind including partially time dependent models in the analysis of winter survival.

In 2004-05, bootstrap GOF testing using 200 bootstrap simulations suggested that the observed deviance of the global model showed strong evidence of fit ($p = 0.465$) and $\hat{c} = 1.0076$. The median \hat{c} test resulted in $\hat{c} = 0.4273$ (SE ± 0.129); the relatively high standard error may have resulted from relatively sparse data compared to 2003-04 (Cooch and White 2006). Adjusting \hat{c} to 1.25, 1.0, 0.75, and 0.5 did not dramatically alter the ranking of the best models (QAIC_c < 7), thus the most parsimonious models were selected using \hat{c} set to 1.0076.

In 2004-05, two models were equally parsimonious (Burnham and Anderson 1998) and both included partially time dependent apparent survival (Table 1.12). One assumed equal apparent survival between fire treatments and age classes. The other assumed a difference between fire treatments. In both models, apparent survival probabilities were considerably lower during the first sampling interval and in the model that assumed differences in apparent survival by fire treatment; there was an interaction between sampling period and fire treatment (Table 1.13). Overall apparent survival probabilities were above 0.80 after the first sampling interval, thus apparent survival probabilities were lower in 2004-05 than in 2003-04 despite capture probabilities being higher in 2004-05 than in 2003-04 (Tables 1.11 and 1.13).

Table 1.9. The effects of fire and days between captures on home range size.

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	1	7	2.82	0.1369
Days Between Captures	1	8	6.90	0.0304
Days Between Captures * Fire	1	8	4.28	0.0724
Age	1	64	0.01	0.9376
Age * Fire	1	64	0.45	0.5058
Age * Days Between Captures	1	64	1.11	0.2953
Age * Fire * Days Between Captures	1	64	< 0.01	0.9954

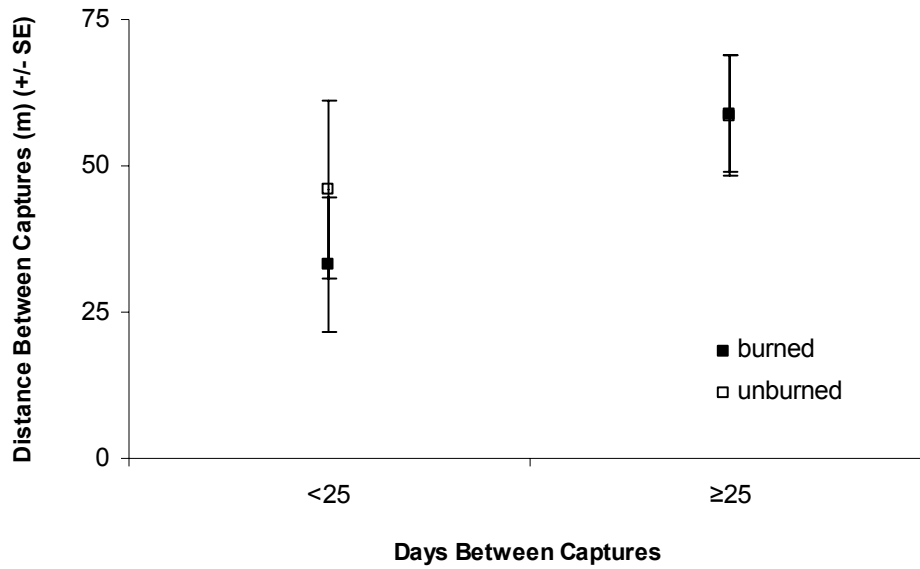


Figure 1.8. The effect of fire treatment on time between captures on home range size, measured by distance between captures (\pm SE).

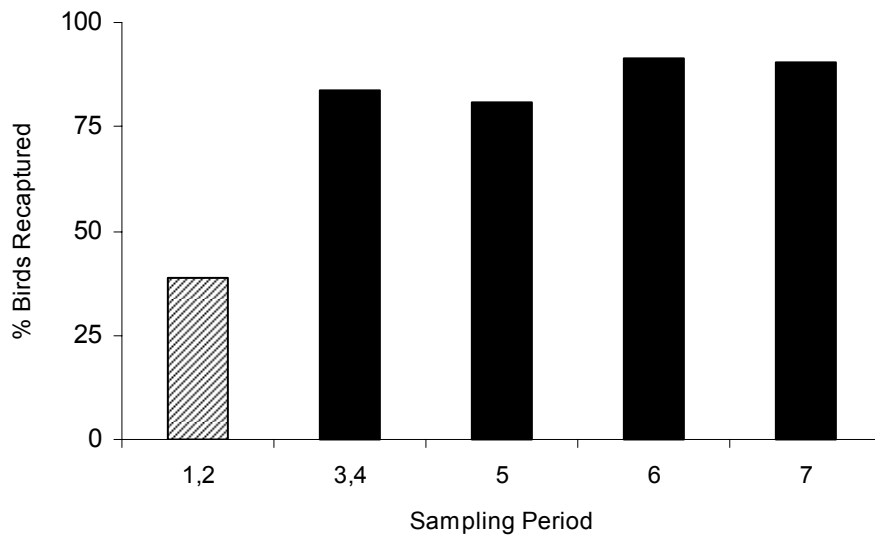


Figure 1.9. The percentage of birds captured during a sampling period that were previously or subsequently recaptured, averaged over the two years. The bar with slash represents less than 50% of birds recaptures. Solid bars represent greater than 50% of birds recaptured.

Capture probabilities were more than three times greater for core birds than peripheral birds (Table 1.14). Capture probabilities were about 60% higher during the second year of

sampling than the first year of sampling for core birds and 12% higher during the second year of sampling for peripheral birds.

Table 1.10. QAIC_c rankings of all models chosen to estimate apparent survival and capture probabilities in 2003-04. All candidate models assume constant capture probability among bird age and fire treatment groups and across sampling periods, but not between “core” and “peripheral” birds. All candidate models assume equal apparent survival among “core” and “peripheral” birds. Groups within models are ordered: location, bird age, fire treatment, and time. “ø” = apparent survival; “.” = constant survival within groups or time; “a” = apparent survival differs by bird age; “f” = apparent survival differs by fire treatment; “t” = apparent survival fully time dependent; “t122222” (and similar) = apparent survival differs between sampling interval “1 to 2” and all others (“partially time dependent”).

Model	QAIC _c	ΔQAIC _c	Model Weight	Model Likelihood	# of Parameters	QDeviance
ø(.*.*)	934.913	0.00	0.390	1.000	5	290.936
ø(*f*)	936.237	1.32	0.201	0.516	6	290.211
ø(a*.*)	936.379	1.47	0.188	0.481	6	290.352
ø(a*f*)	937.711	2.80	0.096	0.247	7	289.627
ø(*.*t122222)	938.881	3.97	0.054	0.138	8	288.731
ø(*.*t112222)	940.505	5.59	0.024	0.061	8	290.355
ø(*.*t123333)	941.651	6.74	0.013	0.034	11	285.252
ø(*f*t122222)	942.259	7.35	0.010	0.025	10	287.952
ø(a*.t122222)	942.273	7.36	0.010	0.025	10	287.966
ø(a*.t112222)	943.501	8.59	0.005	0.014	10	289.194
ø(*f*t112222)	944.004	9.09	0.004	0.011	10	289.697
ø(a*f*t122222)	945.686	10.77	0.002	0.006	12	287.187
ø(a*.t123333)	947.021	12.11	0.001	0.002	14	284.296
ø(a*f*t112222)	947.035	12.12	0.001	0.002	12	288.536
ø(*f*t123333)	947.401	12.49	0.000	0.002	14	284.676
ø(a*f*t123333)	952.850	17.94	0.000	0.000	17	283.719
ø(*.*t)	957.805	22.89	0.000	0.000	20	282.189
ø(*f*t)	969.559	34.65	0.000	0.000	26	280.723
ø(a*.t)	969.971	35.06	0.000	0.000	26	281.134
ø(a*f*t)	982.063	47.15	0.000	0.000	32	279.666

Table 1.11. Survival estimates (± SE) of the highest ranking models in 2003-04.

Model	ΔQAIC _c	ø _{burn-year}	ø _{non-burn-year}	ø _{HY/SY}	ø _{AHY/ASY}
ø(.*.*)	0.00	0.938 (0.021)		0.938 (0.021)	
ø(*f*)	1.32	0.947 (0.022)	0.911 (0.040)	0.938 (0.021)	
ø(a*.*)	1.47	0.938 (0.021)		0.956 (0.029)	0.928 (0.025)

Table 1.12. QAIC_c rankings of all models chosen to estimate apparent survival and capture probabilities in 2004-05. All candidate models assume constant capture probability among bird age and fire treatment groups and across sampling periods, but not between “core” and “peripheral” birds. All candidate models assume equal apparent survival among “core” and “peripheral” birds. Groups within models are ordered: location, bird age, fire treatment, and time. “ø” = apparent survival; “.” = constant survival within groups or time; “a” = apparent survival differs by bird age; “f” = apparent survival differs by fire treatment; “t” = apparent survival fully time dependent; “t1222” = apparent survival differs between sampling interval “1 to 3” and all others (“partially time dependent”).

Model	QAIC _c	ΔQAIC _c	Model Weight	Model Likelihood	# of Parameters	QDeviance
ø(.*. *t1222)	633.562	0.00	0.598	1.000	8	121.678
ø(. *f* t1222)	635.139	1.58	0.272	0.454	10	119.007
ø(a*. *t1222)	637.558	4.00	0.081	0.136	10	121.425
ø(a* f* t1222)	639.191	5.61	0.036	0.060	12	118.755
ø(. *. *)	643.325	9.76	0.005	0.008	5	137.712
ø(. *f* .)	643.694	10.13	0.004	0.006	6	136.004
ø(a*. *.)	645.382	11.82	0.002	0.003	6	137.692
ø(a* f* .)	645.764	12.20	0.001	0.002	7	135.984
ø(. * . *t)	645.998	12.44	0.001	0.002	14	121.202
ø(. *f* t)	649.434	15.87	0.000	0.000	18	115.747
ø(a*. *t)	651.308	17.75	0.000	0.000	18	117.621
ø(a* f* t)	654.953	21.39	0.000	0.000	22	112.134

Table 1.13. Survival estimates (± SE) of the highest ranking models in 2004-05.

Model	Time	ΔQAIC _c	ø _{burn-year}	ø _{non-burn-year}	ø _{HY/SY}	ø _{AHY/ASY}
ø(. *. *t1222)	t1	0.00	0.414 (0.131)		0.414 (0.131)	
ø(. *. *t1222)	t2	0.00	0.805 (0.064)		0.805 (0.064)	
ø(. *f* t1222)	t1	1.56	0.498 (0.183)	0.304 (0.187)	0.414 (0.131)	
ø(. *f* t1222)	t2	1.56	0.714 (0.089)	0.888 (0.078)	0.805 (0.064)	

Table 1.14. Capture probabilities (± SE) for core and peripheral birds in 2003-04 and 2004-05.

Year	ρ _{core}	ρ _{periphery}
2003-04	0.275 (0.028)	0.082 (0.023)
2004-05	0.443 (0.046)	0.092 (0.034)

Although limited October and April sampling was not thorough enough to compare arrival patterns between years and fire treatments, there appears to be a difference in the timing of arrival between years (Figure 1.10). On 23 October 2003, four Henslow’s Sparrows were

found by sampling one burn-year plot (Figure 1.10). None of these birds were aged and none were recaptured; this plot was located in the same field as and slightly overlapped LR01, but was not sampled for the remainder of the winter. During October 2004, two Henslow's Sparrows were found during four sampling periods and four plots were sampled during each sampling period (sixteen total sampling efforts). One bird was found on 17 October and other was found on 23 October (Figure 1.10). Both birds were aged as HY and neither was subsequently recaptured during the remainder of the winter.

Comparing departure patterns between fire treatments and years has limitations because of methodological inconsistencies. During the last sampling date, the three plots sampled in April 2005 had higher densities of Henslow's Sparrows than in April 2004, regardless of fire treatment (Figure 1.11).

Two of the twelve individual birds captured in 2004 were aged ASY, while the other ten were aged SY. Five of the twelve birds captured in April 2004 were captured previously in 2003-04. Of the seven birds not previously captured in 2003-04, four were captured in the core area of their respective plots in April 2004. Two of the ten individual birds captured in April 2005 were aged ASY, while the other eight were aged SY. Five of the ten birds captured in April 2005 were captured previously during the 2004-05 season. All of the five birds captured in April 2005, but not previously captured in 2004-05 were captured in the peripheral area of their respective plots.

During the duration of this two-year study, twelve Le Conte's Sparrows (*Ammodramus leconteii*) and six Grasshopper Sparrows (*A. savannarum*) were located, captured, and banded in thirteen and six captures respectively. All Grasshopper Sparrows were captured in November and December. Le Conte's Sparrows were captured between October 23 and February 9.

Because of the scarcity of these two species during the study period, especially compared to Bechtoldt (2002), no analyses were conducted on these species. Raw data for these species are in Appendix 3.

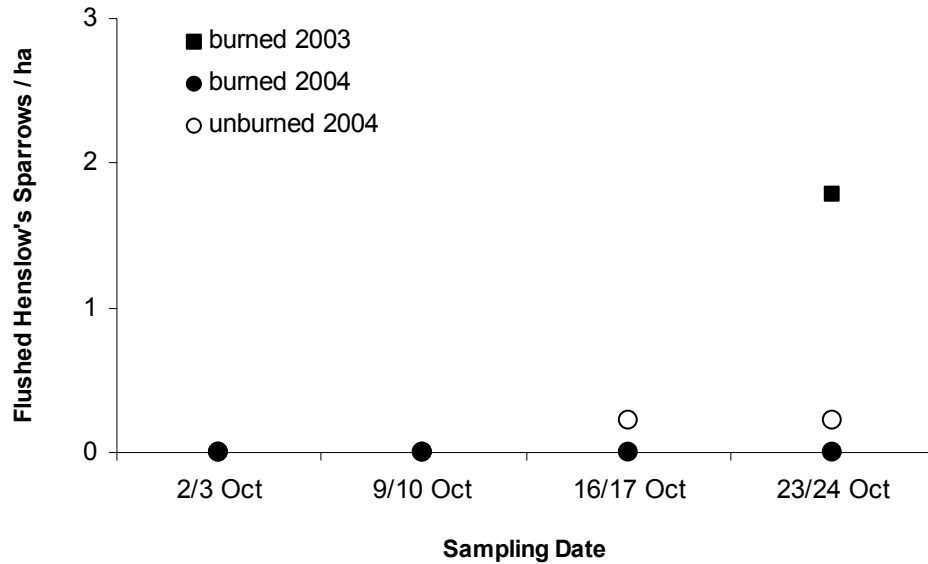


Figure 1.10. Results of sampling a subset of plots in October. Circles represent sampling in 2004, squares represent sampling in 2003, closed shapes represent burn-year plots and open shapes represent non-burn-year plots.

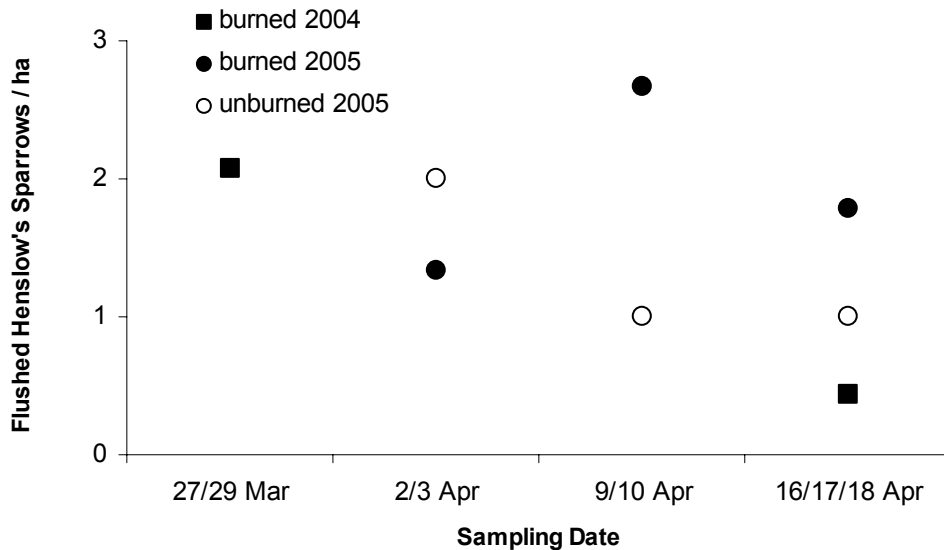


Figure 1.11. Results of sampling a subset of plots in late March and April. Circles represent sampling in 2004, squares represent sampling in 2003, closed shapes represent burn-year plots and open shapes represent non-burn-year plots.

DISCUSSION

Bird Density

As predicted, wintering Henslow's Sparrow densities were higher in burn-year plots compared to non-burn-year plot in this two-year fire rotation scenario. This was true for all sampling periods between November and mid-March (Figure 1.4). There are several plots that do not match the overall trend in bird abundance in relation to fire treatment, including LR03, SH01, and SH02 (Figure 1.5). LR03 was the only plot that was set fire during the dormant season and bird density did not change between the burn-year and non-burn-year treatments. This suggests that dormant season fires may support lower bird densities than growing season fires. The other two plots were upland plots; SH01 showed no difference in bird density between burn-year and non-burn-year treatments and SH02 showed an opposite pattern in bird density compared to the overall pattern such that bird density was higher during the non-burn-year treatment (Figure 1.5).

Fire appears to be less important in driving differences in vegetation patterns in upland savannas compared to flatwoods savannas, specifically in regard to the relative contributions of woody shrubs and herbaceous vegetation (Drewa et al. 2002, Glitzenstein et al. 2003). Other characteristics including long-term site history and edaphic factors are more important in determining vegetation assemblages (Drewa et al. 2002). Therefore, in upland savannas, Henslow's Sparrow densities would be expected to follow vegetation patterns determined by site history and edaphic factors more so than short-term fire effects. In support of this hypothesis, the response of bird density to fire treatment in upland savannas reported by Carrie et al. (2002) was not as dramatic as that in flatwoods savannas reported by Bechtoldt and Stouffer (2005).

Capture Probabilities

Although density estimates were assumed to be comparable because of our sampling technique, capture probabilities could be used to adjust these observed bird densities.

Complexities arise because capture probabilities differed between core and peripheral birds (Tables 1.14). It should be noted that density estimates based on the number of birds flushed are conservative by as much as 36.6% in 2003-04 and 29.6% in 2004-05. The percentage of birds captured of those that were flushed (see Methods) was higher than capture probabilities determined by the CJS model (Tables 1.14), suggesting that our sampling technique does not flush some birds given they are present.

Capture probabilities in 2004-05 were higher than in 2003-04 (Table 1.14). Ratios of burn-year to non-burn-year plots were 7:3 and 3:7 in 2003-04 and 2004-05 respectively (Table 1.1), thus average capture probabilities on average were higher in non-burn-year plots in burn-year plots. Differences in observed bird densities between burn-year and non-burn-year plots (Figure 1.4) are therefore conservative, further supporting that time since fire is important in determining wintering Henslow's Sparrow densities. Differences in capture probabilities also suggest that these may relate to differences in habitat structure or bird density. Baldwin (2005) had higher capture probabilities for wintering Le Conte's Sparrows in denser habitats, however, Tucker and Robinson (2003) did not detect a difference in winter Henslow's Sparrow capture probabilities between burn-year and non-burn-year savannas. In this study, no difference in the percent of flushed birds that were capture was detected between burn-year and non-burn-year savannas (Table 1.3). A more plausible explanation of higher capture probabilities during the second year of sampling is simply a result of experience in catching flushed birds (see Methods)

because of our increasing efficiency in coordinating crews and the increasing experience of many dedicated volunteers.

Body Condition and Home Range

No significant difference in body condition was detected between fire treatments or bird age classes (Tables 1.6 and 1.7). There was a time effect and body condition was lowest during mid-winter months (Figures 1.6 and 1.7), consistent with increased energetic demands associated with cold winter months. The index of condition based on fat scores and mass-corrected for body size was highly dependent on time since sunrise, as predicted. The minimum temperature during previous night was not important in determining fat scores and mass-corrected for body size, perhaps because of the distance between the meteorological station and study sites.

No significant difference in home range size was detected between fire treatments or bird age classes (Table 1.9), despite large differences in bird density (Figure 1.4), which is consistent with radiotelemetry studies (Thatcher 2003, Bechtoldt and Stouffer 2005). Incidental interactions with neighbors or the perception that nearby neighbors may be present could limit movements. In addition, the intimate knowledge of a small area may be more beneficial than the resources gained from utilizing a larger area. Home range estimates were larger when the time between captures was ≥ 25 days (Table 1.9; Figure 1.8), suggesting that birds use a small centralized home range over a relatively short temporal scale and this area either shifts or increases in size over the course of the winter.

Home range size and body condition did not significantly differ between burn-year and non-burn-year treatments. Although composition of food resources differed between burn-year and non-burn-year savannas, total seed densities were similar (Chapter 2). This supports the idea

that Henslow's Sparrows are generalists in their winter diet, at least within the confines of longleaf pine savannas (Fuller 2004, DiMiceli 2006).

There was no significant difference in body size between fire treatments or age classes (Table 1.8). Although a sex-related social hierarchy cannot be rejected from this analysis, there was little evidence to suggest that it exists, as other potential demographic, morphological, and ecological differences were negligible. Blood was collected from 150 Henslow's Sparrows over the two years of study. A future study will use these samples to sex birds (Griffiths et al. 1998) to test differences in sex ratios between burn-year and non-burn-year savannas.

Floaters

“Floater” is a term that applies to an individual that does not show a restricted home range. Empirical evidence suggests that a gradient exists between birds that show strong site fidelity and birds that wander widely (Brown 1999). Floaters in the population may reveal that a habitat is saturated and competition for resources is such that a subset of the population shows relatively large movement patterns (Brown 1969). Selection may maintain floaters in the population because this behavior can be advantageous in situations where waiting to acquire a territory is safer than fighting for the territory directly (Smith 1978).

We have evidence from three birds suggesting that floaters existed in this population of wintering Henslow's Sparrows. Although the bird originally captured in SH01 may simply represent an example of post-migration movement, this bird was not relocated upon two subsequent visits in March and early April. Whether the bird died, continued floating, or simply was not recaptured is not known. These floaters were caught hundreds of meters (450, 800, and 20000 m) away from their original capture location and interestingly, all three went from a

location with high bird density to a low density situation (although one went back), suggesting that limited high quality habitat may force birds into low quality habitat.

Winter Survival

One equally parsimonious best model in 2003-04 indicated a difference in apparent survival between HY/SY and AHY/ASY (Table 1.10). Differences between these groups were minor (< 0.05) and standard errors indicated overlap in estimated parameters. Age-related differences in habitat selection, body condition, home range size, and apparent survival in 2004-05 did not appear to exist, suggesting that age-related differences in survival in 2003-04 are probably negligible.

In 2003-04, apparent survival estimates were about 16% higher than in 2004-05 (Tables 1.11 and 1.13). If multiplied out by five months, which is an upper approximation for the amount of time a Henslow's Sparrow spends on its wintering grounds and offers a conservative estimate of over-winter survival, between-year differences in over-winter apparent survival probabilities are approximately 0.726 and 0.338 for 2003-04 and 2004-05 respectively.

Apparent survival probabilities estimates in a CJS model assume that the population is closed, i.e. with no immigration and emigration. Because of evidence that a small proportion of the population exists as floaters, apparent these survival estimates slightly underestimate actual survival.

Multiple models were equally parsimonious in estimating apparent survival probabilities during each year. One model assumed no effect of fire treatments while another suggested that differences in apparent survival between burn-year and non-burn-year plots exist (Tables 1.10 and 1.12). In the latter model, apparent survival probabilities were higher in burn-year savannas than in non-burn-year savannas in 2003-04, as expected based on bird densities. However, in

2004-05, the opposite was found. Indicated by relatively high standard errors, estimates overlap in 2004-05, possibly because of atypical situations in two of the three burn-year plots. First, LR05 was set fire annually during this study and bird density was lower in the second winter (Figure 1.5) and food resources were also lower during the second winter (Chapter 2) suggesting that an annual fire regime may have negative consequences on wintering Henslow's Sparrows. Second, AS01 had an extremely high density of Henslow's Sparrows in November and December 2004, but density dropped to nearly zero by the end of the winter, as indicated by a relatively large variance (Figure 1.5). Whether this was a result of mortality or dispersal is unknown; however, my results and radio telemetry studies by Bechtoldt and Stouffer (2005) and Thatcher (2003) suggest that this type of widespread dispersal would be extremely unusual. Abita Creek Flatwoods Preserve is the most recently restored savanna ecosystem in our study. *Panicum verrucosum*, an annual small-seeded species associated with disturbed soils (Plentovich et al. 1999), was much more abundant at this site than other sites (Appendix 4). Although structurally the habitat appeared ideal for wintering Henslow's Sparrows, the plant community may not yet be fully developed, as indicated by relatively low densities of seeding stems and seeds for a savanna that was burn-year during the previous growing season (Appendix 4).

For the reasons above, the 2003-04 data probably better represent differences in survival between fire treatments. Over-winter apparent survival probabilities are 0.761 and 0.623 for burn-year and non-burn-year savannas respectively, which are slightly higher than radiotelemetry estimates, as reported by Thatcher (2003).

The difference in survival estimates between fire treatments was less dramatic than differences in survival estimates between years. Thatcher (2003) also found this and related lower survival with extreme draught. In the two years of this study, the second winter was drier

than the first, but not to the degree that Thatcher (2003) reported. Low overnight temperatures (below freezing) were more frequent during the second year, when survival rates were lower, and the combination of this with drier conditions may be responsible for differences in survival between years. Henslow's Sparrows clearly increase fat storage during mid-winter when temperatures are at their lowest.

Survival estimates based this mark-recapture study and radiotelemetry (Thatcher 2003) suggest that Henslow's Sparrows suffer reduced survival in non-burn-year savannas because of causes not related to body condition. Thatcher (2003) determined that avian predators were responsible for a large percentage of mortality, suggesting that avian predation risk may be higher in non-burn-year savannas. Whether and why this is the case is not clear, but I will discuss a few possible non-mutually exclusive explanations.

Known Henslow's Sparrow avian predators, which include Loggerhead Shrikes (*Lanius ludovicianus*) (Thatcher 2003), and perhaps Northern Harriers (*Circus cyaneus*) (Hyde 1939), *Accipiter* spp., and *Falco* spp. (personal observation) may also be preying upon other small birds such as other sparrows and Sedge Wrens (*Cistothorus platensis*), small mammals, and large insects. Although I know of no study that has directly addressed the relative densities of predatory and prey species in response to time since fire (although Baldwin (2005) reports higher Swamp Sparrow (*Melospiza georgiana*) and Sedge Wren densities in savannas with greater herbaceous density), it is possible that non-burn-year savannas may attract higher densities of predators and wintering Henslow's Sparrows are indirect victims.

Another explanation of increased predation risk in non-burn-year savannas may relate to differences in vegetation structure. Henslow's Sparrows likely forage on the ground and travel between gaps between bunch-grasses (e.g. *Ctenium aromaticum*, *Muhlenbergia expansa*,

Andropogon spp., *Schizachyrium* spp, and *Panicum* spp.). Non-burn-year savannas have higher densities of vegetation and litter (Chapter 2; Bechtoldt and Stouffer 2005), which may reduce the number of pathways between clumps of vegetation. This could result in birds using available pathways more frequently or may occasionally force them into the open, thus increasing their predation risk (Thatcher 2003).

It is also possible that body condition estimates may not represent true habitat-related differences because heavier than expected birds and/or birds suffering from reduced condition are most likely to be captured by predators. Therefore, by removing these birds from the population, body condition estimates appear essentially equivalent. Another possibility is that birds in non-burn-year savannas are leaner than expected in response to increased predation pressures, thus the interaction between predation risk and food abundance may lead to observed similarities in body condition (Gosler et al. 1995).

Finally, differences in searching rates for food resources could lead to differences in survival probabilities. It is generally accepted that increased searching rates can lead to reduced vigilance and can decrease survival probabilities (Mangel and Clark 1986). In addition to greater densities of herbaceous vegetation potentially obscuring views of aerial predators, birds may spend more time searching for preferred food items in non-burn-year savannas, thus leading to reduced survival despite possessing equivalent condition to birds in burn-year savannas (Whittingham et al. 2004).

Arrival and Departure Patterns

In 2004-05, the most parsimonious models estimating apparent survival were partially time-dependent; apparent survival probabilities were lower in the first sampling interval compared to other sampling intervals (Table 1.13). An equally parsimonious model suggested

that apparent survival probabilities interacted between time and fire treatments, with non-burn-year savannas having lower apparent survival than burn-year savannas during the first sampling interval, but vice versa for the remainder of the winter (Table 1.13). This helps explain why bird density increases from November to December in burn-year savannas, but not non-burn-year savannas (Figure 1.4). Differences in apparent survival between these sampling periods are best explained by post-migration movements and not food (i.e. seed) availability, which is higher during this period than mid-winter and late-winter periods (unpublished data).

In contrast to 2004-05, the time constant models were the most parsimonious in 2003-04. It is likely that we did not begin our sampling effort early enough to detect post-migration movements in 2003-04. On 23 October 2003 we found a relatively high density of Henslow's Sparrows compared to the same period in 2004 (Figure 1.10), suggesting that birds arrived earlier in 2003 than in 2004, which explains differences in the detection of post-migration movements between years. Patterns of winter arrival may be adjusted by proximate mechanisms and are likely the result of differences in regional weather patterns among years, which can delay or accelerate fall migration (Akesson et al. 2002, Schaub et al. 2004), especially in short-distance migrants (Butler 2003, Marra et al. 2005)

Cues for spring migration are driven by endogenous urges to breed; concordantly age and sex classes can differ in their respective timing of migration (Stewart et al. 2002). Year-to-year variation also exists; major frontal systems have been shown to inhibit northward progression (Bearhop et al. 2004) and long-term studies have indicated a relationship between the timing of spring migration and climate change (Marra et al. 2005). In this study, more birds remained on their wintering grounds in 2005 than in 2004 (Figure 1.11), even though the sampled habitat contained fewer birds. Also, a higher percentage of SY birds were found in late March and April

than in the wintering population that we sampled (81.8 % compared to 68.0%), suggesting an SY delayed departure strategy that is consistent with many passerine species in North America (Stewart et al. 2002).

Pre-migration movements did not appear to occur in this population (Figure 1.9). Most of the birds captured at the end of the season were recaptures from earlier in the season or were captured on the periphery of the plot and could have been overlooked earlier in the winter. In 2003-04, however, several new “core” birds appeared in late March. I believe that these were birds that were displaced by a nearby early spring fire at WMA4, the largest patch of continuous savanna at Lake Ramsay WMA (Table 1.2). McNair (1998) observed many Henslow’s Sparrows fleeing an early spring fire and it is likely that they search out suitable habitat nearby.

Levels of body condition during the last sampling period were similar between fire treatments, between age classes, and body condition during the first sampling period (Figure 1.6) and fat scores and mass-corrected body size were lower in late March/April than in the first and last sampling period (unpublished data). This is consistent with the hypothesis that accumulation of body fat prior to spring migration may not be crucial for short-distance migrants (Long and Stouffer 2003).

Between-year Site Fidelity

Based on our limited number of between-year recaptures (9 birds returned out of 154), Henslow’s Sparrows appear to show between-year site fidelity, despite significant differences in habitat structure and food availability between years. Although it is difficult to relate current observations to historical patterns because of extreme habitat loss, this suggests that natural selection has maintained site fidelity because the benefits of returning to a potentially poor habitat are greater than the costs, as supported by body condition estimates and relatively small

differences in survival between burn-year and non-burn-year savannas. Site familiarity may be extremely important, as locating irregularly positioned escape refugia, which includes shrubs and possibly burrows, may be essential to avoid avian predation (personal observation, Carrie et al. 2002).

Inconsistent with the idea of between-year site fidelity, although not mutually exclusive, is that many Henslow's Sparrows exhibit post-migration movements. I would not expect to detect post-migration movements in birds that show between-year site fidelity because locating this site probably takes place relatively quickly upon arrival. Although differences in post-migration movements were not detected between immature and adult birds, limited sample size and statistical power coupled with infrequent sampling during this period may have constrained the ability to discern this pattern. Site familiarity in adult birds may outweigh the benefits of attempting to locate a potentially higher quality habitat, as changes in winter habitat are unpredictable from one year to the next. On the other hand, immature birds without prior knowledge of nearby microhabitats may roam until an available suitable patch is located.

Conclusions

Understanding the role of the winter season on Henslow's Sparrow populations is integral to their conservation. Habitat-mediated differences in bird density does not match those of many Neotropical migrants (Parrish and Sherry 1994, Stutchbury 1994, Wunderle 1995) because a class-structured hierarchy does not appear to exist for Henslow's Sparrows. Outward appearance and similarity in sizes among age and sex classes are likely an important explanatory factors (Langmore and Bennett 1999). For instance, sexually dimorphic species like American Redstarts and Black-throated Blue Warblers (*Dendroica caerulescens*) show strong class mediated social hierarchies (Parrish and Sherry 1994, Johnson and Sherry 2001, Bearhop et al. 2004), while

several sexually monomorphic species like Ovenbirds (*Seiurus aurocapillus*) (Strong 2000) and Hermit Thrushes (*Catharus guttatus*) (Brown et al. 2002) do not.

Growing season fires may be better for Henslow's Sparrow populations than dormant season fires, which was also suggested by Thatcher (2003) and Tucker and Robinson (2003). Compared to dormant season fires, advantages of growing season fires include the capacity to support higher bird densities (Figure 1.5), support more birds in future winters not following a fire (Tucker and Robinson 2003), it likely mimics the historic pattern of fire (Olson and Platt 1995, Platt 1999, Drewa et al. 2002), and does not eliminate that land as habitat for the remainder of the winter (McNair 1998). Growing season fires also encourage grass seed production, forb growth, and reduces shrub growth relative to dormant season fires (Streng et al. 1993, Glitzenstein et al. 2003).

A two-year prescribed fire rotation with approximately half of the available habitat set fire each year appears to be an effective management strategy for wintering Henslow's Sparrows. All else equal, implementing a two-year rotation may be a better management strategy than a three-year (or longer) time interval between prescribed fires. A larger percentage of savanna habitat set fire in a landscape each year would result in a larger number of Henslow's Sparrows that can be supported at a higher over-all survival rate. Land managers must consider their specific restoration goals, which unavoidably go beyond Henslow's Sparrow management and necessarily involves the management of a diverse and unique assemblage of animal and plant species.

CHAPTER 2. THE EFFECTS OF FIRE ON HABITAT STRUCTURE, FOOD AVAILABILITY, AND PLANT SPECIES COMPOSITION AND THEIR RESPECTIVE IMPORTANCE IN PREDICTING HENSLOW'S SPARROW DENSITIES

INTRODUCTION

Henslow's Sparrows (*Ammodramus henslowii*) winter throughout the longleaf pine (*Pinus palustris*) ecosystem and prefer savannas that were set fire during the previous growing season (Chapter 1, Plentovich et al. 1999, Carrie et al. 2002, Thatcher 2003, Tucker and Robinson 2003, Bechtoldt and Stouffer 2005). One thought is that by burning savanna vegetation, a structural arrangement is created for Henslow's Sparrows to secretively and efficiently forage on the ground while avoiding predation, much like the behavior of a mouse, suggesting that structure is ultimately important. Thatcher (2003) observed that avian predation plays an important role in winter mortality and, therefore, Henslow's Sparrows may utilize habitats in which avoiding detection or escaping predation is most efficient. In contrast, Bechtoldt and Stouffer (2005) and Tucker and Robinson (2003) suggest that although habitat structure is important, seed-bearing stalk density best predicts Henslow's Sparrow density because of particular food resources that closely associate with recent fire. While these studies provide significant contributions to the understanding of wintering Henslow's Sparrow ecology, a thorough investigation of plant species composition and seed abundance relative to Henslow's Sparrow density has not been conducted. Specifically *Rhynchospora* spp., which is a dominant genus and has been shown to be a significant component of the winter diet of Henslow's Sparrows (Fuller 2004), was not included in previous studies.

Because it is important to understand how habitat characteristics relate to winter Henslow's Sparrow to develop effective management strategies, I assessed the understory physiognomy to determine the relative importance of habitat structure, species composition, and

food (seed) availability to wintering Henslow's Sparrows. Based on previous studies, the abundance of particular seeds associated with recent fire appears to be an important predictor of Henslow's Sparrow density (Thatcher 2003, Bechtoldt and Stouffer 2005). Because the composition and relative abundance of herbaceous species dramatically changes across the wintering range of Henslow's Sparrows (Peet and Allard 1993), I predicted that differences in habitat structure are more important in predicting Henslow's Sparrows densities. This study was performed in conjunction with J. DiMiceli, who conducted food selection experiments on captive Henslow's Sparrows and analyzed fecal samples from wild caught birds to determine the diet of wintering Henslow's Sparrows.

METHODS

Study Site

This study was conducted in the Florida Parishes of southeastern Louisiana during the winters of 2003-04 and 2004-05. Ten 2.25 ha study plots in longleaf pine savannas were sampled, distributed among four study sites (Chapter 1, Table 1.1).

Habitat Structure

For continuity and ease of comparison, I followed the methods used by Bechtoldt (2002). Because her study clearly revealed patterns in vegetation structure related to fire, I selected a reduced number of data points to measure within a 2.25 ha plot. Regardless, patterns in structure were similar to Bechtoldt (2002) (see Results). Five random 113 m² non-overlapping circles were selected within each 2.25 ha plot. Within each circle, nine points were spaced along an x-y axis, three meters apart, starting at and including the x-y origin. At every point, a two cm wide pole was placed to measure the number of herbaceous "hits" (scale of 0-10+) between vertical

increments of 0-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, and 40-50 cm. A “hit” was a blade of dead or living vegetation that contacted the pole.

Shrub percent cover and tree density was estimated within each 113 m² circle. Trees were categorized based on species (longleaf pine, other *Pinus* spp., and deciduous) and by size class (< 7.5 cm dbh or > 7.5 cm dbh and longleaf pines < 1 m tall). Because tree density was not shown to influence Henslow’s Sparrow abundance, nor was it related to time since fire (unpublished data, Bechtoldt 2002), these data are not included in the analysis. Habitat structure data were collected in February and early March, before the onset of the growing season, consistent with other wintering Henslow’s Sparrow studies (Plentovich et al. 1999, Bechtoldt 2002, Carrie et al. 2002, Tucker and Robinson 2003).

Species Composition and Food Availability

Food availability estimations were conducted in late November of each year, approximately when the majority of Henslow’s Sparrows have arrived and settled into over-winter home ranges (Chapter 1). Each 2.25 ha plot was divided into six equally sized blocks and one random square meter of vegetation was collected from each block. Vegetation was clipped at the base with gardening sheers, stored in large garbage bags, and brought back to the lab for analysis. The bare 1 m² was then vacuumed with a Poulon Pro BVM200 leaf blower, set in reverse, to pick up fallen seeds. These samples were also stored in garbage bags and brought back to the lab for analysis.

In the lab, every stalk of seed-bearing vegetation was counted and identified to genus or species level. Identification to the most specific taxon was determined based on ease of identification and assumed ability of Henslow’s Sparrows to differentiate seeds of closely related species. For instance, several species in the genus *Andropogon* have seeds that I cannot easily

differentiate with a 7-30x dissecting scope, so it was assumed that Henslow's Sparrows would not select one species preferentially over another, thus seeds and stalks were counted at the genus level. On the other hand, many *Rhynchospora* species have similar-sized and -shaped stalks, but seeds are relatively unique, thus seeds and stalks were counted at the species level. The only species in which stalks were not counted was *Schizachyrium tenerium* because of the difficulty in separating seed-bearing stalks from vegetative growth. Seeds were also rarely present and therefore this species was excluded from all analyses.

I counted the number of seeds present on all flowering stalks. If more than 30 stalks were present in the 1 m² sample for a given taxon, I counted a haphazardly selected subsample of stalks to estimate seed abundance. These data provide an estimate of unavailable seeds at the beginning of the winter season and would be expected to fall sometime during the winter to become available to foraging Henslow's Sparrows. Some stalks do not drop all of their seeds by mid-March, when Henslow's Sparrows are departing for their breeding grounds, but the number of these remaining seeds is negligible relative to seed abundance in November (unpublished data).

Vacuumed samples contained a large amount of vegetation and dirt relative to seed content. Therefore, I stratified samples into six size and weight classes and subsampled from stratifications to make seed counting more accurate and efficient. I stratified vacuumed samples by first placing samples through a series of sieves, sizes 16 (1190 μm opening), 40 (420 μm opening), and 120 (125 μm opening). The sieves were stacked on top of each other and shaken by a Ro-Tap Testing Sieve Shaker for about five minutes to agitate and separate particles into three size classes. I then used an E.L. Erickson Products particle blower to separate each size class into two weight classes. Each stratified sample was then weighed ($\pm 0.01\text{g}$).

I subsampled from stratified samples that were too large to count in an effective time frame. I identified and counted seeds from at least four subsamples of each stratified sample and calculated the coefficient-of-variation (CV) using known-population variance estimates (Lohr 1999). I subsampled stratified samples until the CV for each common seed species was less than 0.4. This level precision was only applied to common species because rare seeds never reached this threshold unless the majority of the sample was counted (unpublished data). Common seeds were those that were present at least once per 1.0 g of a given stratified section. A common seed in one stratified section was not necessarily common in others because seed taxa were generally specific to particular size and weight classes. These vacuumed samples provided a snapshot of what was available to Henslow's Sparrows as they settle into winter home ranges.

Unavailable and available seeds were summed for each taxon to provide an estimate of seed density. Seeds that were small enough to pass through the size 40 sieve were not included in seed analyses for two reasons: 1) because of the likelihood that such small seeds were not sampled adequately using the vacuum method and 2) because it was assumed that granivorous birds would not be interested in such seeds. Most Poaceae, Cyperaceae, and Asteraceae were analyzed using this criteria (with the exception of *Rhynchospora pusilla* and *Eragrostis* spp.), while many forbs were not (e.g. *Agalinis* spp., *Eriocaulon* spp., *Drosera* spp., *Hypericum* spp., *Lobelia* spp., and *Rhexia lutea*).

Statistical Analysis

Residuals of dependent variables from all analyses were tested for normality and homogeneity using a Shapiro-Wilk test of normality and Levene's Test of homogeneity; transformations of the dependent variables were made when appropriate. Treatment means are presented as Least Squares Means (LSMEANS, SAS Institute 2003).

In this two-year prescribed fire rotation scenario, I tested the effect of time since fire (burn-year vs. non-burn year) on the density of seed-bearing herbaceous stalks (not including *Drosera* spp. L., *Utricularia subulata* L., and *Sisyrinchium* spp L.. because they were too small to adequately sample) was tested using a quasi Latin rectangle multivariate analysis-of-variance mixed-model (MANOVA). Time since fire was a fixed effect while year and plot were random effects. A parallel profiles test was used to determine differences in seed-bearing stalk composition between fire treatments and a coincident profiles test was used to determine differences in total seed-bearing stalk abundance. Univariate analyses testing the effect of fire on differences in seed-bearing stalk densities for each group were conducted.

A quasi Latin rectangle mixed-model MANOVA was also used to test the effects of time since fire on the density of all species of seeds combined (not including, *Eragrostis* spp., *Rhynchospora pusilla*, and small forb seeds). A parallel profiles test was used to determine differences in seed-bearing stalk composition between fire treatments and a coincident profiles test was used to determine differences in total seed-bearing stalk abundance. Univariate analyses testing the effect of fire on differences in seed densities for each group were conducted. Because the plot not set fire for three growing seasons (AS03) was determined to be an outlier, both years of data from this plot were excluded from this analysis. Seed-bearing stalk density was log transformed to meet assumptions of parametric statistics.

Associations of bird density (Chapter 1) and fire treatments with habitat structure, seed-bearing stalk density, and seed density were examined using non-metric multidimensional scaling (NMDS) (PROC MDS, SAS Institute 2003). A dissimilarity matrix using Euclidian distances among the 2.25 ha plots to ordinate relationships among plots (PROC DISTANCE, SAS Institute 2003). Stress, which is a measure of the badness-of-fit between the ordination and

measured ecological distances, was examined as a function of the number of axes (Kruskal 1964). Stress was low (between 0.06 and 0.09) with a two dimension representation for each of the three habitat categories (structure, seed-bearing stalk density, and seed density). Therefore, two dimensions were used to present relationships among plots.

Groups of seed-bearing stalks and seeds were constructed based on taxonomy and form. Seed and stalk densities were summed for each group and then log transformed. Tables 2.1 and 2.2 list the groups of seed-bearing stalks and seeds and their constituent species. Groups of stalks and seeds are not the same because similar stalks could yield a completely different type of food resource and vice versa.

Multiple regressions of seed-bearing stalk groups, seed groups, and structural measurements on each dimension were run to determine the important variables. The r^2 selection method was used to eliminate highly correlated and unimportant variables (PROC REG, SAS Institute 2003). Partial correlations of species groups and structural measurements were overlaid on NMDS ordination plots (PLOTIT MACRO, SAS Institute 2003).

Scores from dimensions one and two were regressed against bird density for each plot. This was done for each of the three habitat categories and an additional four models were run using combinations of habitat categories. I then used an information theoretic approach to rank regression models based on Akaike's Information Criterion (AIC) to determine which habitat category (structure, seed-bearing stalk density, or seed density) best predicted Henslow's Sparrow density (Burnham and Anderson 2002). Bird density was log transformed to meet assumptions of parametric statistics. Models were ranked according to Akaike's Information Criterion corrected for small sample size (AIC_c).

Table 2.1. Seed-bearing stalk groups and the taxa that comprise each group.

Panic-grass	Fire Grass	Dichanthelium	Andropogon-type	Unplumed Asteraceae
<i>Dichanthelium scabriusculum</i>	<i>Ctenium aromaticum</i>	<i>Dichanthelium</i> spp.	<i>Andropogon</i> spp.	<i>Balduina uniflora</i>
<i>Panicum</i> spp.	<i>Muhlenbergia expansa</i>		<i>Schizachyrium scoparium</i>	<i>Bidens mitis</i>
				<i>Boltonia diffusa</i>
				<i>Coreopsis linifolium</i>
				<i>Eryngium integrifolium</i>
				<i>Helianthus</i> spp.
Other Grass	Forb	Tall Sedges	Short Sedges	Plumed Asteraceae
<i>Anthraenantia</i> spp.	<i>Achalypha gracilens</i>	<i>Carex turgescens</i>	<i>Eleocharis</i> spp.	<i>Symplotrichium adnatus</i>
<i>Aristida</i> spp.	<i>Agalinis</i> spp.	<i>Fimbristylis</i> spp.	<i>Rhynchospora chapmanii</i>	<i>S. concolor</i>
<i>Digitaria</i> sp.	<i>Aletris</i> spp.	<i>Fuirena breviseta</i>	<i>R. ciliaris</i>	<i>S. dumosus</i>
<i>Eragrostis</i> spp.	<i>Crotalaria</i> spp.	<i>Rhynchospora cephalantha</i>	<i>R. fascicularis</i>	<i>S. patens</i>
<i>Gymnopogon brevifolius</i>	<i>Eriocaulon decangulare</i>	<i>R. compressa</i>	<i>R. filifolia</i>	<i>Ionactis linariifolius</i>
<i>Paspalum</i> spp.	<i>Hypericum</i> spp.	<i>R. corniculata</i>	<i>R. gracilentia</i>	<i>Eurybia hemispherica</i>
<i>Saccharum giganteum</i>	<i>Lechea</i> spp.	<i>R. elliotii</i>	<i>R. microcephala</i>	<i>Bigelowia</i> spp.
<i>Sporobolus junceus</i>	<i>Linum medium</i>	<i>R. globularis</i>	<i>R. oligantha</i>	<i>Eupatorium</i> spp.
<i>Tridens ambiguus</i>	<i>Lobelia</i> spp.	<i>R. inexpansa</i>	<i>R. plumosa</i>	<i>Euthamia caroliniana</i>
	<i>Ludwigia</i> spp.	<i>R. latifolia</i>	<i>R. pusilla</i>	<i>Liatris</i> spp.
	<i>Mitreola</i> sp.		<i>R. rariflora</i>	<i>Pityopsis graminifolia</i>
	<i>Oxypolis filiformis</i>		<i>Scleria hirtella</i>	<i>Solidago</i> spp.
	<i>Rhexia</i> spp.		<i>S. pauciflora</i>	<i>Carphephorus pseudoliatris</i>
	<i>Polygala ramosa</i>		<i>S. reticularis</i>	<i>Elephantopus tomentosus</i> .
	<i>Polypremum procumbens</i>		<i>Juncus trigonocarpus</i>	Unknown Asteraceae 1
	<i>Stylosanthes biflora</i>			Unknown Asteraceae 2
	<i>Tephrosia</i> spp.			
	<i>Xyris</i> spp.			
	Unknown 7, 20, 25, 30B, 31			

Table 2.2. Seed groups and the taxa that comprise each group.

Sedge	Rhynchospora	Large-seed Plumed Asteraceae	Small-seed Plumed Asteraceae	Forbs
<i>Carex turgescens</i>	<i>R. cephalantha</i>	<i>Carphephorus pseudoliatris</i>	<i>Bigelowia</i> spp.	<i>Balduina uniflora</i>
<i>Eleocharis</i> spp.	<i>R. chapmanii</i>	<i>Elephantopus tomentosus</i>	<i>Eupatorium</i> spp.	<i>Bidens mitis</i>
<i>Fimbristylis</i> spp.	<i>R. ciliaris</i>	<i>Eurybia hemispherica</i>	<i>Euthamia caroliniana</i>	<i>Boltonia diffusa</i>
<i>Fuirena breviseta</i>	<i>R. compressa</i>	<i>Ionactis linariifolius</i>	<i>Solidago</i> spp.	<i>Coreopsis linifolia</i>
<i>Juncus trigonocarpus</i>	<i>R. corniculata</i>	<i>Liatris</i> spp.	<i>Symplotrichium adnatus</i>	<i>Eryngium integrifolium</i>
<i>Scleria hirtella</i>	<i>R. elliottii</i>	<i>Pityopsis graminifolia</i>	<i>S. dumosum</i>	<i>Helianthus</i> spp.
<i>S. pauciflora</i>	<i>R. filifolia</i>	<i>Symplotrichium concolor</i>	<i>S. patens</i>	<i>Linum medium</i>
<i>S. reticularis</i>	<i>R. globularis</i>		Unknown seed G	<i>Oxypolis filiformis</i>
Unknown seed M	<i>R. gracilentia</i>			<i>Rhexia alifanus</i>
	<i>R. inexpansa</i>			<i>Stylosanthes biflora</i>
	<i>R. latifolia</i>			<i>Tephrosia</i> spp.
	<i>R. microcephala</i>			Unknown seeds C, E, F, Q, R, S, T, U, X, AA, AB
	<i>R. oligantha</i>			
	<i>R. plumosa</i>			
	<i>R. rariflora</i>			
Andropogon-type	Panic-grass	Other Grass	Fire Grass	
<i>Andropogon</i> spp.	<i>Panicum</i> spp.	<i>Anthaenantia</i> spp.	<i>Ctenium aromaticum</i>	
<i>Saccharum giganteum</i>	<i>Dichantherium</i> spp.	<i>Aristida</i> spp.	<i>Muhlenbergia expansa</i>	
<i>Schizachyrium</i> spp.	<i>Digitaria</i> sp.	<i>Gymnopogon brevifolius</i>		
		<i>Sporobolus junceus</i>		
		<i>Tridens ambiguus</i>		
		<i>Paspalum</i> spp.		
		Unknown seed D		

RESULTS

I tallied a total of 116 taxa of stalks and seeds that were used for analyses (Appendix 4). Ten seed taxa were found with no corresponding stalks. Therefore 103 taxa were used in the NMDS analysis for seed-bearing stalks. Twenty-three of these were stalks that produced seeds too small for analyses and five additional stalks were found with no seeds. Therefore 91 taxa were used in the NMDS analysis for seeds.

Fire significantly reduced the total number of seed-bearing stalks (Table 2.3; Figure 2.1). Although community composition did not significantly differ between burn-year and non-burn-year savannas (Table 2.4), the decrease in the total number of seed-bearing stalks was because of decreases in Short Sedge and Dichanthelium (Table 2.5; Figure 2.1). Fire also significantly increased the density of Fire Grass, Andropogon-type, and Unplumed Asteraceae (Table 2.5).

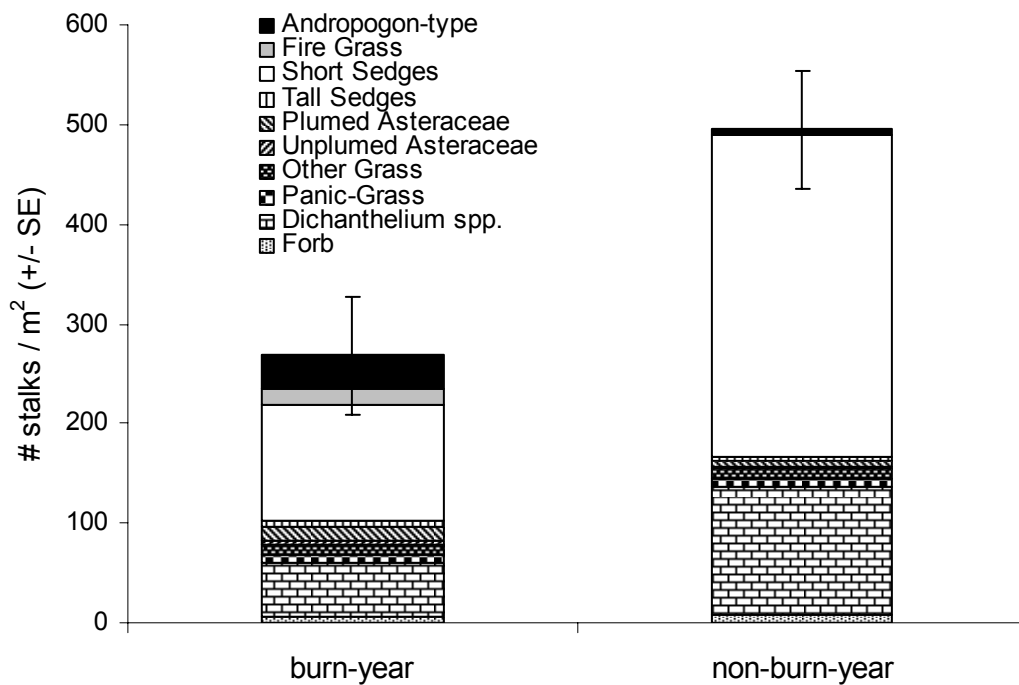


Figure 2.1. The density and composition of seed-bearing stalk groups in burn-year and non-burn-year plots (\pm SE for total seed density).

Table 2.3. The effects of fire treatments on total seed-bearing stalk density.

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	1	8	14.01	0.0057

Table 2.4. The effects of fire treatments on seed-bearing stalk composition. The six least common groups were combined because there were not enough degrees of freedom to perform this test using all ten groups.

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	4	5	1.77	0.2722

Table 2.5. Univariate tests to show the effect of time since fire on seed-bearing stalk group densities and whether fire significantly increased (+), decreased (-), or had no effect (ns).

Group	df _{effect}	df _{error}	F	Pr > F	Trend
Andropogon-type	1	8	17.09	0.0033	+
Fire Grass	1	8	8.33	0.0203	+
Short Sedge	1	8	1.66	0.2342	ns
Tall Sedge	1	8	2.83	0.1310	ns
Plumed Asteraceae	1	8	0.96	0.3566	ns
Unplumed Asteraceae	1	8	7.72	0.0240	+
Other Grass	1	8	0.01	0.9208	ns
Panic-grass	1	8	1.58	0.2438	ns
Dichanthelium	1	8	5.66	0.0447	-
Forb	1	8	0.21	0.6585	ns

The density of seeds was not significantly different between burn-year and non-burn-year savannas (Table 2.6; Figure 2.2); however, composition shifted from an Andropogon-type dominated community to a *Rhynchospora* spp. dominated community (Table 2.7; Figure 2.2). The density of Fire Grass and Large Plumed Asteraceae was also higher in burn-year savannas (Table 2.8).

Table 2.6. The effect of time since fire on seed density.

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	1	7	0.09	0.7692

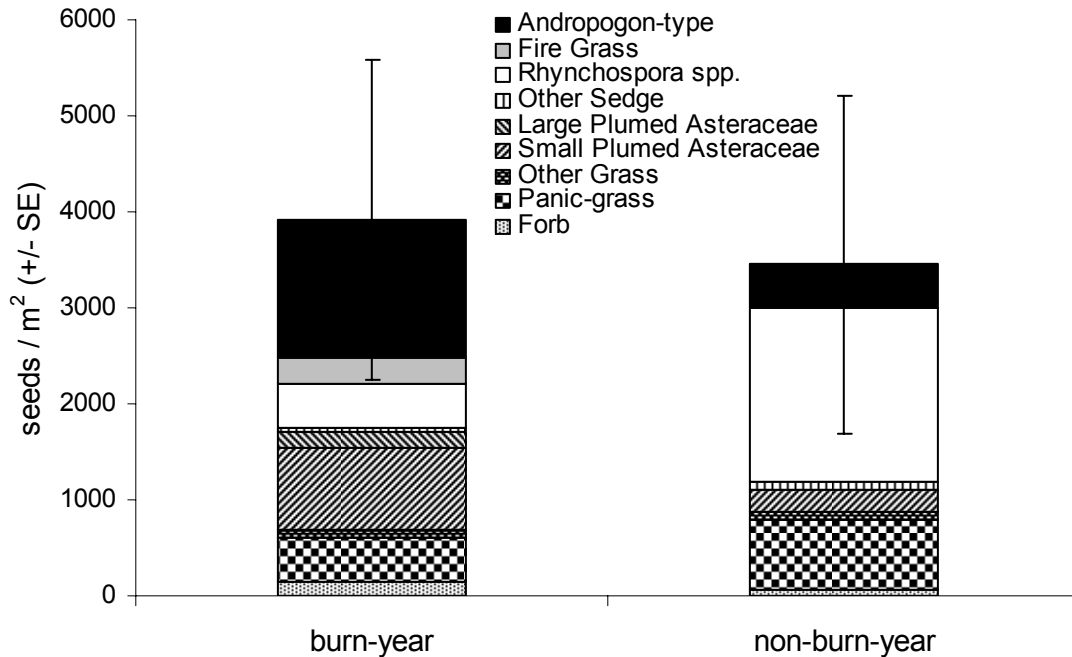


Figure 2.2. The density and proportion of seed groups in burn-year and non-burn-year plots (\pm SE).

Table 2.7. The effects of fire treatments on seed composition. The Large Plumed and Small Plumed Asteraceae were combined and the groups Panic-grass, Other Grass, Other Sedge, and Forb were combined because there were not enough degrees of freedom to perform this test using all nine groups.

Effect	df _{effect}	df _{error}	F	Pr > F
Fire	4	5	7.53	0.0240

Table 2.8. Univariate tests to show the effect of time since fire on seed group densities and whether fire significant increased (+), decreased (-), or had no effect on densities (ns).

Group	df _{effect}	df _{error}	F	Pr > F	Trend
Andropogon-type	1	8	1.45	0.2630	ns
Fire Grass	1	8	14.48	0.0052	+
Rhynchospora	1	8	6.51	0.0341	-
Other Sedge	1	8	0.06	0.8149	ns
Large Plumed Asteraceae	1	8	11.88	0.0087	+
Small Plumed Asteraceae	1	8	0.29	0.6073	ns
Other Grass	1	8	0.32	0.5869	ns
Panic-grass	1	8	4.65	0.0631	ns
Forb	1	8	0.51	0.4973	ns

NMDS was used to determine the associations among plots in two dimensions based on seed-bearing stalk composition and density (Figure 2.3). Bird density increases with increasing Andropogon-type, Fire Grass, Plumed Asteraceae, and Unplumed Asteraceae groups and these groups decrease with increasing time since fire. Forb, Short Sedge, and Other Grass increase with time since fire. Tall Sedge, Dichanthelium, and Panic-Grass do not correlate with bird density or time since fire. The cluster of burn-year plots largely overlaps the cluster of non-burn-year plots.

A MLR revealed that Panic-grass, Other Grass, Plumed Asteraceae, and Unplumed Asteraceae groups are important in explaining dimension 1 ($r^2 = 0.805$; Table 2.9). Fire Grass, Tall Sedge, and Dichanthelium groups are important groups in explaining dimension 2 ($r^2 = 0.551$; Table 2.10).

Table 2.9. Seed-bearing stalk groups that associate with dimension 1.

Source	df	SS	MS	F	Pr > F
Model	4	17.40	4.35	15.50	< 0.0001
Error	15	4.21	0.28		
Corrected Total	19	21.61			

Variable	df	Slope	SE	t	Pr > t
Panic-grass	1	0.591	0.103	5.76	< 0.0001
Other Grass	1	0.193	0.047	4.09	0.0010
Rhynchospora	1	0.110	0.044	2.49	0.0248
Forb	1	-0.538	0.125	-4.29	0.0006

Table 2.10. Seed-bearing stalk groups that associate with dimension 2.

Source	df	SS	MS	F	Pr > F
Model	3	10.13	3.38	6.55	0.0043
Error	15	8.26	0.52		
Corrected Total	19	18.39			

Variable	df	Slope	SE	t	Pr > t
Fire Grass	1	-0.530	0.168	-3.15	0.0062
Dichanthelium	1	-0.762	0.202	-3.77	0.0017
Tall Sedge	1	0.412	0.181	2.28	0.0369

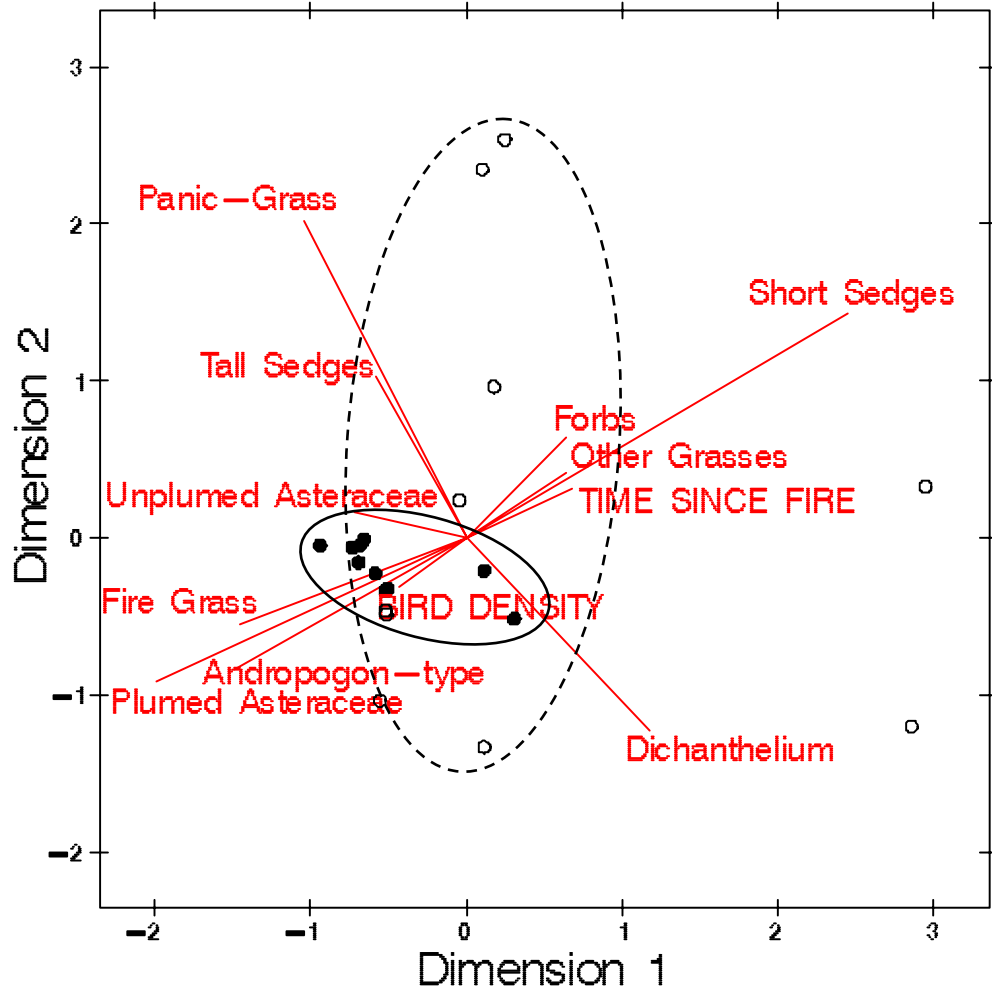


Figure 2.3. NMDS plot of relationships among plots based on taxa-specific seed-bearing stalk densities. Seed-bearing stalk group scalings are overlaid to show relationships among stalk groups, time since fire, and bird density and illustrate their partial correlations with dimensions 1 and 2. Filled circles are burn-year plots and open circles are non-burn-year plots. The ovals cluster plots by fire treatment (solid oval = burn-year, dashed oval = non-burn-year).

NMDS was used to illustrate the associations among plots in two dimensions (stress = 0.067) based on seed composition and density (Figure 2.4). Time since fire, Rhynchospora, and Other Sedge show similar partial correlations and negatively correlate with bird density, Fire Grass, Plumed and Unplumed Asteraceae, Panic-grass, Andropogon-type and Other Grass. The

Forb group shows no correlation with other seed groups, time since fire, or bird density. The cluster of burn-year plots largely overlaps the cluster of non-burn-year plots.

A multiple linear regression (MLR) was used to determine the relative effects of species guilds on each of the two dimensions. Andropogon-type, Fire Grass, and Rhynchospora groups were determined to be important on dimension 1 ($r^2 = 0.668$; Table 2.11). Rhynchospora, Forb, Other Sedge, and Small Plumed Asteraceae groups were determined to be important on dimension 2 ($r^2 = 0.543$; Table 2.12).

Table 2.11. Source table of MLR and seed guilds that associate with dimension 1.

Source	df	SS	MS	F	Pr > F
Model	3	18.79	6.26	10.72	0.0004
Error	16	9.35	0.58		
Corrected Total	19	28.14			

Variable	df	Slope	SE	t	p
Andropogon-type	1	0.466	0.180	2.58	0.0200
Fire Grass	1	0.189	0.083	2.28	0.0367
Rhynchospora	1	-0.121	0.081	-1.50	0.1530

Table 2.12. Seed guilds that associate with dimension 2.

Source	df	SS	MS	F	Pr > F
Model	4	6.45	1.61	4.47	0.0141
Error	15	5.41	0.36		
Corrected Total	19	11.86			

Variable	df	Slope	SE	t	p
Rhynchospora	1	0.164	0.081	2.04	0.0595
Forb	1	0.582	0.166	3.50	0.0032
Other Sedge	1	-0.581	0.277	-2.09	0.0536
Small Plumed Asteraceae	1	-0.300	0.101	-2.98	0.0094

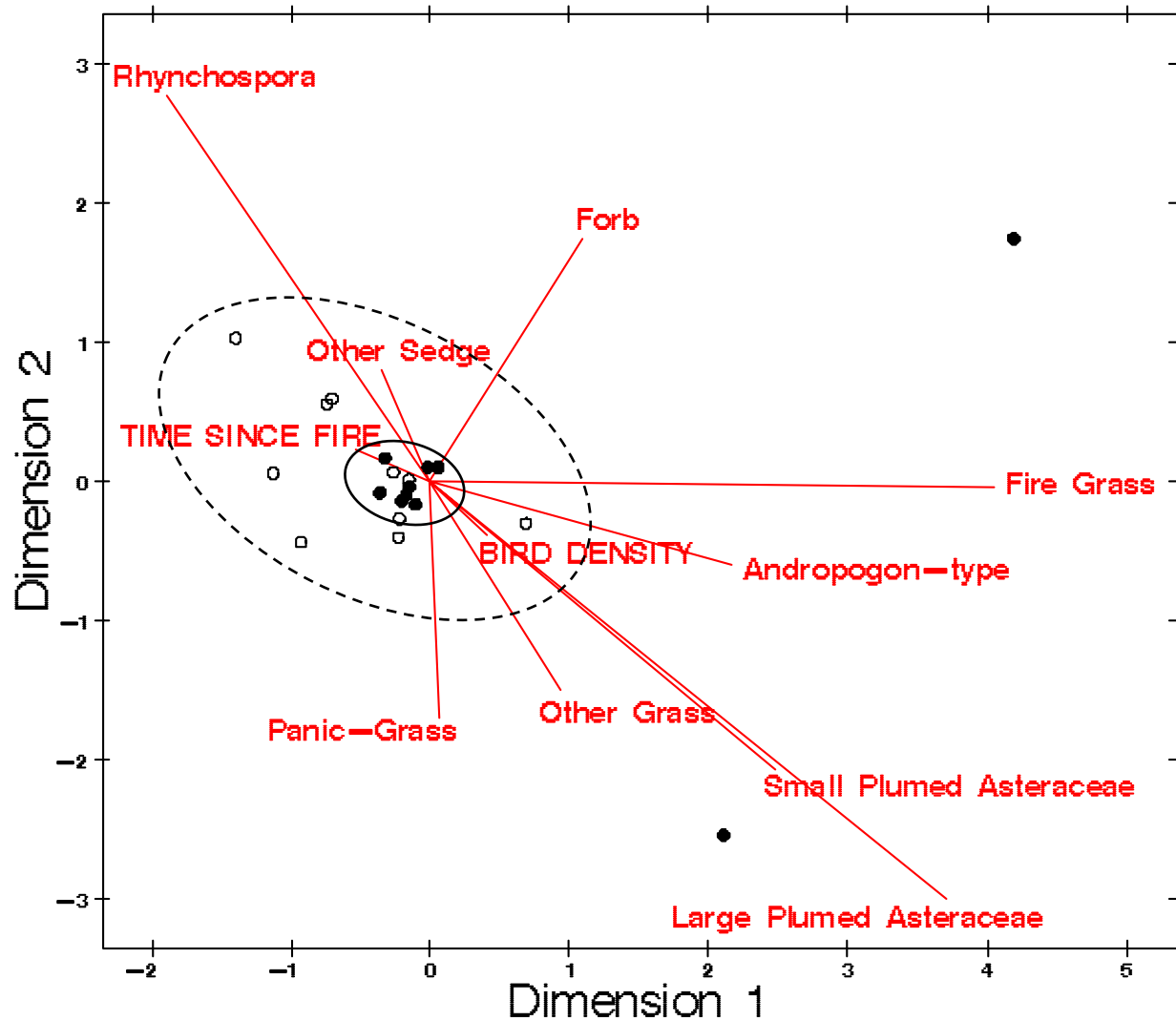


Figure 2.4. NMDS plot of relationships among plots based on taxa-specific seed densities. Seed group scalings are overlaid to show relationships among seed groups, time since fire, and bird density and illustrate their partial correlations with dimensions 1 and 2. Filled circles are burn-year plots and open circles are non-burn-year plots. The ovals cluster plots by fire treatment (solid oval = burn-year, dashed oval = non-burn-year).

NMDS was used to determine the associations among plots in two dimensions (stress = 0.064) based on structural characteristics (Figure 2.5). Bird density increases with increasing Andropogon-type, Fire Grass, Plumed Asteraceae, and Unplumed Asteraceae groups and decreases with increasing time since fire, Forb, Short Sedge, and Other Grass. Tall Sedge,

Dichanthelium, and Panic-Grass do not strongly correlate with bird density or time since fire. The cluster of burn-year plots largely overlaps the cluster of non-burn-year plots.

A MLR was used to determine that ground-level herbaceous density and the number of shrubs are important in explaining dimension 1 ($r^2 = 0.893$; Table 2.13). Ground-level herbaceous density, the number of shrubs, and height of herbaceous vegetation were important variables in explaining dimension 2 ($r^2 = 0.868$; Table 2.14).

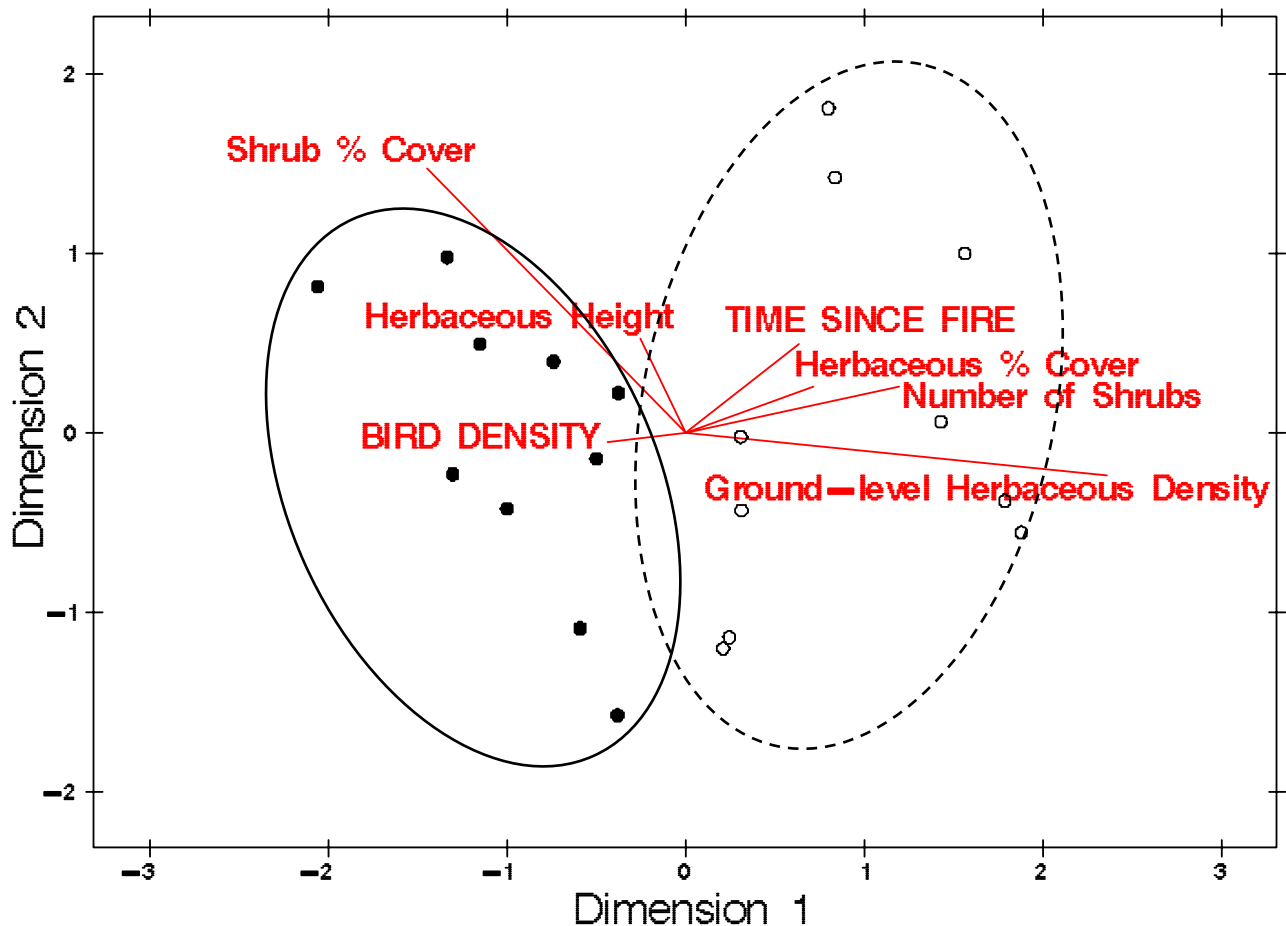


Figure 2.5. NMDS plot of relationships among plots based on habitat structural parameters. Structural parameter scalings are overlaid to show relationships among stalk guilds, time since fire, and bird density and illustrate their partial correlations with dimensions 1 and 2. Filled circles are burn-year plots and open circles are non-burn-year plots. The ovals cluster plots by fire treatment (solid oval = burn-year, dashed oval = non-burn-year).

Table 2.13. Habitat structure parameters that associate with dimension 1.

Source	df	SS	MS	F	Pr > F
Model	2	21.74	10.87	71.13	< 0.0001
Error	17	2.60	0.15		
Corrected Total	19	24.34			

Variable	df	Slope	SE	t	Pr > t
Ground-level Density	1	0.487	0.048	10.23	< 0.0001
Number of Shrubs	1	0.308	0.055	5.61	< 0.0001

Table 2.14. Habitat structure parameters that associate with dimension 2.

Source	df	SS	MS	F	Pr > F
Model	3	13.59	4.53	35.00	< 0.0001
Error	16	2.07	0.13		
Corrected Total	19	15.66			

Variable	df	Slope	SE	t	Pr > t
Ground-level Density	1	0.176	0.049	3.63	0.0022
Number of Shrubs	1	0.136	0.051	2.65	0.0174
Herbaceous Height	1	2.828	0.278	10.17	< 0.0001

Table 2.15. Model selection results of habitat parameters regressed against bird densities, ranked according to Akaike's Information Criterion corrected for small sample size (AIC_c). ΔAIC_c is AIC_c between each model and the best model.

Model	AIC_c	ΔAIC_c	AIC_c weight	Model Likelihood	# of parameters
Structure	23.1	0.0	0.881	12.4	4
Structure + Seed	29.4	6.3	0.038	11.0	6
Structure + Stalk	29.8	6.7	0.031	11.4	6
Seed	30.1	7.0	0.027	19.5	4
Stalk	30.5	7.4	0.022	19.8	4
Seed + Stalk	36.5	13.4	0.001	18.1	6
Structure + Seed + Stalk	38.9	15.8	0.000	9.8	8

Model selection was used to determine whether habitat structure, seed-bearing stalk density, or seed density best predict Henslow's Sparrow densities. Table 2.15 lists all the models, ranked according to lowest ΔAIC_c . The habitat structure model was the most parsimonious in explaining Henslow's Sparrow densities. The habitat structure model is more

than ten times as well supported than the seed-bearing stalk density model and there is “considerable support” that real differences between these models occur. All other models are even less likely candidates.

DISCUSSION

The response of savanna vegetation to fire has been well documented (Lewis and Harshbarger 1976, Glitzenstein et al. 1995, Olson and Platt 1995, Brewer 1999, Glitzenstein et al. 2003, Tucker and Robinson 2003, Fuller 2004, Bechtoldt and Stouffer 2005). Consistent with previous studies, I found that burning savanna vegetation reduces ground-level vegetation density, reduces the number of shrubs (Table 2.13; Figure 2.5), and stimulates flowering of several species of grasses, sedges, and forbs (Tables 2.5 and 2.8; Figures 2.2 and 2.4). In particular, *Andropogon* spp., *Schizachyrium scoparium*, *Ctenium aromaticum*, and *Muhlenbergia expansa* increased flowering when burned (Table 2.5; Figure 2.2). *Dichanthelium* spp. and *Rhynchospora* spp. each showed reduced seed-bearing stalk density leading to lower seed densities resulting from prescribed fires (Tables 2.5 and 2.8; Figures 2.1 and 2.2).

Wintering Henslow’s Sparrows associate with savannas that were set fire during the previous growing season (Chapter 1; Plentovich et al. 1999, Carrie et al. 2002, Thatcher 2003, Tucker and Robinson 2003, Bechtoldt and Stouffer 2005). Although structural changes associated with burning vegetation have shown to be important, previous studies have associated the density of Henslow’s Sparrows with a guild of pyrogenic grass species, specifically *Ctenium aromaticum* and *Muhlenbergia expansa* (Tucker and Robinson 2003, Bechtoldt and Stouffer 2005). These species are highly sought after (DiMiceli 2006), however, they comprise only a small proportion of the seed-bearing stalk or seed densities and may not be as important predicting bird densities as previously thought (Figures 2.1 and 2.2). Unlike previous studies, I

found that Henslow's Sparrow densities are best predicted by changes in habitat structure, not seed or seed-bearing stalk densities (Table 2.15). There are several explanations for this discrepancy. First, unlike other studies concerning wintering Henslow's Sparrows, I sampled in upland and flatwoods savannas. Plant community compositions between these savanna types are distinctive because of differences in soil moisture and other edaphic properties (Drewa et al. 2002, Glitzenstein et al. 2003). Second, this study provides a more detailed investigation of plant species composition and seed abundance than previous studies associating bird densities to habitat characteristics.

There several explanations to why habitat structure, and not food availability, best explains Henslow's Sparrow densities. Birds that occupy non-burn-year savannas appear to shift their diet in response to food availability (Fuller 2004), but do not suffer reduced body condition or the need for larger home range sizes (Chapter 1; Thatcher 2003, Bechtoldt and Stouffer 2005). This suggests that Henslow's Sparrows are generalist foragers, which is supported by the diet of wild caught birds; these studies found dozens of species of grass, sedge, and forb seeds including *Rhynchospora* spp., *Dichanthelium* spp., *Panicum* spp., *Scleria* spp., and *Ctenium aromaticum* (Fuller 2004, DiMiceli 2006). Some of these taxa show an increase in seed availability in burn-year savannas, but others show an increase in abundance in non-burn-year savannas (Figures 2.2, 2.3, and 2.4; Fuller 2004).

Ground-level herbaceous density and the number of shrubs were the most important variables in explaining habitat structure (Table 2.9). Both variables increase with time since fire and cause a decrease in bird density (Tables 2.9; Figure 2.5). Bird survival estimates were higher in burn-year savannas (Chapter 1; Thatcher 2003) and differences in predation risk are consistent with differences in habitat structure (see Chapter 1 for this discussion). Land

managers can easily manipulate the structural components of longleaf pine savannas through frequent prescribed growing season fires, which is critical to the conservation of Henslow's Sparrows.

CHAPTER 3. CONCLUSIONS AND IMPLICATIONS FOR CONSERVATION

In a two-year prescribed fire rotation, Henslow's Sparrow densities are higher in the winter following a fire, largely explained by changes in habitat structure. Their diet shifts in response to food availability and this does not appear to negatively affect body condition. Lower survival rates in non-burn-year savannas therefore do not appear to be resource-related and suggest that changes in habitat structure may lead to differences in predation risk. While this may be important in explaining patterns of habitat selection, differences in survival between years is more dramatic than between fire treatments (burn-year vs. non-burn-year), suggesting that additional ecological mechanisms may be driving habitat selection. With consideration of the wide-spread destruction of the longleaf pine ecosystem, differences in survival between fire treatments may have been more dramatic in historical times and such evolutionary selection processes may be not be operating today.

Supporting a greater density of birds at a higher survival rate clearly requires prescribed fires as a management tool. Fires conducted in April to June are concordant with historical fire regimes and would not disrupt wintering Henslow's Sparrows. All else equal, implementing a two-year fire rotation may be a better management strategy than a three-year (or longer) time interval between fires, as it would maintain a 0.5:0.5 ratio of burn-year to non-burn-year savannas over the long-term.

Despite increases in land area enrolled in the Conservation Restoration Program (CRP), grassland habitats have declined across the breeding range of Henslow's Sparrows (Murphy 2003) and it remains one of the fastest declining birds in North America (Sauer et al. 2005). The rate of population decline exceeds the rate of breeding habitat loss, perhaps because of sensitivities to fragmentation (Winter and Faaborg 1999), or loss of winter habitat (Newton

1998). Winter habitat has reduced to less than 3% of its historical range (Varner and Kush 2004) and its loss may be significant in explaining declines in Henslow's Sparrow populations.

In Louisiana, about 2,000 ha of land is available for restoration to a longleaf pine community and would cost between \$500,000 and \$630,000 to plant vegetation and implement a fire regime (Dolan 2001). Assuming a two-year fire rotation, this land could provide habitat for about 3,000 additional Henslow's Sparrows, about 4% of the estimated world population (Smith 1999). If restoration strategies such as this were to be implemented across the Gulf and southern Atlantic states, a significant number of wintering Henslow's Sparrows could be supported. In addition, habitat would be provided for numerous other rare and endangered species (Tucker et al. 2003), including dozens of plant and insect species, Red-cockaded Woodpecker (*Picoides borealis*), Bachman's Sparrow (*Aimophila aestivalis*), Bachman's Fox Squirrel (*Sciurus niger bachmani*), Gopher Tortoise (*Gopherus polyphemus*), Flatwoods Salamander (*Ambystoma cingulatum*), Striped Newt (*Notophthalmus perstriatus*), Dusky Gopher Frog (*Rana capito*), Eastern Indigo Snake (*Drymarchon corais couperi*), Eastern Diamondback Rattlesnake (*Crotalus adamanteus*), and Florida Pine Snake (*Pituophis melanoleucus mugitus*). Land managers must consider their specific restoration goals, which unavoidably go beyond Henslow's Sparrow management and necessarily involves the management of a diverse and unique assemblage of animal and plant species.

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APPENDIX 1. LIST OF VOLUNTEERS THAT HELPED CATCH HENSLOW'S SPARROWS

Barrett Ainsworth	Andrew Guempel	Laura Palasz
Laura Alexander	Rebekka Guillory	Dave Patton
Blake Amos	Ken Hackman	Thais Perkins
David Anderson	Tiffany Hawkins	Erica Perrer
Patrick Anthony	Rachel Holbertson	Jed Pitre
Greg Badon	Brooke Hopkins	Lisa Pitre
Jenny Baggett	Glen Hurst	Kevin Prevot
Heather Baldwin	Evelyn Jackson	Joe Ramspott
Mike Baldwin	Dean Jenson	Eugenie Regan
Doug Barron	Willa Jo Jenson	Steve Regan
Phred Benham	Delia Johnson	Leigh Ann Reynolds
David Billesbach	Ivan Johnson	Ceci Ripoll
Chris Brantley	Brooks Kennedy	Lisa Robichaux
Adriana Bravo	Will Kennedy	Teri Root
Dave Brown	Mary Jo Krieger	Lea Scott
Maria Brown	Christie Landry	Jane Stevens
Winston Caillouet	Keri Landry	Scott Somershoe
Jane Carlson	Micah Landry	Stephen Sorensen
Steve Carpenedo	Dan Lane	Bob St. Amant
Mollie Cashman	Janina Lamb	Linsday St. Amant
Santiago Claramunt	Holly LeGrand	Paul Stouffer
Meghann Clark	Ellen Leichty	Phil Stouffer
Alain Clavette	Haw-Chuan Lim	Danielle Temple
Heather Conkerton	Randy Little	Sarah Temple
Brian Crother	Eve Loftman	Angelle Vallaire
Sam Cusack	Lisa McArthur	Scarlett Vallaire
George Domas	Tiffany McFalls	Rachel Villani
Lynn Duda	Katie McKearan	Sandy Walczyk
Ryan Dziedzic	Matt McKearan	Ryan Walsh
Devin Eby-Bosler	Bob Marshall	Peter Wegman
Justin Eby-Bosler	Melissa Miller	Meg Williamson
Katie Faust	Pam Morgan	Walker Wilson
Joelle Finley	Luciano Naka	Stefan Woltman
Dave Fox	Brian O'Shea	Jack Young
Kelli Gilbert	Tim Paine	Jason Zoller

**APPENDIX 2. EXPERIMENTAL DESIGN AND STATISTICAL ANALYSIS USING A
QUASI LATIN RECTANGE**

Source	F-Test
Fire	Fire * Plot * Year
Plot	Fire * Plot * Year
Year	Fire * Plot * Year
Fire * Plot * Year	
Sampling Period	Sam. Per. * Fire * Plot * Year
Sam. Per. * Fire * Plot * Year	
Bird Age	Birds (Sam. Per. * Fire * Plot * Year)
Birds (Sam. Per. * Fire * Plot * Year)	

**APPENDIX 3. RAW DATA FOR CAPTURED LE CONTE'S SPARROWS (LCSP) AND
GRASSHOPPER SPARROWS (GRSP)**

Species	Plot	Date	Time	Wing	Weight	Fat Score	Age
LCSP	near LR01	23 Oct03	1441	52.0	12.00	2	U
LCSP	LR05	23 Nov 03	1219	46.0	11.50	2	AHY
LCSP	LR04	5 Dec 03	947	52.0	12.50	2	U
LCSP	LR05	8 Dec 03	1020	51.0	12.50	3	U
LCSP*	LR05	9 Feb 04	1022	51.0	14.00	nr	U
LCSP	LR05	8 Dec 03	1245	47.5	11.25	2	HY
LCSP	LR01	20 Dec 03	1119	48.0	14.00	2	U
LCSP	LR05	24 Jan 04	936	49.0	12.50	4	SY
LCSP	LR05	9 Feb 04	950	51.0	14.00	3	U
LCSP	LR05	9 Feb 04	1158	48.0	14.00	6	U
LCSP	LR03	6 Nov 04	1427	52.0	11.75	2	U
LCSP	LR04	6 Nov 04	1430	51.0	11.75	2	HY
GRSP	LR01	1 Nov 03	1045	60.0	18.00	4	U
GRSP	CWP	6 Nov 03	1620	60.0	22.00	6	U
GRSP	CWP	6 Dec 03	921	60.0	16.50	1	HY
GRSP	CWP	15 Dec 03	1100	62.0	18.00	1	HY
GRSP	CWP	15 Dec 03	1121	61.0	19.50	2	U
GRSP	LR05	13 Nov 04	1132	62.0	19.00	5	HY

* same individual as previous entry

nr: not recorded

APPENDIX 4. RAW DATA FOR SEED-BEARING STALK (TOP ROW FOR EACH SPECIES) AND SEED (BOTTOM ROW FOR EACH SPECIES) DENSITIES FOR EACH PLANT SPECIES AT EACH SITE IN BURN-YEAR AND NON-BURN-YEAR SAVANNAS WHEN COLLECTED IN LATE NOVEMBER

	Abita Creek Flatwoods		Lake Ramsay WMA		Sandy Hollow WMA		Camp Whispering Pines	
	burn-year	non-burn-year	burn-year	non-burn-year	burn-year	non-burn-year	burn-year	non-burn-year
Number of plots	1	3	6	4	2	2	1	1
APIACEAE								
<i>Oxypolis filiformis</i> (Walt.)	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Britt.	0.2	4	0	0	0	0	0	0
ASTERACEAE								
<i>Balduina uniflora</i> Nutt.	0.0	0.7	0.3	0.1	0.0	0.0	0.0	0.0
	0.0	12.3	2.8	0.7	0.0	0.0	0.0	0.0
<i>Bidens mitis</i> (Michx.) Sherff	4.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	15.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Bigelovia</i> spp. DC.	0.0	0.2	0.9	2.0	0.0	0.0	0.0	0.0
	0.0	19.2	26.7	199.0	0.0	0.0	0.0	0.0
<i>Boltonia diffusa</i> Ell.	0.0	0.0	0.1	0.1	1.9	1.0	0.0	0.5
	0.0	0.0	0.0	1.1	3.9	23.4	0.0	1.5
<i>Carphephorus pseudoliatris</i>	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Cass.	0.0	0.0	17.5	0.0	0.0	0.0	0.0	0.0
<i>Coreopsis linifolia</i> Nutt.	9.3	0.7	0.9	0.4	0.0	0.0	0.0	0.0
	25.9	20.5	9.4	20.1	0.0	0.0	0.0	0.0
<i>Elephantopus tomentosus</i> L.	0.0	0.0	0.0	0.0	0.3	0.1	0.2	0.0
	0.0	0.0	0.0	0.0	0.7	3.6	0.2	0.0
<i>Eryngium integrifolium</i>	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Walt.	0.0	0.0	1.8	4.2	0.0	0.0	0.0	0.0
<i>Eupatorium</i> spp. L.	0.0	0.1	0.5	0.1	0.0	0.0	0.2	0.3
	0.9	1.1	82.0	0.3	3.2	2.4	474.2	20.1

APPENDIX 4. continued.

<i>E. leucolepis</i> (DC.) Torr. & Gray	0.5	0.0	5.4	0.1	0.0	0.0	0.2	0.0
	5.5	0.0	431.5	0.2	0.0	0.0	37.7	0.0
<i>E. rotundifolium</i> L.	0.0	0.0	0.0	0.0	0.3	0.8	2.3	0.2
	0.0	0.0	0.0	0.0	8.4	53.6	1505.2	0.8
<i>E. serotinum</i> Michx.	0.0	0.0	0.0	0.1	0.3	0.0	0.0	0.0
	0.0	0.0	0.0	0.6	5.6	0.0	0.0	0.0
<i>Eurybia hemispherica</i> (Alexander) Nesom	0.0	0.0	0.0	0.0	0.0	0.1	0.3	0.5
	0.0	0.0	0.0	0.0	0.0	5.5	14.3	5.2
<i>Euthamia caroliniana</i> (L.) Greene ex Porter & Britt.	0.0	0.2	0.1	0.1	0.0	0.0	0.0	0.0
	0.0	14.3	1.0	0.0	0.0	0.0	0.0	0.0
<i>Helianthus</i> spp. L.	0.0	0.0	0.0	0.0	0.4	0.1	0.0	0.0
	0.0	0.0	0.0	0.0	0.1	4.4	1.6	0.0
<i>H. angustifolius</i> L.	1.7	0.3	0.8	0.4	1.6	1.0	0.2	0.3
	1.7	2.9	17.9	4.0	54.9	7.8	4.3	0.2
<i>Ionactis linariifolius</i> (L.) Greene	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.3
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.3
<i>Liatris</i> spp. Gaertn. ex Schreb.	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0
	0.0	0.1	0.4	0.0	107.9	0.0	0.0	0.0
<i>Pityopsis graminifolia</i> (Michx.) Nutt.	0.0	0.0	1.6	0.0	0.0	0.0	7.3	0.3
	0.0	0.0	54.5	0.0	0.3	0.0	788.4	0.0
<i>Solidago</i> spp. L.	0.0	0.0	0.1	0.1	0.7	0.6	0.2	0.2
	0.0	2.0	1.7	0.1	0.8	117.3	0.0	0.0
<i>Symplotrichium adnatus</i> (Nutt.) Nesom	0.0	0.0	2.0	0.0	4.3	2.8	4.5	2.0
	0.0	0.0	53.4	0.0	65.0	70.6	315.2	30.8
<i>S. concolor</i> (L.) Nesom	0.0	0.0	0.2	0.0	0.3	0.0	0.5	0.3
	0.0	0.0	32.1	0.0	4.2	0.0	64.0	28.0
<i>S. dumosum</i> (L.) Nesom	0.8	2.7	1.4	1.0	2.9	4.5	22.5	25.7
	42.7	126.8	42.4	26.7	21.1	22.6	318.4	212.8

APPENDIX 4. continued.

<i>S. patens</i> (Ait.) Nesom	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	4.6	0.0	0.0	0.0
Unknown Asteraceae 1	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Unknown Asteraceae 2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
BUDDLEJACEAE								
<i>Polypremum procumbens</i> L.	0.0	0.0	0.0	0.0	0.3	0.2	0.0	0.0
	0.0*	0.0*	0.0*	0.0*	12.1*	0.0*	0.0*	0.0*
CAMPANULACEAE								
<i>Lobelia</i> spp. L.	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
	0.0*	0.0*	0.0*	0.0*	1.2*	0.0*	0.0*	0.0*
<i>L. brevifolia</i> Nutt. ex A. DC.	1.5	0.0	0.2	0.3	0.0	0.1	0.0	0.0
	10.2*	0.0*	15.9*	33.8*	0.0*	15.3*	0.0*	0.0*
CISTACEAE								
<i>Lechea</i> spp. L.	0.0	0.0	0.4	0.1	0.8	1.3	0.0	0.8
	0.0*	0.0*	12.3*	0.2*	5.5*	87.0*	0.5*	56.0*
CLUSIACEAE								
<i>Hypericum</i> spp. L.	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0
	0.0*	0.0*	0.5*	45.1*	0.0*	0.0*	0.0*	0.0*
<i>H. setosum</i> L.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7
	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	70.8*
CYPERACEAE								
<i>Carex turgescens</i> Torr.	0.2	0.3	0.0	0.0	0.0	0.0	0.0	0.0
	1.8	13.3	0.1	0.0	0.0	0.0	0.0	0.0
<i>Eleocharis</i> spp. R. Br.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0
<i>E. dulcus</i> (Burm.) Trin. ex Hensch.	0.0	0.5	0.5	0.0	0.0	0.0	0.0	0.0
	0.0	1.8	0.8	0.0	0.0	0.0	0.0	0.0

APPENDIX 4. continued.

<i>Fimbristylis</i> spp. Vahl	0.0	0.0	0.0	0.0	0.1	0.0	0.3	0.0
	0.0	0.0	0.0	0.0	0.0	8.2	0.1	0.0
<i>Fuirena breviseta</i> (Coville)	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Coville	6.8	19.3	1.0	0.0	0.0	0.0	0.0	0.0
<i>Rhynchospora</i> spp. Vahl	0.0	1.5	3.4	0.0	0.2	0.0	0.0	0.0
	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.2
<i>R. cephalantha</i> Gray	0.5	3.7	6.8	0.0	0.0	0.0	0.0	0.0
	20.8	430.1	2.8	0.3	0.0	0.0	0.0	0.0
<i>R. chapmanii</i> M.A. Curtis	0.0	36.3	38.4	215.8	0.0	0.0	0.0	0.0
	49.1	701.2	169.4	997.4	0.0	0.0	0.0	0.0
<i>R. ciliaris</i> (Michx.) C. Mohr	0.0	0.4	2.5	0.8	0.0	0.0	0.0	0.0
	0.0	1.7	9.9	1.5	0.0	0.0	0.0	0.0
<i>R. compressa</i> Carey ex Chapman	0.0	2.7	0.5	1.0	0.0	0.0	0.0	0.0
	6.0	108.5	3.9	6.7	0.0	0.0	0.0	0.0
<i>R. corniculata</i> (Lam.) Gray	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2.2	0.0	0.0	0.0	0.0	0.0	0.3	0.0
<i>R. elliotii</i> A. Dietr.	0.2	1.4	0.5	0.2	0.0	0.0	0.0	0.0
	21.2	775.7	100.7	17.4	0.0	0.0	0.0	0.0
<i>R. filifolia</i> Gray	0.0	0.1	0.7	0.2	0.0	0.0	0.0	0.0
	1.9	39.6	3.1	0.1	0.0	0.0	0.0	0.0
<i>R. globularis</i> (Chapman) Small	0.0	0.0	0.0	0.1	1.1	1.4	0.7	3.0
	0.0	0.0	0.1	0.9	6.3	16.6	79.0	57.0
<i>R. gracilentata</i> Gray	17.3	301.1	10.9	36.7	0.0	0.0	0.0	0.0
	66.2	1022.8	80.5	71.4	0.0	0.0	0.0	0.0
<i>R. inexpansa</i> (Michx.) Vahl	0.0	0.2	0.0	0.1	0.0	0.0	0.0	0.0
	4.0	71.0	4.5	3.2	0.0	0.0	0.0	0.0
<i>R. latifolia</i> (Baldw. ex Ell.) Thomas	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0
	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0

APPENDIX 4. continued.

<i>R. microcephala</i> (Britt.)	1.3	6.1	0.4	14.8	0.1	0.0	0.0	0.0
Britt.	0.0	42.1	0.0	0.0	0.0	0.0	0.0	0.0
<i>R. oligantha</i> Gray	10.3	1.8	0.9	0.2	0.0	0.0	0.0	0.0
	20.4	6.1	1.5	0.6	0.0	0.0	0.1	0.0
<i>R. plumose</i> Ell.	0.0	3.6	47.7	115.5	0.0	0.0	0.0	0.0
	13.9	121.8	294.0	640.9	0.0	0.0	0.0	0.3
<i>R. pusilla</i> Chapman ex M.A.	0.0	1.1	8.4	6.0	6.4	1.0	4.7	13.8
Curtis	0.0*	8.6*	25.8*	9.6*	5.6*	4.4*	59.5*	95.0*
<i>R. rariflora</i> (Michx.) Ell.	12.0	63.1	5.2	28.1	0.0	0.0	0.0	0.2
	16.2	303.4	21.4	55.3	0.0	0.0	0.0	0.3
<i>Scleria hirtella</i> Sw.	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
<i>S. muehlenbergii</i> Steud.	26.7	2.4	34.5	1.2	0.0	0.0	0.0	0.0
	55.8	12.8	31.4	4.1	0.0	0.0	0.0	0.0
<i>S. pauciflora</i> Muhl. ex	1.0	3.5	20.2	51.0	19.5	5.7	9.3	49.2
Willd.	2.3	6.0	17.6	58.5	11.4	8.8	16.9	47.7
ERIOCAULACEAE								
<i>Eriocaulon decangulare</i> L.	6.5	7.7	0.8	3.0	0.0	0.0	0.0	0.0
	197.7*	138.4*	20.6*	61.7*	0.0	0.0	0.0	0.0
EUPHORBIACEAE								
<i>Acalypha gracilens</i> Gray	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
	0.0	0.0	0.0	0.0	0.0*	0.0*	0.0*	0.0*
FABACEAE								
<i>Crotalaria</i> spp. L.	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.2
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Stylosanthes biflora</i> (L.)	0.0	0.0	0.0	0.0	0.1	0.0	0.0	1.4
B.S.P.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Tephrosia</i> spp. Pers.	0.0	0.0	0.0	0.1	0.9	0.1	0.0	1.8
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

APPENDIX 4. continued.

GENTIANACEAE								
<i>Sabatia</i> spp. Adans.	0.0	0.1	0.5	0.3	0.0	0.0	0.0	0.0
	0.0*	68.3*	41.0*	4.8*	0.0	0.0	0.0	0.0
JUNCACEAE								
<i>Juncus trigonocarpus</i> Steud.	0.2	0.1	0.1	0.3	0.0	0.0	0.0	0.0
	1.3	0.0	0.0	39.7	0.0	0.0	0.0	0.0
LILIACEAE								
<i>Aletris</i> spp. L.	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0
	0.0*	0.3*	0.0*	10.1*	10.7*	0.0*	0.0*	0.0*
LINACEAE								
<i>Linum medium</i> (Planch.) Britt.	0.5	0.1	0.7	0.9	0.0	0.1	0.2	0.3
	3.8	1.4	16.0	16.3	0.0	7.8	3.0	21.8
LOGANIACEAE								
<i>Mitreola</i> L.	0.2	0.0	0.0	0.1	0.0	0.2	0.7	5.0
	320.5*	0.0*	0.0*	15.2*	0.0*	122.1*	47.2*	314.1*
MELASTOMATACEAE								
<i>Rhexia alifanus</i> Walt.	0.0	0.3	1.9	0.8	0.0	0.1	0.0	0.0
	1.3	6.1	160.8	14.3	0.0	9.0	0.2	0.0
<i>R. lutea</i> Walt.	0.0	0.6	0.1	0.0	0.0	0.0	0.0	0.0
	0.0*	130.7*	9.3*	0.0*	0.0	0.0	0.0	0.0
<i>R. mariana</i> L.	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	37.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*
ONAGRACEAE								
<i>Ludwigia</i> spp. L.	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*
POACEAE								
<i>Anthaenantia</i> spp. Beauv.	1.0	0.1	0.9	0.1	0.2	0.0	0.0	0.0
	12.7	0.2	6.7	1.4	0.2	0.0	0.0	0.0

APPENDIX 4. continued.

<i>Andropogon</i> spp. L.	20.0	2.2	20.8	1.9	3.9	2.5	38.2	0.5
	1009.9	311.7	957.0	190.8	120.1	113.0	2030.4	15.0
<i>Aristida</i> spp. L.	0.8	6.8	7.7	7.0	5.6	1.2	15.0	21.8
	2.2	37.3	13.9	10.4	63.7	12.2	318.4	212.8
<i>A. palustris</i> (Chapman)	0.2	3.6	0.1	0.0	0.0	0.0	0.0	0.0
Vasey	0.3	30.0	0.2	0.0	0.0	0.0	0.0	0.0
<i>Ctenium aromaticum</i>	0.0	0.0	8.0	0.0	0.6	0.0	2.2	0.0
(Walt.) Wood	0.0	0.0	200.1	0.3	5.3	2.4	271.9	0.0
<i>Dichanthelium</i> spp. (A.S.	9.3	66.8	45.9	139.9	114.5	137.8	27.3	247.5
Hitchc. & Chase) Gould	17.8	296.5	89.8	130.3	240.3	1101.3	617.4	966.5
<i>D. scabriusculum</i> (Ell.)	17.7	8.0	0.9	0.8	0.0	0.0	0.0	0.0
Gould & C.A. Clark	106.4	132.2	7.3	17.6	0.0	0.0	0.0	0.0
<i>Digitaria</i> spp. Haller	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.2
<i>Eragrostis</i> spp. von Wolf	0.5	2.2	0.8	0.7	0.5	0.0	2.2	0.7
	3.3*	52.1*	4.4*	19.4*	5.8*	0.0*	55.5*	3.7*
<i>Gymnopogon brevifolius</i>	0.0	0.1	0.1	0.1	0.7	1.8	1.7	3.7
Trin.	0.0	1.3	0.0	0.1	22.2	18.5	53.6	56.3
<i>Muhlenbergia expansa</i>	3.3	0.0	15.2	0.3	0.0	0.0	2.7	0.0
(Poir.) Trin.	8.2	0	225.4	5.5	0	0	36.8	0
<i>Panicum</i> spp. L.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	32.6	34.2	17.9	1.0	18	3.1	2.1	16.8
<i>P. anceps</i> Michx.	3.3	5.2	1.5	0.7	4.6	3.0	21.8	3.0
	119.1	327.3	62.1	58.0	98.9	182.4	1699.5	108.2
<i>P. tenerum</i> Bey. ex Trin.	0.0	2.4	0.4	0.0	0.0	0.0	0.0	0.0
	0.0	22.7	2.7	0.0	0.0	0.0	0.0	0.0
<i>P. verrucosum</i> Muhl.	18.7	1.9	0.0	0.2	0.0	0.0	0.0	1.2
	11.0	18.4	0.9	0.1	0.0	0.0	250.9	55.5

APPENDIX 4. continued.

<i>P. virgatum</i> L.	1.0	1.7	0.2	0.1	0.0	0.0	0.0	1.0
	3.3	132.5	0.1	0.0	0.0	0.0	0.0	5.0
<i>Paspalum</i> spp. L.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0
<i>P. floridanum</i> Michx.	0.2	0.0	0.0	0.2	0.4	0.1	0.7	1.7
	0.0	1.9	0.1	1.3	1.0	4.5	4.6	10.9
<i>P. leave</i> Michx.	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	2.2	0.0	0.6
<i>P. praecox</i> Walt.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.1	0.2	0.0	0.0	0.0	0.0
<i>P. setaceum</i> Michx.	0.0	0.0	0.2	0.1	0.0	0.0	0.2	0.7
	0.0	0.0	0.0	1.0	0.0	0.9	24.6	17.8
<i>Saccharum giganteum</i> (Walt.) Pers.	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5.6	9.3	0.0	0.0	0.0	0.0	0.0	0.0
<i>Schizachyrium scoparium</i> (Michx.) Nash.	3.0	0.5	9.5	1.7	28.8	15.0	31.2	2.0
	102.3	87.7	570.6	22.1	410.0	927.8	932.9	386.3
<i>Sporobolus junceus</i> (Beauv.) Kunth	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	21.6	0.0	0.0
<i>Tridens ambiguus</i> (Ell.) J.A. Schultes	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.0
	0.0	0.0	6.3	0.2	0.0	0.0	44.9	0.0
SCROPHULARACEAE								
<i>Agalinis</i> spp. Raf	0.0	0.2	2.3	0.8	0.0	0.7	0.0	0.2
	0.0*	48.3*	116.4*	42.4*	0.0*	87.3*	0.0*	0.0*
XYRIDACEAE								
<i>Xyris</i> spp. L.	1.5	1.0	0.5	0.3	0.0	0.0	0.0	0.0
	2343.2*	275.4*	91.9*	123.3*	0.0	0.0	0.0	0.0
UNKNOWN								
Unknown # 7	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*

APPENDIX 4. continued.

Unknown # 20	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*
Unknown # 25	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*
Unknown # 30B	0.0	0.1	0.1	0.1	0.0	0.0	0.0	0.0
	0.0*	13.3*	0.0*	3.2*	0.0*	0.0*	0.0*	0.0*
Unknown # 31	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*	0.0*
Unknown seed AA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Unknown seed AB	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9
Unknown seed C	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0
Unknown seed D	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
Unknown seed E	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
Unknown seed F	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Unknown seed G	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	3.2	0.0	0.0	0.0	2.4	11.8	0.0
Unknown seed M	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0
Unknown seed Q	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0
Unknown seed R	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0

APPENDIX 4. continued.

Unknown seed S	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Unknown seed T	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0
Unknown seed X	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0.0	0.0	0.0	0.0	0.0	4.1	0.0	2.6

* These seeds were not included in analyses because they were assumed to be not consumed by Henslow's Sparrows and were not adequately sampled using the vacuum. Numbers represent the seeds on flowering stalk densities and do not include vacuumed samples.

VITA

Erik Johnson was born in Boston, Massachusetts to parents Ivan and Delia. His interest in wildlife and especially birds developed at an early age, inspired by the magic of the White Mountains in New Hampshire and his parents' encouragement. In 1997 he graduated from Upper St. Clair High School in Pennsylvania and went to Dickinson College to study Biology and Environmental Studies. There, he participated in a variety of research projects, involving fish, diatoms, white-tailed deer, American chestnuts, and tropical finches. Upon graduation in May 2001, he moved back to Boston where he developed his skills as a wildlife photographer and worked as a wildlife technician monitoring breeding Least Terns and Piping Plovers. In the fall of 2003 he began his work towards his Master's at Southeastern Louisiana University where he met his fiancée, Ceci. Erik is a candidate for a Master of Science degree in wildlife.