

**POST-FIRE SUCCESSIONAL EFFECTS ON BREEDING GRASSLAND BIRDS  
IN MESQUITE SAVANNA HABITATS OF THE TEXAS ROLLING PLAINS**

A Thesis

by

STEPHANIE L. LEE

Submitted to the Office of Graduate Studies of  
Texas A&M University  
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

December 2006

Major Subject: Wildlife and Fisheries Sciences

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Approved by:

Co-Chairs of Committee:	R. Dean Ransom, Jr. R. Douglas Slack
Committee Member:	Fred E. Smeins
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**ABSTRACT**

Post-Fire Successional Effects on Breeding Grassland Birds in Mesquite Savanna

Habitats of the Texas Rolling Plains. (December 2006)

Stephanie L. Lee, B.S., Auburn University

Co-Chairs of Advisory Committee: Dr. R. Dean Ransom, Jr.  
Dr. R. Douglas Slack

North American grasslands and grassland birds have declined drastically due to habitat degradation by fire suppression (i.e., woody encroachment), fragmentation, and conversion to croplands. A better understanding is needed of the relationships among disturbance regimes (e.g., fire), resultant vegetation changes, and grassland bird communities to effectively manage remaining grasslands and grassland birds. I assessed the relationship between post-fire succession, and mean relative abundance and nesting ecology of breeding grassland birds (i.e., nest-site selection and nest success) in mesquite-dominated rangeland of the Texas Rolling Plains, where prescribed fire is used as a tool to manage shrub encroachment. Brush cover, grass cover, and visual obstruction generally increased with post-fire succession, and bare ground decreased with post-fire succession. Species richness, grasshopper sparrows (*Ammodramus savannarum*), Cassin's sparrows (*Aimophila cassinii*), and dickcissels (*Spiza americana*) responded positively to post-fire succession, and lark sparrows (*Chondestes grammacus*) responded negatively to post-fire succession.; abundance of these avian groups was low on the control sites. During 2004–2005, 90 grassland bird nests were monitored. I found conflicting results for vegetation parameters important to nest site selection and

probability of nest success. For all species except lark sparrows, nest-site location was positively associated with visual obstruction and with grass or forb cover. However, the probability of nest success increased with lower visual obstruction, bare ground cover, or grass cover. Grassland bird abundance, nest-site location, and nest success had differing associations with vegetation variables. These results suggest that to effectively manage remaining grasslands for sustainable breeding grassland bird populations, managers should engage in practices that keep habitat in multiple vegetative successional stages.

**DEDICATION**

For my husband Daniel

*For always believing in me*

## ACKNOWLEDGEMENTS

This project would not have been possible without the help of several people. I would like to start by thanking my co-advisors Dr. Dean Ransom and Dr. Doug Slack and my committee member Dr. Fred Smeins for their help with this project and review of this document. I would also like to thank Brian Pierce and Bret Collier for their help with program DISTANCE, statistics, and always having a sense of humor for my never ending questions. Thank you to the Joe Skeen Institute for Rangeland Restoration for their financial support of this project. In addition, I would like to thank the W. T. Waggoner Ranch for allowing me to use their ranch to conduct this research. The Texas Agricultural Experiment Station-Vernon provided housing and transportation during field research. The Department of Wildlife & Fisheries Sciences provided office space and access to computer facilities.

I could not have completed the project without the help of my summer workers and Dean's research assistant. I would like to thank Gerral Schulz, Sandy Vistine, Brady Serber, Claire Curry, and Daniel Lee for all the long hours they endured in the Texas heat surveying birds, searching for nests, and completing vegetation surveys. I would especially like to thank Brady Serber and Daniel Lee for always having my back even if it means pushing the truck 2 miles to the paved road. Special thanks to e-claire for serving as the project photographer.

I would not be where I am today without the love, support, and encouragement from my family; thanks to my parents, Melissa, Stephen, and my grandparents. A special

thanks to Daniel for his love, support, and friendship over the years. Of course, I cannot leave out Baxter, Rizzo, Jack, and the turtle for making life more interesting with their misadventures.

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## CHAPTER I

### INTRODUCTION

Before European settlement of the North American grasslands, the Great Plains comprised the largest vegetative province in North America and was estimated to cover 162 million ha of land (Knopf 1994). The North American grasslands extended from central Mexico to the Canadian provinces and were characterized by diverse assemblages of vegetation types (Risser et al. 1981, Anderson 1990). The Great Plains consists of 3 types of native prairie: tallgrass, mixed grass, and short grass plains. The mixed grass prairie has declined by 72–99% and the tallgrass prairie has declined 83–99% from their previous distributions due to the cumulative effects of overgrazing, suppression of fire, and conversion to cropland or urban development (Knopf 1996).

The grassland system evolved with and was maintained by climate and periodic disturbances, particularly fire and grazing (Wright and Bailey 1982, Higgins 1986). Fire was a continuous disturbance regime in maintaining these grasslands, but not consistent in frequency or intensity (Curtis 1959, Daubenmire 1968, Vogle 1974). Fire frequency primarily influenced the composition and structure of grassland vegetation communities (Riggs et al. 1996). Wright and Bailey (1982) estimated pristine grasslands of level to gently rolling topography to have had a 5–10 year burn interval. These periodic fires shaped grassland ecology by increasing grass productivity and fire tolerant species while decreasing forbs and woody plant species that are more susceptible to fire (Collins and Gibson 1990). The recovery time from natural prairie fire is estimated to be 3–4 years

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for tallgrass, 6 years for northern mixed prairie, and 5–10 years in a short grass prairie (Bragg 1995). The recent fire history of the North American grassland ecosystem has been one of human suppression.

Continual suppression of fire and intensive grazing by domestic livestock has altered the vegetative composition and structure of many grassland systems, from native grasslands or savannahs to shrub-dominated lands (Archer et al. 1988, Archer 1989, Scifres and Hamilton 1993). One of the most dramatic examples of this vegetation shift has been the increase of mesquite (*Prosopis* spp.) on grasslands of the southwestern United States (Archer 1989). Historical records suggest honey mesquite (*Prosopis glandulosa* var. *glandulosa* Torr.) has always been a dominant species in the Texas Rolling Plains (Malin 1953). Johnston (1963) provided further evidence that the geographic distribution of honey mesquite has changed little from its historical range, but has increased in density and height. Over the past 100 years mesquite has grown into dense thickets due to livestock grazing and fire suppression (Wright et al. 1976). Early explorers traveling in west Texas and eastern New Mexico reported an abundance of honey mesquite on uplands and along streams (Marcy 1849, Micheler 1850). In the Rolling Plains of Texas, Marcy (1849) described the area as "mesquite timber" in a "peach-like orchard."

The changes in fire frequency combined with grazing pressure and climate characteristics determine the structure and composition of grassland vegetation, which influences the grassland bird abundance, diversity, and avian communities present in a

grassland system (Axlerod 1985, Zimmerman 1997). A study on the Konza prairie reported a shift from grassland-dependent avian communities to shrub-dependent avian communities as the vegetation structure of grasslands changed due to decreased fire frequency (Zimmerman 1997). Another study in Arizona showed mesquite-dominated grasslands supported shrub-dependent avian communities, which historically had supported grassland-dependent avian communities (Lloyd et al. 1998). The role of vegetation composition and structure in relation to grassland bird habitat use is well-documented (Cody 1968, Rotenberry and Wiens 1980).

Fire is a common tool used to manage woody encroachment in many rangeland habitats due to the low cost compared to mechanical and chemical management (Scifres and Hamilton 1993, Teague et al. 1997). However, the effectiveness of fire as a restoration tool for grasslands and their passerine breeding bird communities from shrub dominated lands has received little attention (Skinner 1975, Risser et al. 1981, Kirkpatrick et al. 2002). The majority of studies using prescribed fire have typically used only a single burn and control site (Renwald 1977, Huber and Steuter 1984, Pylypec 1991), or they have examined avian responses only 1–3 years post-burn (Forde et al. 1984, Herkert 1994). Studies addressing long term effects are few. More recent short-term studies investigating fire as a restoration tool have reported contrasting results, thus showing no clear patterns of prescribed-fire effects on grassland birds (Reynolds and Krausman 1998, Kirkpatrick et al. 2002).

Two studies that investigated long-term effects (>10 years) of fire on grassland birds noted short-term decreases in grassland bird abundance on 1-year post-burn sites, but no

long-term effects on grassland-bird abundance in the North Dakota mixed prairies (Johnson 1997, Madden et al. 1999).

This thesis explores the role of fire as a grassland restoration tool as it defines the vegetation patterns and bird communities in mesquite dominated rangelands. The objectives of this thesis were to quantify relationships among breeding grassland bird mean relative abundance, vegetation features, and nest success on the mesquite dominated shrubland in the Texas Rolling Plains.

## **STUDY OBJECTIVES**

This project examined the effects of prescribed fire on breeding grassland bird mean relative abundance and nest success along a <1–9 year-old post-burn continuum in a mesquite shrubland of the Texas Rolling Plains. The specific objectives were: 1) to examine trends in the abundance of grassland breeding bird communities and single species, 2) to determine nest success, and 3) to identify important vegetation characteristics associated with grassland breeding bird abundance, nest-site selection and nesting success on the Waggoner Experimental Ranch (WER).

**CHAPTER II**  
**PRESCRIBED FIRE EFFECTS ON BREEDING GRASSLAND BIRD**  
**COMMUNITIES IN MESQUITE SAVANNA HABITATS**

**INTRODUCTION**

For the past 25 years, North American grassland birds have declined more than any other avian guild (Knopf 1994, Peterjohn and Sauer 1999). Breeding Bird Survey (BBS) data have shown 14 of 19 widespread North American grassland bird species and 6 of 9 endemic species are in decline (Knopf 1996). Many grassland-dependent species such as the dickcissel (*Spiza americana*), the grasshopper sparrow (*Ammodramus savannarum*), and the Cassin's sparrow (*Aimophila cassinii*) are Partners in Flight species of concern (Carter et al. 1996) and have shown declining breeding population trends from 1966–1996 (Peterjohn and Sauer 1999). In Texas, breeding densities of lark sparrows (*Chondestes grammacus*) are among the highest in the United States but are declining 4.2% per year (Sauer et al. 1997). Fragmentation, loss, and degradation (e.g., woody encroachment) of habitats are cited as the main causes for observed declines (Johnson and Igl 2001, Vickery and Herkert 2001). The remaining grasslands are vitally important to grassland-dependent bird species persistence and diversity.

Grassland-bird communities are characterized by low diversity and density (Cody 1966) and stable annual species richness (Wiens 1973). A study on the Konza tall-grass prairie in Kansas reported 9 of 12 grassland-dependent species returned annually (Zimmerman 1993). Furthermore, implementing fire annually and on 4-year intervals

had no significant effect on species composition of a grassland-dependent community in the Konza prairie (Zimmerman 1997). This supports the overall lack of response to prescribed fire by grassland-dependent species (Fitzgerald and Tanner 1992, Johnson 1997, Rohrbaugh et al. 1999). Fire does not specifically define the grassland bird communities present so much as it suppresses woody-plant succession and shrub-dependent avian communities (Zimmerman 1992).

Fire is a common tool used to manage woody encroachment in many rangeland habitats due to the low cost compared to mechanical and chemical management (Scifres and Hamilton 1993, Teague et al. 1997). However, the effectiveness of fire as a restoration tool for grasslands and their passerine breeding bird communities from shrub dominated lands has received little attention (Skinner 1975, Risser et al. 1981, Kirkpatrick et al. 2002). Scifres and Hamilton (1993) point out that although wildfires and those set by Native Americans did maintain grasslands from woody-plant encroachment, they did not convert shrublands of present-day proportions to grasslands. The majority of studies using prescribed fire have typically used only a single burn and control site (Renwald 1977, Huber and Steuter 1984, Pylypec 1991), or they have examined avian responses only 1–3 years post-burn (Forde et al. 1984, Herkert 1994). Few studies have investigated long-term fire effects. More recent short-term studies investigating fire as a restoration tool have reported contrasting results, thus showing no clear patterns of prescribed-fire effects on grassland birds (Reynolds and Krausman 1998, Kirkpatrick et al. 2002). Two studies that have investigated long-term effects (>10 years) of fire on grassland birds in the North Dakota mixed prairies reported short-

term decreases in grassland bird abundance on 1-year post-burn sites, but no long-term effects on grassland bird abundance (Johnson 1997, Madden et al. 1999).

The use of prescribed fire as a restoration tool for grassland-bird communities should influence grassland-bird community demographics throughout post-burn succession of vegetation. In shrub-dominated grasslands, fire can alter shrub height and canopy cover, leading to the reestablishment of a grassland matrix. However, many woody plants (e.g., mesquite) are only top-killed by fire and have the ability to re-sprout with a multi-stem growth form, making density reduction by fire difficult to achieve (Scifres and Hamilton 1993). Thus, after an initial fire, vegetation may still approximate a shrubland rather than true grassland, so it remains to be seen if grassland-dependent birds will inhabit burned sites. Kirkpatrick et al. (2002) reported a decline in grassland bird abundance immediately following a burn on mesquite-invaded treatment plots compared to mesquite-invaded control plots in southern Arizona grasslands. However, frequent prescribed fires may enhance breeding bird populations by increasing plant biomass for nest concealment, decreasing predation (Johnson and Temple 1990), and/or increasing insect densities for food available to nestlings (Evans 1988).

A threshold may exist where grassland bird communities and reproductive success begin to decline along a successional gradient of woody vegetation growth. Guthery (1999) noted a threshold likely existed for northern bobwhite (*Colinus virginianus*) where too little herbaceous cover and too much woody cover led to a loss of useable space and ultimately a decline in bobwhite abundance. Similarly, Zimmerman (1992) noted the core grassland bird species were not affected by climatic extremes (i.e.,

extreme dry or wet years) as long as the structural complexity of the habitat did not decline below a threshold. Finding a threshold would help determine a fire frequency interval necessary for grassland-bird persistence, and may make it possible to simultaneously maintain cattle production and grassland- bird habitat by managing mesquite. Addressing this relationship experimentally is necessary to further an understanding of grassland bird ecology and to improve the conservation and management of grassland bird populations and their habitat.

To effectively manage remaining grasslands, biologists need to better understand the relationships among disturbance regimes (e.g., fire), resultant vegetation changes, and grassland-bird communities. The overall objective of this study was to examine the effects of prescribed fire on grassland-breeding bird mean relative abundance along a <1–9 year-old post-burn continuum in mesquite shrublands of the Texas Rolling Plains. The specific objectives were: 1) to examine trends in breeding grassland bird communities and single species abundances and 2) to identify important vegetation characteristics associated with grassland bird abundance on the Waggoner Experimental Ranch.

## STUDY AREA

The study area was located on the 12,141 ha Kite Camp of the Waggoner Experimental Ranch (WER) in Wilbarger County, Texas (Figure 2.1). The region is characterized by a continental and semiarid climate with an average growing season of 220 days. Annual precipitation averages 64.8 cm with seasonal peaks in May and September. Ambient temperatures range from -2.3 C° in January to 36.4 C° in July. Topography is moderately rolling with elevations ranging from 335–396 m. The landscape is characteristic of the Red Rolling Plains ecological region and was historically a grassland or grassland savanna maintained by periodic wildfire (Gould 1975). Currently, the WER is a shrubland savanna dominated by honey mesquite and lotebush (*Ziziphus obtusifolia* (T. & G.) Gray) in various stages of succession resulting from application of herbicides and prescribed fire (Gould 1975). The herbaceous species are a mixture of cool and warm season grasses including Texas wintergrass (*Stipa leucotricha* Trin. & Rupr.), Texas bluegrass (*Poa arachnifera* Torr.), Japanese brome (*Bromus japonicus* Thunb.), buffalograss (*Buchloe dactyloides* (Nutt.) Engelm.), silver bluestem (*Bothriochloa saccharoides* (Sw.) Rydb.), meadow dropseed (*Sporobolus asper* var. *hookeri* Trin.), and sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.) (Gould 1975). The Kite Camp is a cow-calf operation stocked at 1 animal unit (AU) /12 ha in a continuous grazing system and a similarly stocked 1 herd 8 pasture short-duration rotational grazing systems (SDG). The cattle are rotated through the pastures on a weekly basis during the growing season and every 2 weeks during the dormant season.

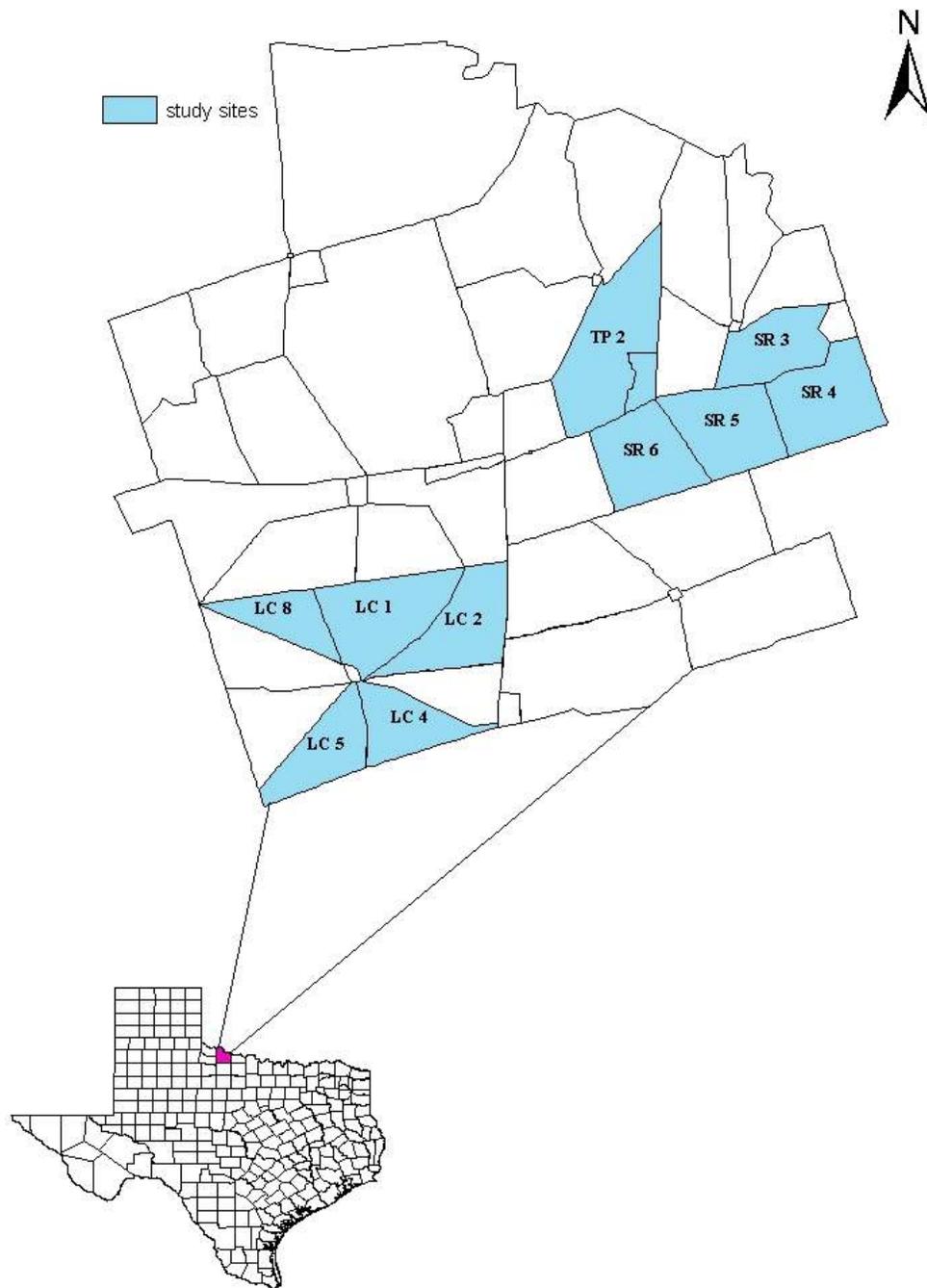


Figure 2.1. Study area used in evaluating the effects of prescribed fire on breeding grassland birds on the Waggoner Experimental Ranch, Wilbarger County, Texas, 2002–2005.

## **METHODS**

### **Study Site Selection**

Individual pastures were managed with prescribed fire to increase forage production for livestock and to control honey mesquite. Pastures were deferred from grazing for a growing season and burned once during February–March when vegetation was dormant or August–September after the nesting season. I selected 8 pastures that ranged from <1–9 years post-burn for disturbed sites and 2 pastures  $\geq 20$  years post-burn for control sites. Pastures ranged in size from 164 ha to 499 ha. Newly summer-burned pastures in 2003 and 2004 were integrated into the study design as disturbed sites to evaluate avian response to same-year fire treatments. All study sites were within the 1 herd 8 pasture short duration rotational grazing systems (9 study pastures) or the continuously grazing system (1 study pasture). All burn and control sites had replicates (2) except for the LC8 pasture where a replicate was lost due to an application of herbicide treatment by the landowner (Table 2.1). Pre-treatment data on fire-treated sites were not available given the pre-existing nature of the fire treatments. As such, my study utilizes an After-Only: Impact Trend-by-Time Interaction design described by Morrison et al. (2001:129).

Table 2.1. Study sites used to sample mean relative abundance for grassland birds, Waggoner Experimental Ranch, Wilbarger County, Texas, 2002–2005.

Study Sites	Size (ha)	$n^a$	2002	2003	2004	2005
			YSB <sup>b</sup>	YSB <sup>b</sup>	YSB <sup>b</sup>	YSB <sup>b</sup>
Unburned – LC 1	338	4	≥20	≥20	≥20	≥20
Unburned – SR 4	317	4	≥20	≥20	≥20	≥20
LC 2	230	4	6	7	8	9
TP 2	499	4	6	7	8	9
SR 5	301	4	4	5	6	7
SR 6	289	4	---	5	6	7
LC 5	225	4	4	5	6	<1
LC 8 <sup>c</sup>	164	4	2	3	4	5
LC 4	206	1	---	---	<1	2
SR 3	235	4	---	---	<1	2

<sup>a</sup> Number of transect lines within burn unit

<sup>b</sup> YSB = Years since last burn; for control sites, previous treatment type was unknown

<sup>c</sup> Pasture without a replicate

## **Abundance Sampling**

I determined the mean relative abundance (birds/km) of breeding grassland birds (hereafter, birds) for each post-burn year using line transects from 2002–2005 (Burnham et al. 1980, Buckland et al. 2001). I placed a set of 4 permanent transects in each pasture by randomly locating the first transect and systematically locating the 3 remaining transects parallel to one another and 200-m apart. In the LC4 pasture there was 1 continuous curved transect line. Transects varied in length from 0.5 km to 2.4 km depending on pasture shape and size. Transects were permanently marked with flagging tape to ensure accurate relocation on each visit (Buckland et al. 2001).

I surveyed a set of transects from 0600 to 1100 hours between May–July of each year. Surveys were not conducted in inclement weather (e.g., rain or heavy winds). In 2002 transects were visited until  $\geq 8$  km per pasture had been surveyed, in 2003 transects were visited until  $\geq 12$  km per pasture had been surveyed, and in 2004–2005 transects were visited until  $\geq 19$  km per pasture had been surveyed. I identified the birds to species and recorded the perpendicular distance from the transect line to detection point with a range finder (Burnham et al. 1980). All bird species seen were recorded (Appendix A), but only species recorded within 100 m were included in the final data set due to decreasing accuracy of range finders (Ransom and Pinchak 2003).

Surveys were conducted by 1 observer in 2002–2003, 4 observers in 2004, and 6 observers in 2005. To test for observer bias in 2004 and 2005, I used a Kruskal-Wallis test to compare the number of birds seen by each observer. The order in which transects were surveyed was alternated such that every observer had the same effort per transect to

reduce bias from the treatment effect. In order to reduce observer bias in study site abundance estimates, I had  $\geq 2$  or more observers surveying each pasture.

### **Vegetation Sampling**

Vegetation composition was measured in each pasture using the same methods each breeding season. I quantified herbaceous vegetation in June–August of each year at random locations off the bird survey transects. Sample points were determined using a 2 step process. First, I randomly chose 2 of the 4 survey transects to be walked, and a random distance was determined and paced off along each transect. A second random distance 0–15 m perpendicular to the transect was then paced to the sampling point (Noon 1981). This process was alternated from left to right of the transect line to avoid biased measurements due to trampling from conducting bird surveys. I visually estimated canopy cover (%) of grass, forbs, litter, and bare ground within a 20 cm x 50 cm quadrat at 30 sample points per study site (Daubenmire 1959). Each point was randomly sub-sampled within a 2-m radius along each of the 4 cardinal directions around the sample point. As a result, each sample point was averaged over the 5 sub-samples to obtain a mean value for a point value. Similarly, I measured visual obstruction using a 1.5 m cover pole that was delineated into 15 1-dm sections (Robel et al. 1970). The lowest strata visible from 4 m at a height of 1 m were recorded in the 4 cardinal directions. The 4 readings were averaged for each sub-sample point.

I measured post-fire density, height, and brush cover of woody shrubs from 8 permanent 60-m transects in each study site in July every two years to monitor long-term response of mesquite. I monitored woody vegetation every 2 years because previous

studies on similar soil types reported mesquite cover recovery rates of 1.1–2.2 % per year (Ansley et al. 2001). I estimated total canopy cover of all woody species using the line intercept method (Canfield 1941). Total shrub density and height were estimated using the point-center-quarter method (Mueller–Dombois and Ellenberg 1974). Every 10-m along each 60-m transect line, I recorded the distance to the nearest woody plant in each of the 4 quadrants and the species of the woody plant. The height of the closest woody plant in each of the 4 quadrants was measured to the nearest 0.1-m. All transect lines were combined and averaged to obtain a mean shrub density value for each study site.

### **Data Analyses**

*Vegetation.* I calculated the mean and coefficient of variation (*CV*) for each vegetation variable for the sub-samples within each plot; the *CV* has been used as a measure of heterogeneity of vegetation parameters (Roth 1976, Madden et al. 1999). To reduce high correlations among vegetation variables, a Spearman rank correlation coefficient ( $r_s$ ) was calculated between all vegetation variables. Variables were considered highly correlated when  $|r_s| \geq 0.80$  or  $P \leq 0.001$  (Grant et al. 2004). Eleven vegetation variables were retained for analysis (Table 2.2). Brush cover was chosen over brush height and grass cover was chosen over litter cover.

*Avian Abundance.* I calculated mean relative abundance for 5 grassland bird species (grasshopper sparrow, Cassin's sparrow, lark sparrow, dickcissel, and eastern meadowlark (*Sturnella magna*)) as the number of birds per linear km of transect. I defined total birds as all 5 species and grassland sparrow assemblage as all 3 sparrows

Table 2.2. Vegetation variables measured for 8 burned sites and 2 unburned sites on the WER, Wilbarger County, Texas, 2002–2005.

Vegetation Code	Description
<b>Structural Variables</b>	
Density	Average mesquite plants per ha
Brush cover	Average percent canopy cover of brush
<i>CV</i> density	Coefficient of variation of density
<i>CV</i> height	Coefficient of variation of brush height
<i>CV</i> cover	Coefficient of variation of brush
Grass cover	Average percent canopy cover of grass
Forb cover	Average percent canopy cover of forbs
Bare	Average percent of canopy cover of bare ground
Visual obstruction	Average visual obstruction (dm)- height/density
<i>CV</i> forb	Coefficient of variation of average forb cover
<i>CV</i> visual obstruction	Coefficient of variation of visual obstruction readings

(i.e., Cassin's sparrow, grasshopper sparrow, and lark sparrow). I estimated a mean relative abundance for each study site. As a result, the mean relative abundance for each post-burn year was averaged over the sub-samples of study sites to obtain a mean point. If a species was not observed on a site, a zero was used as the mean relative abundance.

I described bird communities in each post-burn year by species richness values, and Shannon-Wiener diversity and evenness indices. I quantified post-burn bird community similarities across treatments using Morisita's index (Horn 1966, Krebs 1999). Post-burn year communities are defined as all birds found within a single post-burn year (e.g., 2 year post-burn site). This statistical index (I) measures similarities between communities, such as overlap in seasonal patterns of abundance or nesting habitat (Horn 1966). I used descriptive statistics (i.e., confidence intervals) to investigate differences among total grassland bird abundances, individual species abundances, and species richness across fire treatments and controls.

I intended to use distance sampling methods to estimate absolute bird density, but the number of records was insufficient ( $n \leq 35$ ) to adequately estimate the detection function for individual species within each treatment and survey year. Pooling the detections across sampling years for each site would assume a constant detection function. Such an assumption was unlikely to hold because the brush height and cover increased over time, which likely affected the detectability of birds along the transect line. This posed a problem when trying to compare density estimates among post-burn years with analysis of variance because each observation has to be independent. Buckland et al. (2001) suggested dividing the density estimates by the common detection function to make the

observations independent. However, to do such would result in a comparison of encounter rates (number/km) among treatments rather than a comparison of density estimates.

### **Statistical Analysis**

Although all bird species were recorded, only the 5 breeding grassland species were included in the community statistical analyses. I used simple linear regression (Chatterjee et al. 2000) to quantify relationships between bird abundances and vegetation parameters on years since last burn. I did not include the control sites in this analysis because the data between 9 and 20 year post-burn sites were lacking and thus assumed the relationships were linear over the range of missing data. A repeated-measure ANOVA could not be used because the experimental design was unbalanced and the sample sizes within treatments were unequal.

I used stepwise multiple linear regressions to determine if mean relative abundances of individual bird species, total birds, or the sparrow assemblage were selecting study sites based on certain vegetation characteristics or field size. Regressions were performed on means of vegetation variables and mean abundance of grassland species for each burn unit.

I checked normality and residuals versus predicted value plots to test normality and constant variance. Mean bird abundance was natural-log-transformed where needed to assure constant variances (Zar 1999). I considered all statistical relationships significant at  $P \leq 0.10$  because of reduced power to detect differences due to small sample sizes. I used year-specific survey results for each site, rather than a single value to avoid

pseudoreplication within sites among years. Using a single value could confound effects of vegetation variables on bird abundances, because herbaceous vegetation variables may vary from one year to the next. Previous grassland bird studies have analyzed patterns of bird abundance over multiple years surveying the same sites and treated the data from each year as independent observations (Zimmerman 1992, Vickery et al. 1999, Swengel and Swengel 2001). All data analyses used the Social Sciences statistical software package (SPSS 2003).

## **RESULTS**

### **Vegetation and Fire**

Simple linear regression showed 4 vegetation variables to have significant relationships ( $P \leq 0.10$ ) with years since last burn (Figure 2.2). Brush cover, grass cover, and visual obstruction had a positive response and bare ground cover had a negative response to years since last burn, respectively (Figure 2.2). However, brush cover exhibited the strongest relationship to years since last burn ( $r^2 = 0.43$ ,  $df=34$ ,  $P \leq 0.001$ ). Brush density, forb cover and all *CV*'s did not exhibit significant trends with years since last burn. Annual precipitation during the 4 years vegetation variables were measured, is located in Appendix B.

### **Avian Abundance**

Grasshopper sparrows, Cassin's sparrows, and dickcissels, had a positive relationship to years since last burn and lark sparrows had a negative response to years since last burn (Figure 2.3). Eastern meadowlarks, total grassland bird, and the sparrow assemblage did not exhibit significant linear responses to years since last burn.

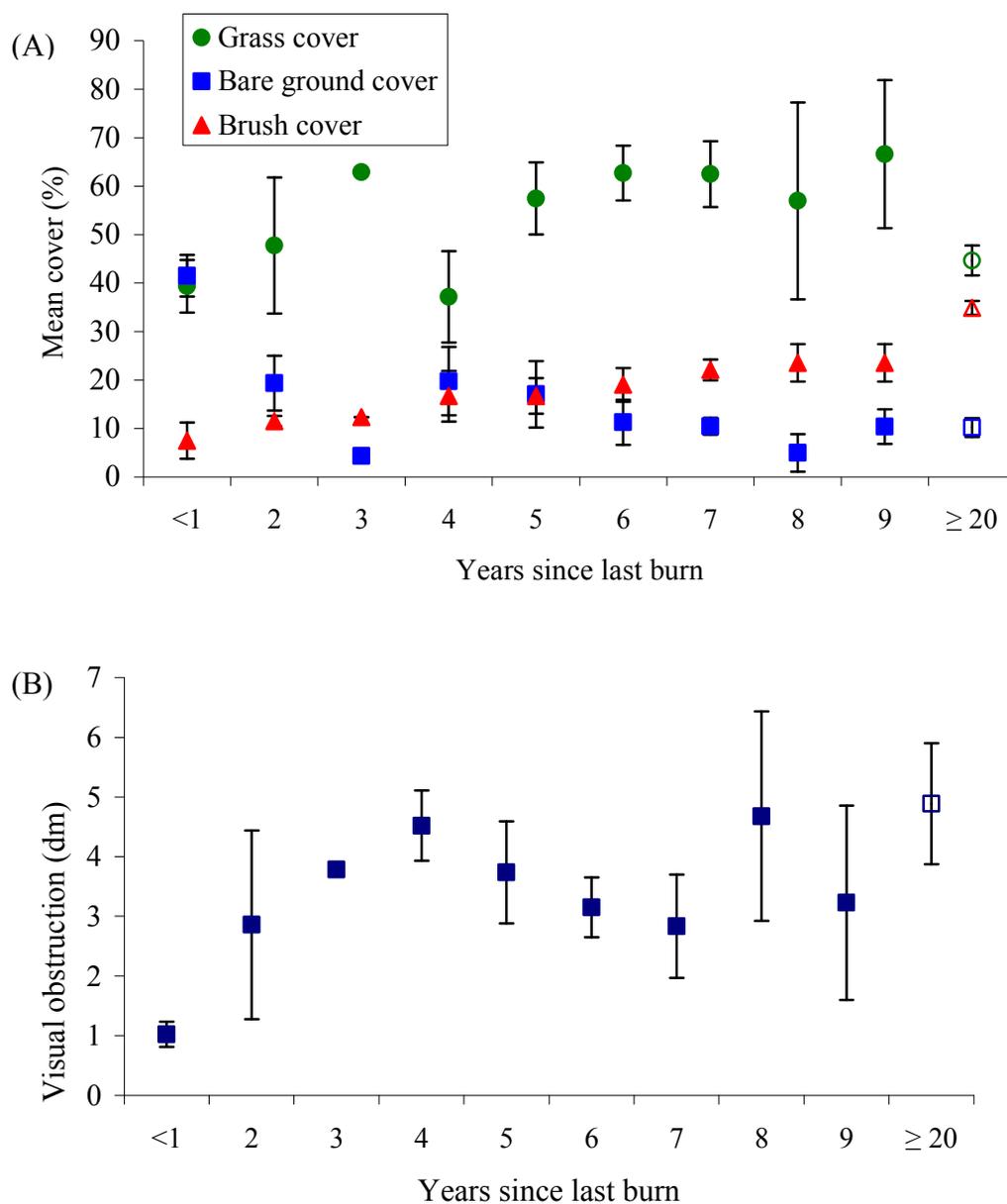


Figure 2.2. Simple linear regression of years since last burn on mean ( $\pm$ SE) grass cover, bare ground cover, and brush cover (A,  $r^2=0.21$ ,  $0.38$ ,  $0.45$ ,  $P=0.016$ ,  $\leq 0.001$ ,  $\leq 0.001$ ,  $df=26$ , respectively), and years since last burn on mean ( $\pm$ SE) visual obstruction (dm) (B,  $r^2=0.13$ ,  $P=0.062$ ,  $df=26$ ) on the WER, Wilbarger County, Texas, 2002–2005. Control sites not included in regressions ( $\blacksquare$  = burn sites;  $\square$  = control sites).

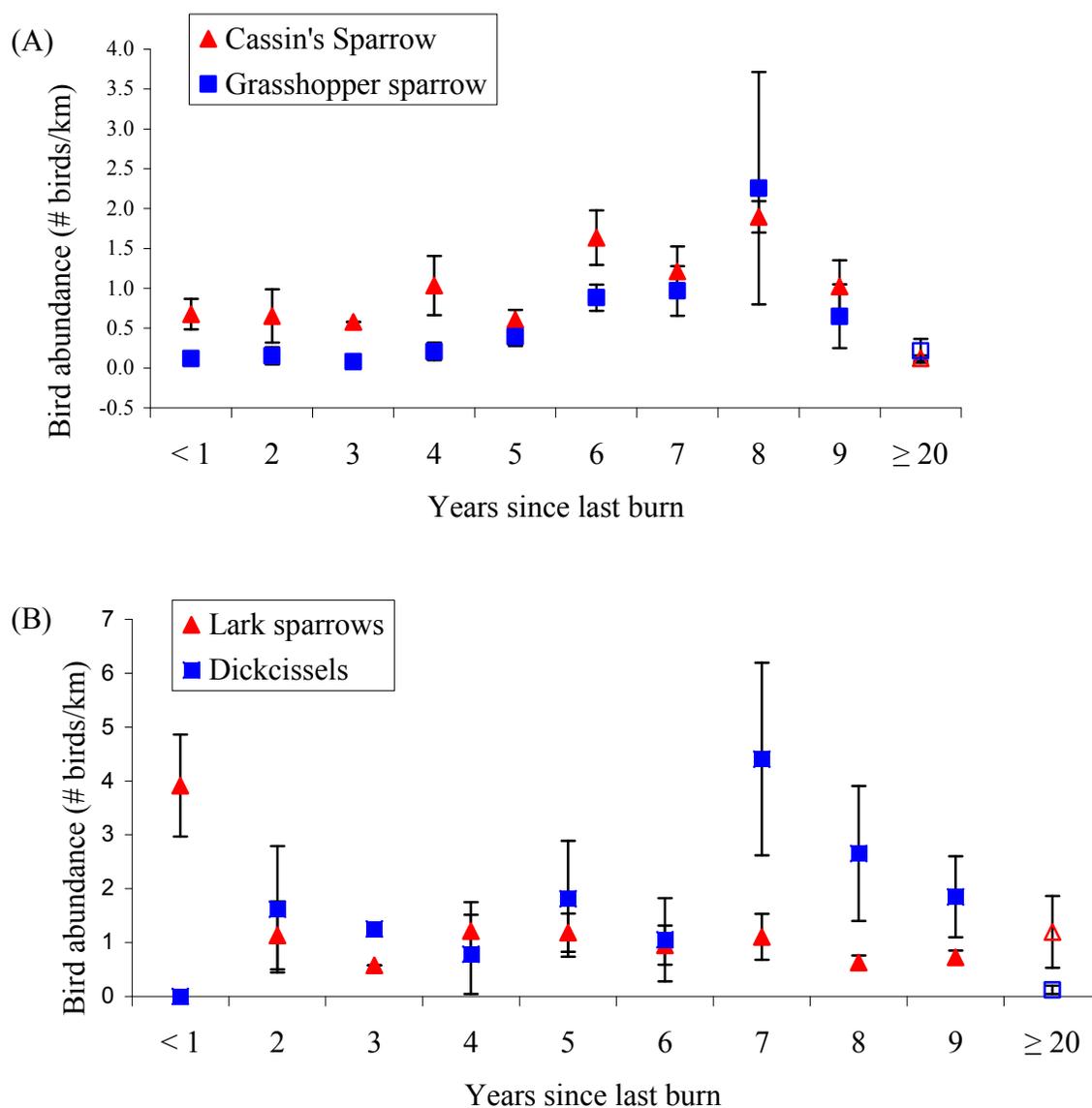


Figure 2.3. Simple linear regression of years since last burn on mean ( $\pm$ SE) abundance of Cassin's sparrows and grasshopper sparrows (A,  $r^2=0.19$ ,  $0.30$ ,  $P=\leq 0.05$ ,  $\leq 0.05$ ,  $df=26$ , respectively), and years since last burn on mean ( $\pm$ SE) abundance of lark sparrows and dickcissels (B,  $r^2=0.29$ ,  $0.14$ ,  $P=0.003$ ,  $0.059$ ,  $df=26$ ) on the WER, Wilbarger County, Texas, 2002–2005. Control sites not included in regression (■ = burn sites; □ = control sites).

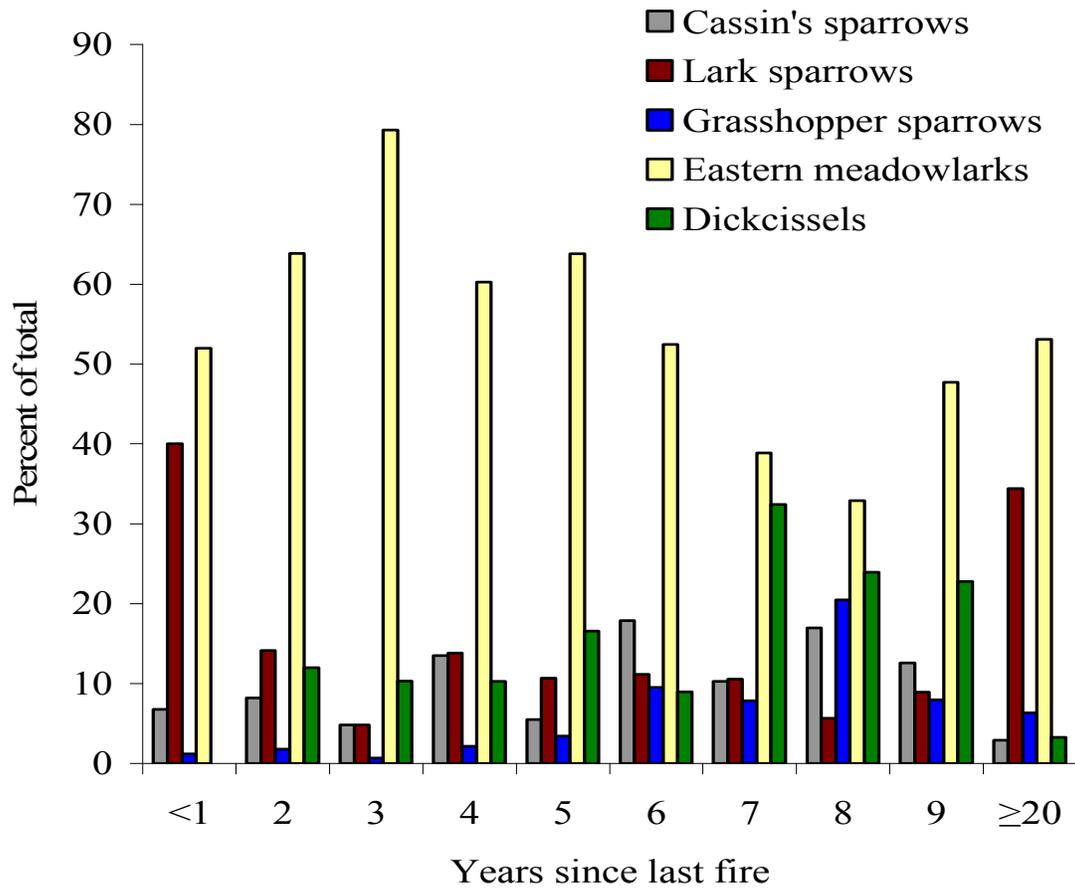


Figure 2.4. Grassland bird species composition for years since last burned ( $n = 35$ ), Waggoner Experimental Ranch, Wilbarger County, Texas, 2002–2005. The study sites are grouped by years since last burn for all 4 years, which yields 35 sample sites.

Eastern meadowlarks were the most abundant species across post-burn year categories (Figure 2.4). The youngest sites were dominated by lark sparrows and eastern meadowlarks; dickcissels and eastern meadowlarks dominated later successional stages. I did not detect significant observer bias in the number of birds observed between observers in either year ( $P = 0.33$ ,  $df = 3$ ;  $P = 0.44$ ,  $df = 5$ ).

Overall, bird species richness did not vary among burn sites, fluctuating from 4 to 5 species. Bird species diversity increased up to 8 years post-burn and then decreased (Figure 2.5). Composition of the grassland species did not differ dramatically with post-burn succession. Morisita's index of similarity values for post-burn avian communities for the 5 grassland species showed no difference with high overlap. Values of  $I$  ranged from 0.62–0.99.

### **Bird-Vegetation Associations**

Brush cover, grass cover, bare ground cover, and visual obstruction were associated with mean bird abundances. The stepwise multiple linear regression models for Cassin's sparrows, eastern meadowlarks, dickcissels, and total birds had a negative correlation with brush cover, while lark sparrows had a positive correlation with brush cover (Table 2.3). Lark sparrows and grassland sparrows were positively correlated with an increasing amount of bare ground cover; whereas, dickcissels were negatively correlated with increasing bare ground cover. Cassin's sparrows and total birds had a positive correlation with grass cover. The model did not indicate any significant predictive vegetation or habitat variables for grasshopper sparrows. No models included pasture size as a significant predictive variable.

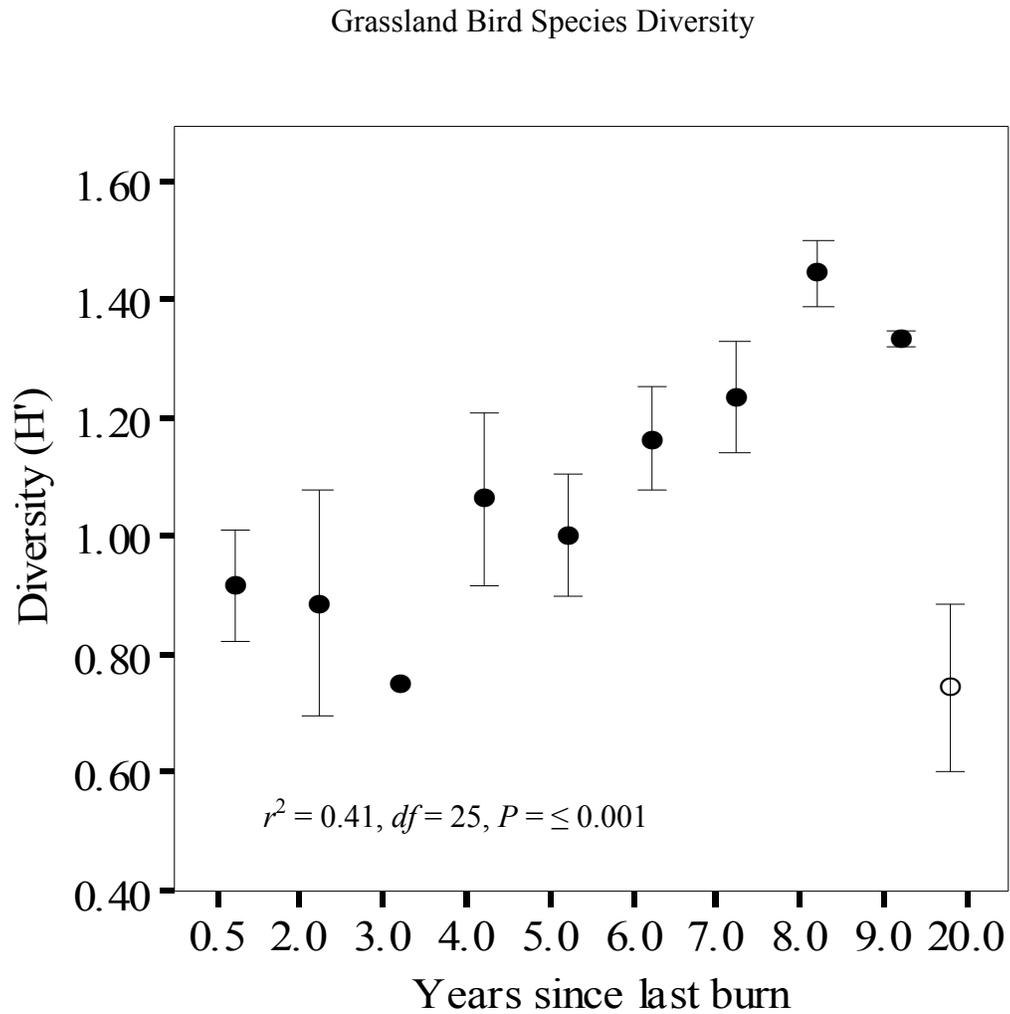


Figure 2.5. Simple linear regression of years since last burned with mean grassland bird diversity ( $\pm$ SE), WER, Wilbarger County, Texas, 2002–2005. Control sites not included in regression ( $\bullet$  = burn sites;  $\circ$  = control sites).

Table 2.3. Vegetation variables included in stepwise multiple linear regressions ( $P \leq 0.10$ ) of grassland bird mean relative abundances, WER, Wilbarger County, Texas, 2002–2005.

Bird Species	$R^2$	Vegetation Variables <sup>a,b</sup>	Partial $R^2$
Cassin's sparrow	0.292	- Brush cover	0.416
		+ Grass cover	0.341
Lark sparrow	0.69	+ Bare ground cover	0.766
		+ Brush cover	0.444
		- Visual obstruction	0.439
Eastern meadowlark	0.33	- Brush cover	0.577
Dickcissel	0.28	- Bare ground cover	0.526
		- Brush cover	0.384
All grassland birds	0.37	- Brush Cover	0.545
		+ Grass cover	0.382
Sparrow assemblage	0.27	+ Bare ground cover	0.522

<sup>a</sup> Independent variables listed in the order they were included in the model

<sup>b</sup> Response based on slope (+ or -) of regression line

## DISCUSSION

### Vegetation and Fire

Effects of fire on wildlife populations and their habitat depend on the frequency, intensity, extent, and season of fire (Whelan 1985, DeBano et al. 1998). The prescribed fires on my study were single winter (1996–2000) or single summer fires (2003–2004). The purpose of the fires was to topkill mesquite and to increase grass production for livestock. The fires produced mesquite topkill that ranged from 20% (poor)–90% (excellent) (Ansley et al. 1999). The prescribed fires were not constant in intensity, which influences the resulting post-fire vegetative community; thus, the vegetative communities present may not be indicative of the number of years post-fire.

On my study, mesquite density did not show a significant response to time since last fire. The mesquite density was relatively high ranging from 803–1318 plants/ha and maintained a patchy distribution, leaving open grass patches on all sites. The high mesquite density within a narrow range and patchy distribution was also characteristic of a semi-arid grassland in Arizona (Lloyd et al. 1998). The lack of response by mesquite density to time since last fire concurs with a past study in the Red Rolling Plains that reported non-significant honey mesquite mortality from single prescribed fires and no reduction in mesquite density (Ansley and Jacoby 1998). The average mesquite density on recently burned fields ( $\bar{x} = 993$  plants/ha) differed from unburned fields ( $\bar{x} = 1257$  plants/ha) by 21% magnitude of difference. This small difference in mesquite density could be due to the ineffectiveness of fire to achieve a good topkill on mesquite (Ansley and Jacoby 1998). Brush cover showed a positive relationship to years since last fire

with the lowest brush cover in <1 year post-burn sites and the highest brush cover in control sites (Figure 2.2 A). The positive response of brush cover to time since last fire could be due to resprouting of mesquite plants, recruitment of new mesquite plants from seed banks, clustering of mesquite patches, or short-lived topkill effects that maintain apical dominance. Past studies have suggested increased brush cover is due to fire suppression and cattle grazing (Archer et al. 1988, Scifres and Hamilton 1993). Overall, the brush cover on the burn sites ranged between 7.5–23.5% and was  $\approx$ 34% on the control sites.

Grass cover and visual obstruction increased with years since last fire, while bare ground cover decreased with years since last fire (Figure 2.2 B). The positive response by grass cover to post-fire years may be explained by a lower amount of brush cover on the burned sites. Previous studies in semiarid grasslands have noted woody cover can increase to  $\approx$  20% and still maintain herbaceous productivity (Warren et al. 1996, Ansley et al. 2004). Furthermore, brush canopy cover around 20% may increase water and nutrient availability for grass production (Scholes and Archer 1997, House et al. 2003). However, in terms of range management, my study sites  $\geq$  6 years post-burn were at the upper end or above the optimal herbaceous productivity threshold range of 0–20% brush cover (Teague et al. 2001). The negative response of bare ground cover to years since last fire may be explained by the positive response of grass cover.

The positive response in grass cover and brush cover to increasing time since last fire may explain the increase in visual obstruction. In addition the positive response of visual obstruction could be related to the amount of precipitation received each year.

The annual precipitation can be extremely variable and influences the characteristic plant forms found in grasslands (e.g., short grass prairies). In extremely dry years the productivity of herbaceous vegetation variables (i.e., biomass) may be reduced, thus decreasing the height and density of grasses and forbs. I did not measure the impact grazing had on herbaceous cover and visual obstruction, which could confound the relationship to time since last burned. However, I did not visually observe a difference in herbaceous height due to grazing.

### **Avian Abundance**

Mcpherson (1995) and Whelan (1995) reported bird species response to fire was closely linked to the effect of fire on the vegetation community. Furthermore, Fuhlendorf and Engle (2001) noted that the grassland avian guild sustainability was linked to the heterogeneity of vegetation structure. In my study, the prescribed fires were incomplete in their coverage and left unburned patches creating a mosaic of mesquite stands and mesquite grassland patches. I believe this contributed to grassland birds being present in later successional stages and the limited change in the grassland avian community as a whole. Similarly, Zimmerman (1993) noted dickcissels, grasshopper sparrows, and eastern meadowlarks did not decrease in total numbers due to annual or less frequent fires. Species richness was highest among burn sites and lowest in unburned sites ( $\geq 20$  years post-burn). Other studies have reported species richness highest on sites associated with lack of fire, but have included non-grassland species associated with woody habitats (Arnold and Higgins 1986, Zimmerman 1992). Species richness of breeding grassland birds ranged from 4–5 among burned sites. This finding

was consistent with other studies that characterize breeding grassland bird communities as having low diversity and density with relatively constant annual species richness (Cody 1966, Wiens 1973).

Abundances of grasshopper sparrows, Cassin's sparrows, and dickcissels were positively related to time since last burned, but were reduced in numbers on the control sites (Figure 2.3 A, B). Lark sparrows decreased with time since last burn (Figure 2.3 B). The response to fire may be related to each species habitat requirements in relation to vegetative parameters. For example, lark sparrows have a negative association with grass cover and litter cover (Renwald 1977), which is typically associated with earlier successional stages. Cassin's sparrows, which prefer high amounts of grass cover for foraging and nesting (Bock and Webb 1984), increased with time since fire, while mean grass cover increased and bare ground cover decreased with time since fire (Figure 2.2 A).

Species diversity is the number of species in an area weighted by the number of individuals for each species, thus species richness and species evenness. The general increase in species diversity (Figure 2.5) with time since last burned appears to be attributed to species evenness because species richness fluctuated from 4–5 species over time. The distribution among the number of individuals for each species increased with years since last burned. As the distribution of species became less even, the diversity index ( $H'$ ) increased. The increase in grassland bird diversity has been associated with increasing spatial variability in vegetation variables (Roth 1976). In addition the grassland patches suitable for nesting and foraging should increase with more spatial

variability in vegetation parameters. The number of patches suitable in relation to each species specific habitat requirements may shift, thus the distribution of species becomes more dissimilar.

I found vegetation variables correlated to the abundance of all birds except grasshopper sparrows. Past studies have suggested response of breeding grassland birds to fire is related to the natural history requirements of each species (Bock and Bock 1992, Kirkpatrick et al. 2002). Vegetation variables important to relative abundance concurred with general habitat preferences reported for each species. For example, lark sparrow abundance increased with more bare ground cover and brush cover, and have been noted to prefer low litter and low grass cover (Renwald 1977), and sparse vegetation with scattered small trees (Martin and Parrish 2000). Cassin's sparrow abundance was correlated with lower brush cover and more grass cover. Similarly, Bock and Webb (1984) observed Cassin's sparrows on sites with high grass cover and scattered mesquite bushes. Eastern meadowlark abundance increased with reduced amounts of mean brush cover. My eastern meadowlark findings concurred with a previous study in the mixed-grass prairies, where meadowlarks were negatively associated with visual obstruction and shrub cover (Madden et al. 1999). Similarly, dickcissel abundance was negatively correlated with bare ground cover and brush cover. My dickcissel results were similar to other studies that indicate abundance was correlated positively with grass cover and negatively with bare ground cover (Wiens 1973, Rotenberry and Wiens 1980). Total grassland bird abundance was positively associated with grass cover and negatively associated with brush density. The positive

association with grass cover and the negative association with brush cover are not surprising with grassland birds preferring open grasslands as optimal habitat for breeding and foraging. The grassland sparrow assemblage was positively correlated with bare ground cover, which could be due to lark sparrows and grasshopper sparrows preferring to forage and nest in areas containing bare ground. The variability in habitat requirements for each species suggests a heterogeneous (i.e., variability in vegetation height and density) grassland is necessary to maintain the grassland avian guild (Fuhlendorf and Engle 2001).

I observed that brush cover increased with time since last fire to the higher end of the herbaceous productivity threshold between 7–9 years post-fire (Figure 2.2 A), and total grassland bird numbers decreased with increasing brush cover (Table 2.4). Although, birds responded positively or negatively to time since last burn, all birds were present on <1–9 year post-burn sites to varying degrees. The presence of grassland birds across post-burn years suggests the single burns on a 5–6 year rotation were able to maintain suitable habitat for grassland bird populations. Prescribed fire may not be able to fully restore a shrub-invaded grassland to a pristine grassland state, but may be able to maintain grassland-bird populations at these local scales by creating vegetation communities at various successional stages. Similarly, Fuhlendorf and Engle (2001) suggested that restoring rangelands using fire and grazing to a more heterogeneous state would maintain a diverse community of the grassland avian guild by providing various successional stages in vegetation structure.

## **MANAGEMENT IMPLICATIONS**

In my study, changes in mean bird abundance were associated with vegetation variables that differed among bird species. Therefore there is no single management strategy to simultaneously increase abundance for all 5 study species, which may pose logistical challenges for ranchers interested in maintaining grassland bird populations as a secondary goal. To maintain herbaceous productivity and improve visibility for livestock a single large scale (i.e., pasture level) fire is the most efficient and cheapest management tool. On a rotational grazing system with several pastures prescribed fire can be implemented on multiple pastures every 5–6 years to manage for livestock and grassland birds in the Red Rolling Plains (Teague et al. 1997). Results from this study suggest a burn regime of 5 years might be adequate to maintain habitat conditions for grassland bird communities.

**CHAPTER III**  
**PRESCRIBED FIRE AND GRASSLAND BIRD NESTING ECOLOGY**  
**IN MESQUITE SAVANNA HABITATS**

**INTRODUCTION**

Before European settlement of the North American grasslands, the Great Plains comprised the largest vegetative province in North America and was estimated to cover 162 million ha of land (Knopf 1994). Grassland systems have decreased in size, with the mixed grass prairie declining by 72–99% and the tallgrass prairie declining 83–99% from their previous distributions due to the cumulative effects of overgrazing, suppression of fire, and conversion to cropland (Knopf 1996). For the past 25 years, North American grassland birds have declined more than any other avian guild (Knopf 1994, Peterjohn and Sauer 1999). Breeding Bird Survey (BBS) data have shown that 14 of 19 widespread North American grassland bird species and 6 of 9 endemic species are in decline (Knopf 1996). Many grassland-dependent species such as the dickcissel, grasshopper sparrow, and Cassin's sparrow are Partners in Flight species of concern (Carter et al. 1996), and have shown declining breeding population trends from 1966–1996 (Peterjohn and Sauer 1999). Habitat fragmentation, loss, and degradation (e.g., woody encroachment) are cited as the main causes for observed declines (Johnson and Igl 2001, Vickery and Herkert 2001).

Implementing prescribed fire, as a restoration tool for grassland bird communities, should influence grassland bird community demographics throughout post-burn years by providing various stages of vegetation succession. In shrub-invaded grasslands, fire can

alter shrub density, canopy cover and height, and in some cases shrub density, which could lead to the reestablishment of a grassland matrix. Many woody plants (e.g., mesquite) are top-killed by fire, but have the ability to re-sprout with a multi-stem growth form (Scifres and Hamilton 1993). Thus, after an initial fire, vegetation may still approximate a shrubland physiognomy rather than true grassland. It remains to be seen if grassland-dependent birds will inhabit burned sites. Previous studies reported a decline in grassland bird abundance immediately following a burn on shrub-invaded treatment plots (Reynolds and Krausman 1998, Kirkpatrick et al. 2002). However, frequent prescribed fires may enhance breeding-bird populations by increasing plant biomass for nest concealment, decreasing predation (Johnson and Temple 1990), and/or increasing insect densities for food available to nestlings (Evans 1988).

Past studies have estimated grassland bird abundance and related this to habitat features measured within their study areas (Herkert 1994, Madden et al. 1999). It has been suggested that density estimates alone may produce misleading conclusions about habitat quality (Van Horne 1983). One problem could be time lags resulting from species-specific site fidelity (Rotenberry and Wiens 1978), which might obscure expected short-term responses to habitat disturbances (Wiens and Rotenberry 1985, Wiens et al. 1986). For example, increased brush density may continue to attract breeding grassland birds, but may be an ecological trap resulting in lower reproductive success due to predation and parasitism (Ratti and Reese 1988). Remes (2003) found higher breeding densities of blackcaps (*Sylvia atricapilla*) in exotic black locust plantations than native floodplain forests. However, reproductive success was lower in

black locust plantations (15.5%) than in native floodplains (59%). Thus, prescribed fire implemented for shrubland to grassland conversions must also be evaluated in terms of nest survival and fledging success and over longer time frames (e.g., >5 years) (Johnson and Temple 1984, Johnson and Temple 1990, Kirkpatrick et al. 2002).

To effectively manage remaining grasslands, a better understanding is needed of the relationships among disturbance regimes (e.g., fire), resultant vegetation changes, and grassland-bird communities. The objective of this study was to examine the effects of prescribed fire succession on grassland bird reproductive success along a <1–9 year post-burn continuum in a mesquite shrubland of the Texas Rolling Plains. My research focused on answering the following questions:

- 1) How does grassland bird nest success vary over the stages of fire succession in a mesquite dominated shrubland?
- 2) What vegetation characteristics are associated with nest-site selection by grassland birds on the Waggoner Experimental Ranch?

## **STUDY AREA**

The study was conducted on the 12,141 ha Kite Camp of the Waggoner Experimental Ranch (WER) in Wilbarger County, near Vernon, Texas (Chapter II, Figure 2.1). The region is characterized by a continental and semiarid climate with an average growing season of 220 days. Annual precipitation averages 64.8 cm with seasonal peaks in May and September. Ambient temperatures range from -2.3 C° in January to 36.4 C° in July. Topography is moderately rolling with elevations ranging from 335–396 m (Gould 1975).

The landscape is characteristic of the Red Rolling Plains ecological region and was historically a grassland or grassland savanna maintained by periodic wildfire. Currently, the WER is a shrubland savanna dominated by honey mesquite and lotebush in various stages of succession resulting from application of herbicides and prescribed fire (Gould 1975). Herbaceous species are a mixture of cool and warm season grasses including Texas wintergrass, Texas bluegrass, Japanese brome, buffalograss, silver bluestem, meadow dropseed, and sideoats grama (Gould 1975). The Kite Camp is a cow-calf operation stocked at 1 animal unit (AU)/12 ha in a continuous grazing system and a similarly stocked 1 herd 8 pasture short duration rotational grazing systems (SDG).

## **METHODS**

### **Study Site Selection**

Individual pastures were managed with prescribed fire to increase forage production for livestock and to control honey mesquite. Pastures were deferred from grazing for a growing season and burned once during February–March when vegetation was dormant or August–September after the nesting season. I selected 8 pastures that ranged from <1–9 years post-burn for disturbed sites and 2 pastures  $\geq 20$  years post-burn for control sites with closed shrub canopies. Pastures ranged in size from 164 ha to 499 ha. Newly burned pastures in 2003 and 2004 were integrated into the study design as disturbed sites to evaluate avian response on same year fire treatments. All study sites were within the 1 herd 8 pasture short duration rotational grazing systems (9 study pastures) or the continuously grazing system (1 study pasture). All burn and control sites had replicates (2) except for the LC8 (Long Creek) pasture where a replicate was lost due to an application of herbicide treatment by the landowner (Chapter II Table 2.1). Pre-treatment data on fire treated sites were not available given the pre-existing nature of the fire treatments. As such, my study utilizes an After-Only: Impact Trend-by-Time Interaction design described by Morrison et al. (2001:129).

### **Nest Monitoring**

I located and monitored dickcissel, eastern meadowlark, Cassin's sparrow, grasshopper sparrow, and lark sparrow nests from April–July of 2004 and 2005 following U.S. Department of Agriculture Forest Service guidelines (Ralph et al. 1993). I located 4 nest plots (100 x 100 m) along each transect in each study site. I

systematically searched the study sites using a sweeping pole (Winter et al. 2003). Each nest plot received equal search effort and areas outside the nest plots were also searched every 2–3 days. Nests were also located opportunistically when birds flushed or were observed carrying food or nesting materials. Nest locations were marked with a global positioning system (GPS) unit and flagged approximately 10 m north of the nest to reduce disturbance (Ralph et al. 1993). I monitored nests every 3–5 days to determine the clutch size and nest fate. When nests became inactive, I categorized nest fate as successful (fledged) or failed (depredated or abandoned). A nest was considered to be successful if field observations provided evidence that  $\geq 1$  young fledged (Johnson and Temple 1990, Martin and Geupel 1993). Each nest was considered an independent observation.

### **Vegetation Sampling**

I measured vegetation at nest sites and random sites after the nest became inactive. The random plots were located by pacing a predetermined random distance (5–30m) and cardinal direction from the nest site. I measured 2 random sites for every nest site to reduce the variability of measuring an atypical random site (Sutter 1997). I limited the distance to 30m to avoid placing the random site in a distinctly different vegetative community (e.g., underneath a mesquite plant, mesquite stand). All plots were assessed within 7 days of nest completion. At the nest site I recorded the nest substrate species, substrate height, vegetative height above nest, distance to the closest woody plant, and percent nest concealment in the 4 cardinal directions (Ralph et al. 1993). All nest site herbaceous vegetation parameters were measured at the center of the nest and at random

locations near the nest in each of the 4 cardinal directions from the nest. The 5 values for each vegetation attribute were then averaged for a mean value. For both nest plots and random plots, I measured visual obstruction using a 1.5 m cover pole that was delineated into 15 1-dm sections (Robel et al. 1970). The lowest strata visible from 4 m at a height of 1 m were recorded in the 4 cardinal directions. The readings were then averaged for an overall vertical structure estimate for each plot. I determined canopy cover (%) of grass, forb, bare ground, and litter using a 20 cm x 50 cm Daubenmire sampling frame (Daubenmire 1959). Each variable consisted of 4 values that were averaged to obtain a mean value.

### **Data Analysis**

*Nest Success.* Nest success was estimated for the entire nesting and fledging period using program MARK (White and Burnham 1999), which takes unknown failure dates into account. Nest survival was calculated for each post-burn year and for each sampling year (Krebs 1999) and then compared using program CONTRAST (Hines and Sauer 1989). I used stepwise discriminant function analysis for grasshopper sparrows, Cassin's sparrows, lark sparrows, eastern meadowlarks, and dickcissels to determine which vegetation variables were significantly different between successful and unsuccessful nests (Warren and Anderson 2005). I used a significant level of entry (SLE) into the model of 0.25 for vegetation variables. To reduce high correlations among vegetation variables, a Spearman rank correlation coefficient ( $r_s$ ) was calculated between all vegetation variables. Variables were considered highly correlated when  $|r_s| \geq 0.80$  or  $P \leq 0.001$  (Grant et al. 2004). The vegetation variables retained for

analysis were % grass cover, % forb cover, % bare ground, visual obstruction, nest height, substrate height, vegetative height above nest, distance to the closest woody plant, and percent nest concealment.

*Nest-site Selection.* –Nest-site characteristics were evaluated by comparing vegetation variables at nest sites with random sites. I used discriminant function analysis to determine which vegetation variables were different between nest sites and random sites (Warren and Anderson 2005). The same model approach used with nest success was applied to evaluate nest-site selection. The vegetation variables used for analysis were % grass cover, % forb cover, % bare ground, and visual obstruction. I used simple linear regression to evaluate the relationship between the number of nests and vegetation parameters (i.e., woody and herbaceous).

All percentage data were transformed by arcsine square root; height data were transformed by log for data analysis. I checked normality and residuals versus predicted value plots to test normality and constant variance. I considered all statistics significant at  $P \leq 0.10$  due to small sample size. All data analyses were done using the Social Sciences statistical software package (SPSS 2003).

## **RESULTS**

### **General Nesting**

I found a total of 90 nests during 2004 ( $n = 36$ ) and 2005 ( $n = 54$ ) for the 5 bird species: grasshopper sparrows ( $n = 5$ ), Cassin's sparrows ( $n = 22$ ), lark sparrows ( $n = 27$ ), eastern meadowlarks ( $n = 22$ ) and dickcissels ( $n = 14$ ). I found 81 nests in burned pastures (grasshopper sparrows ( $n = 5$ ), Cassin's sparrows ( $n = 22$ ), lark sparrows ( $n =$

21), eastern meadowlarks ( $n = 19$ ) and dickcissels ( $n = 14$ )), and 9 nests in unburned pastures (lark sparrows ( $n = 6$ ) and eastern meadowlarks ( $n = 3$ )). I found 9 nests in <1 year post-burn sites, 21 nests in 2 year post-burn sites, 1 nest in the 4 year post-burn site, 1 nest in the 5 year post-burn site, 13 nests in 6 year post-burn sites, 18 nests in 7 year post-burn sites, 12 nests in 8 year post-burn sites, 6 nests in 9 year post-burn sites, and 9 nests in  $\geq 20$  year post-burn sites. Mean clutch size was 3.4 (SE = 0.5) for grasshopper sparrows, 3.9 (SE = 0.3) for Cassin's sparrows, 3.9 (SE = 0.2) for lark sparrows, 4.6 (SE = 0.2) for eastern meadowlarks, and 3.8 (SE = 0.3) for dickcissels.

### **Nest Success**

Overall survival rates for nests did not differ among post-burn years ( $\chi^2 = 2.992$ ,  $df = 8$ ,  $P = 0.93$ ) or between 2004 and 2005 ( $\chi^2 = 0.046$ ,  $df = 1$ ,  $P = 0.83$ ), so nest data were pooled for 2004–2005. Nest success for individual species ranged from 14% for Cassin's sparrows to 100% for grasshopper sparrows (Table 3.1). During my study the main cause for unsuccessful nests was depredation (90%). Nest success for individual species was not tested due to small sample sizes across post-burn years.

I found vegetation variables at nest sites that were correlated with nest success for lark sparrows, eastern meadowlarks, and dickcissels. Successful lark sparrow nests had lower percent grass cover than depredated nests ( $F = 3.97$ ,  $df = 1$ ,  $P = 0.06$ ) (Table 3.2). Visual obstruction was taller at depredated nests than at successful nests for eastern meadowlarks ( $F = 3.47$ ,  $df = 1$ ,  $P = 0.08$ ) and dickcissels ( $F = 6.60$ ,  $df = 1$ ,  $P = 0.03$ ) (Table 3.2). Bare ground cover was greater at unsuccessful dickcissel nests than at successful nests ( $F = 5.14$ ,  $df = 1$ ,  $P = 0.04$ ) (Table 3.2).

Table 3.1. Number of nests, percent of the total nests found, and nest success for the five grassland species on the Waggoner Experimental Ranch, Wilbarger County, Texas, 2004–2005.

Species	<i>n</i> <sup>a</sup>	% of total nests	Nest success
Grasshopper sparrow	5	5.6	1.00
Cassin's sparrow	22	24.4	0.14
Lark sparrow	27	30.0	0.20
Eastern meadowlark	22	24.4	0.19
Dickcissel	14	15.6	0.26

<sup>a</sup>Number of nests observed

Table 3.2. Mean  $\pm$ SE of vegetation variables that differed significantly ( $P \leq 0.10$ ) between successful and unsuccessful nests using stepwise discriminant analysis on the Waggoner Experimental Ranch, Wilbarger County, Texas, 2004–2005.

Species	Variables	Successful nests			Unsuccessful nests			F	P
		n	$\bar{x}$	SE	n	$\bar{x}$	SE		
Lark sparrow	Grass cover (%)	7	26.80	8.79	20	41.89	6.07	3.97	0.06
Eastern meadowlark	Visual obstruction (dm)	7	1.84	0.08	15	2.09	1.00	3.47	0.08
Dickcissel	Visual obstruction (dm)	4	2.72	0.54	10	4.90	1.00	6.60	0.03
	Bare ground cover (%)		0.10	0.06		1.16	0.61	5.14	0.04

### **Nest-site Selection**

Vegetation variables differed between nest sites and random sites for all species. Grasshopper sparrow nest sites had greater visual obstruction ( $F = 3.93$ ,  $df = 1$ ,  $P = 0.07$ ) and higher percent forb cover ( $F = 2.93$ ,  $df = 1$ ,  $P = 0.09$ ) than random sites (Table 3.3). Cassin's sparrows had taller visual obstruction ( $F = 28.69$ ,  $df = 1$ ,  $P = \leq 0.001$ ) and higher percent grass cover ( $F = 2.06$ ,  $df = 1$ ,  $P = \leq 0.001$ ) at nest sites than at random sites (Table 3.3). For lark sparrows, visual obstruction was taller ( $F = 12.60$ ,  $df = 1$ ,  $P = \leq 0.001$ ) and percent grass cover was lower ( $F = 7.62$ ,  $df = 1$ ,  $P = \leq 0.001$ ) at nest sites compared to random sites (Table 3.3). Eastern meadowlarks had higher percent grass cover ( $F = 2.83$ ,  $P = 0.10$ ) and percent forb cover ( $F = 2.98$ ,  $df = 1$ ,  $P = 0.06$ ) at nest sites compared to random sites (Table 3.3). Dickcissels had taller visual obstruction ( $F = 20.34$ ,  $df = 1$ ,  $P = \leq 0.001$ ) and higher percent forb cover ( $F = 11.67$ ,  $df = 1$ ,  $P = \leq 0.001$ ) at nest sites than at random sites (Table 3.3). The number of nests found in each post-burn site was positively correlated with increasing heterogeneity of brush density ( $CV\%$ ) (Figure 3.1).

## **DISCUSSION**

### **Nest Success**

I utilized a pre-existing design from a management practice of replicated pastures subjected to prescribed fire at different times to evaluate grassland-bird nest success and nest-site selection. Nest success was similar among post-burn years suggesting that although prescribed fire altered the vegetative structure, the time since burning did not

Table 3.3. Mean  $\pm$ SE of vegetation variables that significantly ( $P \leq 0.10$ ) differed between nest sites and random sites using stepwise discriminant analysis on the Waggoner Experimental Ranch, Wilbarger County, Texas, 2004–2005.

Species ( <i>n</i> )	Variables	Nest Site		Random Site		F	<i>P</i>
		$\bar{x}$	SE	$\bar{x}$	SE		
Grasshopper sparrow ( <i>n</i> = 5)	Visual obstruction (dm)	1.95	0.21	1.39	0.16	3.93	0.07
	Forb cover (%)	2.60	1.25	0.90	0.73	2.93	0.09
Cassin's sparrow ( <i>n</i> = 22)	Visual obstruction (dm)	2.99	0.28	1.53	0.12	28.69	$\leq 0.001$
	Grass cover (%)	72.78	3.27	70.66	3.75	20.06	$\leq 0.001$
Lark sparrow ( <i>n</i> = 27)	Visual obstruction (dm)	2.57	0.50	1.24	0.11	12.60	$\leq 0.001$
	Grass cover (%)	37.98	5.12	43.84	3.95	7.62	$\leq 0.001$
Eastern meadowlark ( <i>n</i> = 22)	Grass cover (%)	88.88	2.81	82.80	2.74	2.83	0.10
	Forb cover (%)	5.53	1.35	3.55	0.67	2.98	0.06
Dickcissel ( <i>n</i> = 14)	Visual obstruction (dm)	4.23	0.75	2.03	0.20	20.34	$\leq 0.001$
	Forb cover (%)	4.59	1.87	3.61	1.13	11.67	$\leq 0.001$

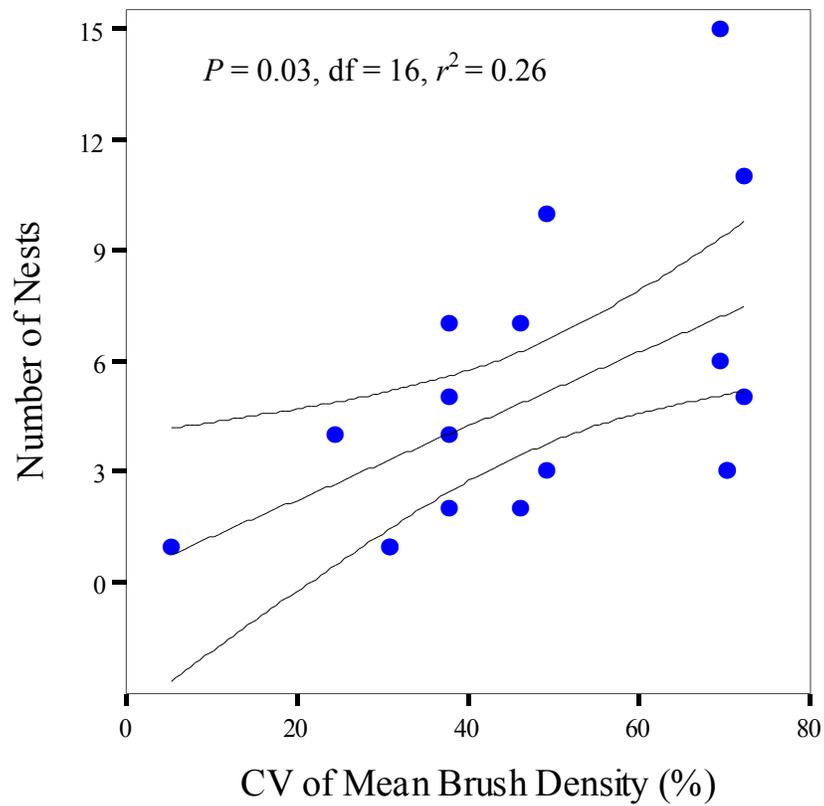


Figure 3.1. Simple linear regression of the number of nests and CV (%) of mean brush density with 90% confidence intervals, Waggoner Experimental Ranch, Wilbarger County, Texas, 2004–2005.

influence overall nest success. Similarly, Zimmerman (1997) noted no change in reproductive success for dickcissels, eastern meadowlarks, and grasshopper sparrows despite fire frequency. It has been suggested that the lack of grassland bird response to prescribed fire is due to fire being a natural disturbance in the grassland system (Zimmerman 1997). Although not significantly different, the overall nest success was highest on the 2 year post-burn sites ( $\approx 30\%$ ) and lowest on the  $\geq 20$  year post-burn site ( $\approx 10\%$ ), which lends support to past studies that noted an increase in reproductive success on areas that were recently burned (Zimmerman 1992, Shriver and Vickery 2001, Delaney et al. 2002).

The effect prescribed fire has on the herbaceous vegetation and woody plants (i.e., cover and density) coupled with the woody plant regrowth rate should influence the amount of suitable nesting habitat present. Overall, my study sites had relatively low brush canopy cover, ranging from 7.5–34.9%. Prescribed fire may provide a mosaic of vegetative communities (Riggs et al. 1996), thus creating a mix of open grasslands or grassland savanna and scattered mesquite. The patchy shrub distribution with the low brush cover rate and grassland or grassland savanna patches may have provided suitable nesting habitat for all grassland bird species. On my study site,  $\geq 20$  year post-burn sites contained open, grassy patches, which provided limited nesting habitat. In my study, post-burn years did not significantly affect the probability of nest success for grassland birds, but vegetation characteristics seemed to influence nest success and suitable nesting habitat.

I found 3 vegetation variables that were important for predicting nest success: visual obstruction, grass cover (%), and bare ground cover (%) (Table 3.2). Successful lark sparrow nests had lower percent grass cover compared to unsuccessful nests. This was not surprising, as lark sparrows have been reported to prefer sites with bare ground and short grass height (Bock and Webb 1984). Similarly, Luske et al. (2003) reported nest success was predicted best by less bare-ground and more litter cover. For dickcissels lower visual obstruction and bare-ground cover differed between successful and unsuccessful nests. The low bare ground cover may provide more nest concealment by increased amounts of herbaceous cover, thus reducing potential nest predation. Eastern meadowlarks had shorter visual obstruction at successful nests compared to depredated nests. This agrees with past descriptions of their breeding habitat as Schroder and Sousa (1982) reported nest sites occurred with lower visual obstruction in more heterogeneous habitats.

The important vegetative parameters for successful nests (i.e., shorter visual obstruction and lower grass cover (%)) reduced nest concealment. The selection for lower visual obstruction and lower percent grass cover may seem counterintuitive, but this may provide a wider view of the surrounding area for better surveillance of predators and a more efficient escape route (Götmark et al. 1995). The potential trade off between reduced nest concealment for increased predator avoidance is important, because predation is thought to be the primary cause for nest failure in birds (Martin 1993). Natural selection should favor selection of nest sites and life history traits that reduce nest predation risk and increase reproductive success.

### **Nest-site Selection**

Vegetative parameters selected for nest sites were greater than random sites (Table 3.3). The only exception was nest sites for lark sparrows, which had lower selection of grass cover than random sites. The 5 grassland birds often overlapped in their vegetation parameters for nest sites. Grasshopper sparrows, Cassin's sparrows, and dickcissels all selected nest sites with taller visual obstruction and more forb cover or grass cover compared to random sites, which agrees with past descriptions of their nesting habitat (Whitmore 1981, Maurer et al. 1989, Roth 1980). Grasshopper sparrows have been associated with breeding territories characterized by low to intermediate height and intermediate forb cover (Whitmore 1981, Herkert et al. 1993). Nest sites of Cassin's sparrows were associated with intermediate vegetation height, grass cover and small shrubs (Maurer et al. 1989). Dickcissels have been associated with structurally complex nest sites surrounded by dense herbaceous cover (Roth 1980). Furthermore, Zimmerman (1993) noted male territories with greater forb and grass cover attracted more females. The selection of nest sites by these 3 species with taller visual obstruction and more forb cover or grass cover provides greater nest concealment which should reduce predation risks (Martin 1993). In addition, greater forb cover could provide singing perches and potentially support higher insect numbers (Dechant et al. 2003).

Eastern meadowlarks selected nest sites with more grass and forb cover than random sites. Eastern meadowlarks are known to select nest sites in short to intermediate vegetation height with nests placed under grass or forb clumps (Skinner et al. 1984). The increased grass and forb cover provides additional nest concealment which should

reduce predation rates, increase nesting materials, and increase the number of singing perches. As mentioned previously, areas with increased forb cover have been suggested to support more insects for foraging (Dechant et al. 2003).

Nest sites of lark sparrows had lower grass cover and higher visual obstruction compared to random sites. Similarly, previous studies have reported lark sparrow nests in areas with sparse ground cover in depressions located at the base of small shrubs (Zimmerman 1993, Martin and Parrish 2000). The reduced nest concealment from low grass cover may be countered by the increased visual obstruction. The higher visual obstruction should provide nest concealment from potential predators and protection from rain or heat.

I found that greater vegetative visual obstruction was important for nest-site selection for 4 of the 5 grassland species. While visual obstruction was lower at successful nests for 3 species and had no significant difference for the remaining 2 species. These seemingly contradictory results suggest that the grassland birds in the Kite Camp are selecting nest sites with greater vertical density, while successful nests had lower vertical density. One reason for the contradictory results may be the presence of a diverse predator community, which might reduce the chance of safe nest sites being related to specific vegetation parameters. Dion et al. (2000) noted small mammals predated nests in short grass with sparse herbaceous cover, while medium sized predators predated nests in taller vegetation. Thus, the nest-site selection variables may be confounded due to a variety of predators searching subhabitats with different vegetation characteristics (Filliater et al. 1994). In addition, small sample size of nests for each species coupled

with the smaller sample size for successful nests (e.g., lark sparrow,  $n = 7$ ) compared to unsuccessful nests (e.g., lark sparrow,  $n = 20$ ) also make conclusions problematic (Table 3.2). Bird species preference for specific vegetation parameters has been well documented (Zimmerman 1993, Herkert 1994, Madden et al. 1999, Rohrbaugh et al. 1999) and is related to their life history traits.

The number of nests per site was positively correlated with the increasing spatial variability of mesquite density ( $CV$ ) (Figure 3.1), suggesting that prescribed fires on my study sites might sustain open grassland patches suitable for grassland bird nesting. Suitable habitat for all 5 grassland species has been associated with sparse shrub density or shrub savanna (Zimmerman 1993, Ruth 2000). Current theory regarding nest success and productivity has centered on native grasslands or fragmented grasslands with forested borders. The consistent theme has been reproductive success is greater with increasing distance from woody cover (Gates and Gysel 1978, Johnson and Temple 1990, Burger et al. 1994); however, I did not find a significant relationship between distance of woody cover and nest success, which may be due to the contiguous landscape of the study sites. Also, reproductive success is generally higher in areas that were recently burned (Zimmerman 1992, Shriver and Vickery 2001, Delaney et al. 2002).

Although, I found vegetation variables important to nest site selection and nest success, sample sizes were small and pooled across years, and should not be extrapolated beyond the Waggoner Experimental Ranch. More research to increase sample size and the suite of vegetation parameters measured (e.g., litter depth, vegetation density) should

be conducted for a more conclusive assessment of nest-site selection and nest success related to post-burn years and to vegetative parameters. However, based on the results, I would recommend a rotational burning regime to provide habitat in various vegetative successional stages to maintain suitable and productive nesting habitat for the 5 grassland species.

## CHAPTER IV

### CONCLUSIONS

My data indicated brush cover, grass cover, and visual obstruction generally increased with post-fire succession and bare ground decreased with post-fire succession. Species richness, grasshopper sparrows, Cassin's sparrows, and dickcissels responded positively to post-fire succession and lark sparrows decreased with post-fire succession. All birds were present but reduced in numbers on the control sites. Results suggest that grassland birds were not randomly selecting breeding sites on the Kite Camp. However, each species had different vegetation characteristics that were significantly related to abundance.

During 2004–2005, 90 grassland bird nests were monitored. For all species except lark sparrows, greater visual obstruction and more grass or forb cover were selected for at nest sites. Probability of nest success increased as visual obstruction declined, bare ground cover declined, or grass cover declined. Bird abundance was most associated with different vegetation variables than nest-sites or the probability of nest success. However, all grassland species were present and nesting on all treatment sites, suggesting that brush encroached areas can still provide habitat for these birds. Prescribed fire may not completely restore shrub-invaded areas to true grasslands, but it can maintain local grassland areas in various vegetative successional stages which are beneficial for sustaining the local grassland bird populations.

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## APPENDIX A

Table A1. All bird species detected on transects during the breeding season at the Waggoner Experimental Ranch, Wilbarger County, Texas, 2002–2005.

Species	Scientific Name
Cassin's sparrow	<i>Aimophila cassinii</i>
Grasshopper sparrow	<i>Ammodramus savannarum</i>
Lark sparrow	<i>Chondestes grammacus</i>
Dickcissel	<i>Spiza americana</i>
Eastern meadowlark	<i>Sturnella magna</i>
Northern Bobwhite	<i>Colinus virginianus</i>
Burrowing Owl	<i>Athene cunicularia</i>
Scissor-tailed flycatcher	<i>Tyrannus forficatus</i>
Field Sparrow	<i>Spizella pusilla</i>
Brown-headed Cowbird	<i>Molothrus ater</i>
Mourning Dove	<i>Zenaida macroura</i>
Wild Turkey	<i>Meleagris gallopavo</i>
Eastern Bluebird	<i>Sialia sialis</i>
Barn Swallow	<i>Hirundo rustica</i>
Red-bellied Woodpecker	<i>Melanerpes erythrocephalus</i>
Golden-fronted Woodpecker	<i>Melanerpes aurifrons</i>
Ladder-backed Woodpecker	<i>Picoides scalaris</i>

Table A1Continued

Species	Scientific Name
Downy Woodpecker	<i>Picoides pubescens</i>
Hairy Woodpecker	<i>Picoides villosus</i>
Northern Flicker	<i>Colaptes auratus</i>
Ash-throated Flycatcher	<i>Myiarchus cinerascens</i>
Great Crested Flycatcher	<i>Myiarchus crinitus</i>
Loggerhead Shrike	<i>Lanius ludovicianus</i>
Northern Mockingbird	<i>Mimus polyglottos</i>
Northern Cardinal	<i>Cardinalis cardinalis</i>
Blue Jay	<i>Cyanocitta cristata</i>
Yellow-billed Cuckoo	<i>Coccyzus americanus</i>
Greater Roadrunner	<i>Geococcyx californianus</i>
Painted Bunting	<i>Passerina crisis</i>
Blue Grosbeak	<i>Guiraca caerulea</i>
Baltimore Oriole	<i>Icterus galbula</i>
Orchard Oriole	<i>Icterus spurius</i>
Bewick's Wren	<i>Thryomanes bewickii</i>
Black-crested Titmouse	<i>Parus bicolor</i>
Red-winged Blackbird	<i>Agelaius phoeniceus</i>
Canyon Towhee	<i>Pipilo fuscus</i>
Curve-billed Thrasher	<i>Toxostoma curvirostre</i>
Western kingbird	<i>Tyrannus verticalis</i>

Table A1 Continued

Species	Scientific Name
Eastern Kingbird	<i>Tyrannus tyrannus</i>
Chihuahuan Raven	<i>Corvus cryptoleucus</i>
Common Raven	<i>Corvus corax</i>
Common Grackle	<i>Quiscalus quiscalus</i>
Great-tailed Grackle	<i>Quiscalus mexicanus</i>
Common Nighthawks	<i>Chordeiles minor</i>
Killdeer	<i>Charadrius vociferus</i>
Mississippi Kite	<i>Ictinia mississippiensis</i>
Swainson's Hawk	<i>Buteo swainsoni</i>
Yellow-throated Vireo	<i>Vireo flavifrons</i>
Cattle Egret	<i>Bubulcus ibis</i>
Great Egret	<i>Ardea alba</i>
Snowy Egret	<i>Egretta thula</i>
Little Blue Heron	<i>Egretta caerulea</i>
Great Blue Heron	<i>Ardea herodias</i>
Yellow-crowned Night-Heron	<i>Nyctanassa violacea</i>

## APPENDIX B

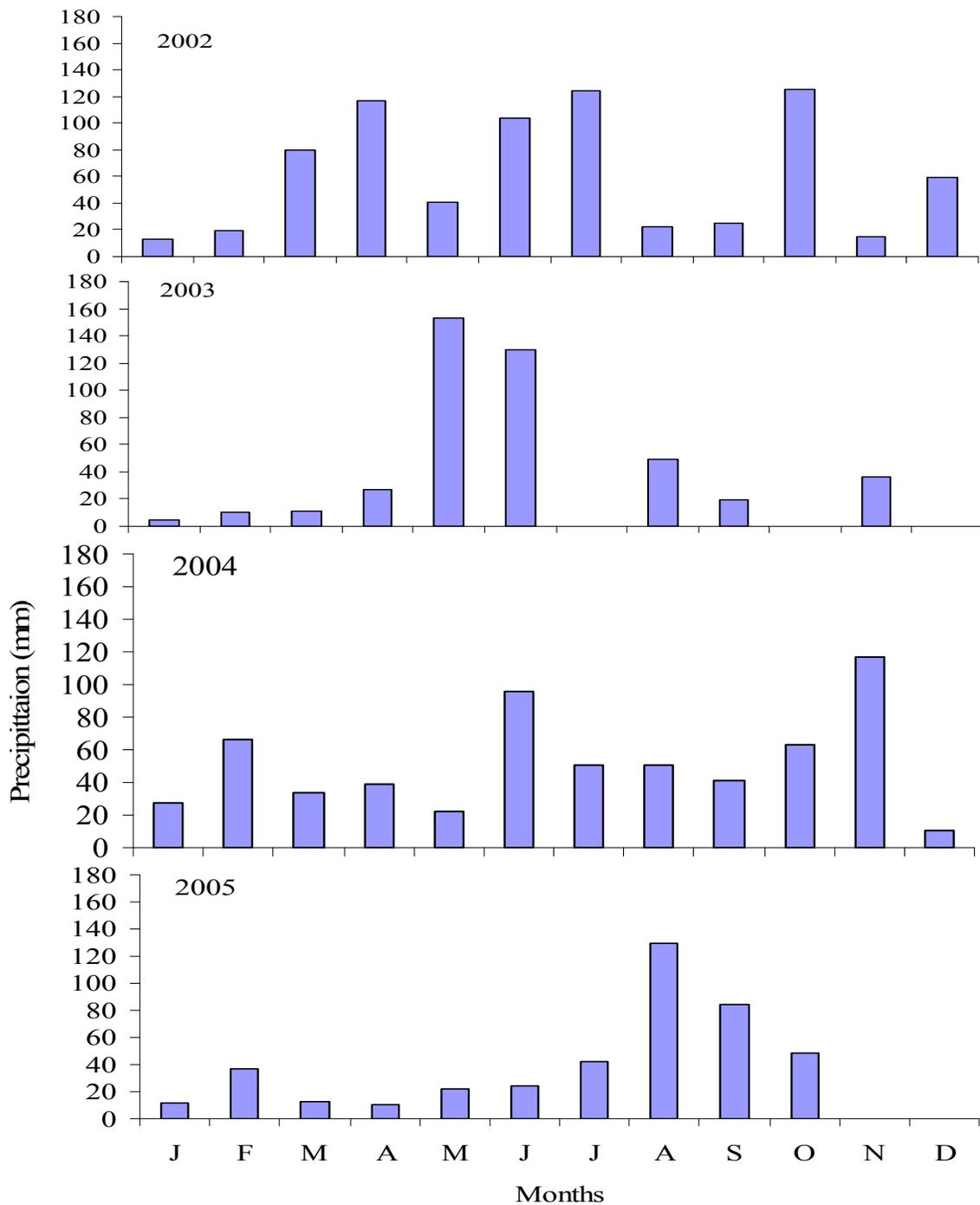


Figure B1. Monthly mean precipitation on the Kite Camp of the Waggoner Experimental Ranch, Wilbarger County, Texas, 2002–2005.

## VITA

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