

Assessing the Influence of Wind Turbines and Land-use  
on Mortality Risk of Avian Species on the  
Llano Estacado with Emphasis on Birds of Prey

by

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## CHAPTER I

### AVIAN POPULATIONS AND ECOLOGY IN RELATION TO WIND ENERGY DEVELOPMENT ON THE LLANO ESTACADO

Wind energy generation is a promising alternative energy industry with a large potential for continued development in West Texas due to the unique landscape and climate conditions. The Southern High Plains (hereafter SHP) of Texas experiences some of the state's greatest expanses of high winds (Combs 2008). Texas continues to possess the highest installed capacity of wind energy in the United States – more than double that of any other state at 12,753 megawatts. Texas is also the top state for wind projects under construction; more than 57% of reported wind capacity under construction in the United States is in Texas (AWEA 2014). Texas also boasts the highest bird diversity in the United States (Stein 2002). The SHP falls within the North American Central Flyway, where hundreds of species fly through during migration. West Texas historically possessed broad expanses of native grassland, and grassland birds as a group are experiencing long-term declines (Brennan and Kuvlesky 2005). Many grassland-dependent species occur in the studied region of the high plains (Seyffert 2001) for breeding, overwintering, or a migration stopover.

When avian habitat preferences overlap with landscapes that will facilitate the highest energy yield, wind turbines can affect populations directly and indirectly. The cumulative effect of wind energy development on avian populations over time may be of great consequence. Collisions with wind turbine rotor blades, towers, or associated structures can occur when birds fly close to the machinery. There are many sensory reasons that birds collide with conspicuous objects; birds have higher resolution lateral

vision than frontal vision, they often assign greater importance of lateral vision for foraging and predator avoidance instead of looking ahead into presumed open airspace, and many birds turn their head during flight for a better lateral or ground-view, which blinds them to what is ahead (Martin 2011). Ecological costs of wind farms on avian communities include direct mortality through collision with wind turbines; Erickson et al. (2001), in a nationwide meta-analysis, estimated that approximately 33,000 birds will be killed annually by wind turbines. Indirect ecological costs, which may be more important than direct costs, include habitat loss due to avoidance of areas with wind turbines (Kuvlesky et al. 2007, NRC 2007), or visual or sound impact (de Lucas et al. 2007). Grassland species, especially, are susceptible to prairie fragmentation by development, among other factors (Brennan and Kuvlesky 2005).

Sandia National Laboratories has joined Texas Tech University National Wind Institute in an agreement to conduct wind energy research at the Reese Technology Center (hereafter RTC) in Lubbock, Texas. The RTC is on the Llano Estacado, which is a high, flat mesa within the SHP (Leatherwood 2010). This wind energy research partnership is facilitating gains in knowledge regarding alternative and also allows for valuable ecological assessments such as effect of wind turbines and their construction on wildlife. The objectives of these entities are to initiate research on wind turbine-to-turbine interactions and to advance rotor technology. The construction of three new test turbines at the Scaled Wind Farm Technology facility began in November 2012. There are seven or more additional turbines planned for future construction (Slyker 2012). Two of these turbines are funded by the Department of Energy and Sandia National Laboratories, while the third turbine is funded by Vestas. I conducted a research program which

assessed three concerns associated with wind energy development and avian populations in regards to the Scaled Wind Farm Technology facility at the RTC. Avian research is pertinent in the Llano Estacado because in the wind energy field, avian abundance is a potential indicator for mortality risks from turbines. Risk management of potential avian collisions is a high priority for this industry.

My first objective was to assess how wind turbine construction and operation influence avian abundance and diversity at the SWiFT facility in Lubbock. The ecological cost of turbine construction and activity can be measured by assessing mortality of birds (i.e., collisions with rotor blades) or by assessing the avoidance of an area (i.e., presence or absence of a species post-wind energy development where it occupied an area pre-construction; de Lucas et al. 2007). While avian mortality surveys were planned, they were not initiated due to unforeseen circumstances surrounding turbine operation. Point-count distance sampling is a primary method to estimate the density or abundance of birds in an area (Buckland et al. 2001). I conducted two types of monthly point-count surveys to allow for a comparison of temporal patterns of species abundance and diversity, per season, before and after construction. One type, large-plot point counts, is often used to assess estimated bird use of wind resource areas (Strickland et al. 2011). The other type, smaller radius surveys for breeding birds, intensively survey the study area only during the breeding season.

Second, I explored how raptors associate with the local landscape and what factors can predict raptor abundance. Raptors typically have large home ranges and naturally occur in low densities, so among all bird species their assemblage experiences

proportionally greater effects of mortality or habitat loss. Additionally, in a meta-analysis of wind farm mortality reports by Erickson et al. (2001), diurnal raptors and owls comprised 43.4% of all avian collision fatalities. In an assessment of mortalities at a wind energy center in the Texas Panhandle, 47% of avian fatalities were raptors (Miller 2008). It is likely that birds of prey associate with certain landscape qualities more than others. By identifying these patterns and relating them to raptor density, I plan to provide the wind industry with information that can potentially minimize wind-wildlife conflict. Within the Llano Estacado, I assessed raptor association with different land-use categories and proximity to playas (regionally important intermittent wetlands) to predict where the highest densities of certain raptor species would occur. To accomplish this, I conducted road surveys that, in addition to the landscape features they explored, also assessed abundance and diversity of raptor species per season.

Third, I studied the nesting ecology of the American Kestrel (*Falco sparverius*) in Lubbock County, Texas, by expanding a nest-box monitoring program. The American Kestrel is a small, abundant falcon that is undergoing a range-wide population decline (Farmer and Smith 2009, Smallwood et al. 2009a, AKP 2012b) except in the SHP (AKP 2012a). Here they may face risks from increasing wind energy development, as they have a high propensity for turbine collisions compared to other avian species (Smallwood et al. 2009b). Thus, it is relevant to develop an understanding of their ecology in this region for conservation purposes. Installing nest-boxes can serve to bolster American Kestrel populations (Toland and Elder 1987) and is accepted as an effective platform for population monitoring (Katzner et al. 2005). The arid and windy climate on the Llano Estacado desiccates wooden boxes quickly; I explored other nest box materials for

durability and use by kestrels. Nest-box programs can provide insights into kestrel breeding ecology for long-term conservation and management of American Kestrels.

These three research directions are presented as discrete components of my research. The impact of wind energy facilities on bird populations may be reduced if adequate information is provided to wind energy during the planning and design process. This body of research may prove informative for wind energy planning and risk management by increasing local ecological knowledge and preventing avian mortalities.

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## CHAPTER II

### AVIFAUNAL COMPOSITION, DIVERSITY, AND DENSITY PRE- AND POST-WIND TURBINE CONSTRUCTION AT THE REESE TECHNOLOGY CENTER

#### INTRODUCTION

As the energy demands of the United States continue to increase, the importance of alternative energy development grows. Wind power has the potential to supply growing energy demands and has no fuel cost, emissions, or water use (Combs 2008). The state of Texas already has double the installed capacity of wind energy of any other state in the United States at 12,753 megawatts (AWEA 2014). Additionally, more than 57% of reported wind capacity under construction is in Texas (AWEA 2014). Texas also supports the highest bird diversity in the United States (Stein 2002). Within Texas, the Southern High Plains (SHP) region is characterized by broad expanses of high quality winds (Combs 2008) and has great potential for continued wind energy development.

Sandia National Laboratories and the Texas Tech National Wind Institute began construction of three test turbines at the Scaled Wind Farm Technology (SWiFT) facility at the Reese Technology Center (RTC) in November 2012, with up to seven additional turbines planned for future construction (Slyker 2012). The RTC lies within the Llano Estacado region of the SHP; the Llano Estacado is a caprock tableland covering northwestern Texas and northeastern New Mexico (Leatherwood 2010). Its ecological features and grasslands provide habitat for a variety of avian species. Numerous species that occur in the Llano Estacado and are found in the studied region are also listed in the United States Fish and Wildlife Service's (USFWS) "Birds of Conservation Concern" list for Bird Conservation Region 18, the Shortgrass Prairie (USFWS 2008; Appendix A).

Three of the species listed as "Species of Concern" by Texas Parks and Wildlife also occur on the study area (TPWD 2014; Appendix A). Grassland birds, as a group, have been experiencing long-term declines across the continent (Brennan and Kuvlesky 2005). Many grassland-dependent species occur in the shortgrass prairie on the Llano Estacado (Seyffert 2001). Numerous playas, or shallow ephemeral wetlands, are present on the Llano Estacado and provide food and cover resources for multitudes of migrating and overwintering waterfowl, wading birds, and shorebirds (Anderson and Smith 1999). Playas are also important breeding sites for certain shorebirds (Conway et al. 2005). Black-tailed prairie dog (*Cynomys ludovicianus*) colonies, also numerous in the Llano Estacado, have increased avian abundance in non-drought years compared to adjacent habitat (Barro et al. 1999), provide nest sites for Burrowing Owls (*Athene cunicularia*), and are a prey resource for Ferruginous Hawks (*Buteo regalis*; Cully 1991). Today, the land-use in the study area is a patchwork of agricultural activity, grazed lands, residential development, Conservation Reserve Program grassland, and native grassland remnants.

Potential ecological costs of wind farms on avian communities include mortality through collisions with rotor blades, a direct effect, or habitat loss or avoidance of areas with wind turbines, an indirect effect (National Research Council 2007). A summary report by Kuvlesky et al. (2007) states that on average, bird collisions range from zero to 30 collisions per turbine each year. In a recent study, Smallwood et al. (2010) used new methods to estimate avian fatalities from carcass searches and found that his methods resulted in fatality estimates almost 70% higher than reported, indicating that avian collision fatalities are possibly more numerous than previously perceived. Certain seasons, activities, avian abundance or behavior, weather conditions, and other factors

can increase a bird's risk of a turbine collision. First, migration season appears to increase avian collision rates: Johnson et al. (2002) found that of all individual avian mortalities, approximately 71% of collision victims were likely migrating through the area whereas only 20% were breeding species and 9% were permanent residents. The study area in the Llano Estacado lies within the North American Central Flyway - an avian migration route – which potentially poses a risk to migrating birds. Avian abundance may be related to turbine collision risk in high-density migration corridors or foraging areas (Drewitt and Langston 2006; but see de Lucas et al. 2008).

Among avian species, certain taxonomic groups are reported to have greater collision rates or risk of collisions. Raptors appear disproportionately susceptible to collisions with turbine rotor blades (Nelson and Curry 1995). Smallwood and Thelander (2005) explain that mortality estimates demonstrate that ongoing operations in California kill relatively large numbers of raptors and other birds. Some common raptors that nest on the RTC include the Burrowing Owl and the American Kestrel (*Falco sparverius*), both of which are known to experience mortality by colliding with wind turbine rotor blades (Smallwood and Thelander 2008). The Altamont Pass Wind Resource Area, an area with a high density of and high collisions of breeding Burrowing Owls, may be an example of an ecological sink for the species (Smallwood et al. 2007). Raptors have longer lifespans and lower reproductive potential, so collisions with turbines may have a more dramatic negative impact on raptor populations (Kuvlesky et al. 2007). Outside of California, approximately 82% of avian collision mortalities are passerines (Erickson et al. 2002). Geese, swans, and other large birds with poor agility in flight also experience higher risk of collisions (Brown et al. 1992).

Osborn et al. (1998) studied the avian flights within a rotor-swept zone and proximity of flights to turbines and determined that waterfowl and raptors had the greatest risk of colliding with turbines. Weather patterns also appear to impact on avian collision rates: Johnson et al. (2002) estimated that up to 93% of detected mortalities were associated with inclement weather like fog, gusty winds, and thunderstorms. Finally, avian behavior likely influences an individuals' risk of collision; foraging activities (e.g., diving for prey) or territorial defense possibly prevent birds from detecting moving rotor blades (Smallwood et al. 2009). Bird visual acuity is well-developed in their peripheral vision compared to their forward vision, which helps explain why they seemingly do not detect conspicuous structures in their pathway while they are foraging or interacting with conspecifics (Martin 2011).

For some bird species, the risk of habitat displacement is greater than the risk of collision with moving turbines. The infrastructure required to support an array of turbines including roads and transmission lines represents a potential threat to birds because of fragmentation and facilitation of invasion by exotics (Kuvlesky et al. 2007). Among other threats they face, grassland species are susceptible to prairie fragmentation (Brennan and Kuvlesky 2005), and wind energy development on prairie landscapes could further the problem. Grasslands without turbines supported higher densities of breeding birds than grasslands with turbines within 80 m of transects (Leddy et al. 1999). In a summary of turbine effects on groups of birds, Kuvlesky et al. (2007) reported that wind energy development could displace migrating or breeding waterfowl by disrupting their daily movements or altering their migration activity. Brennan et al. (2008) overview the ecology of many upland game birds in relation to wind energy development and conclude

that the habitat fragmentation and associated infrastructure will likely negatively affect many upland game bird species. In one study, Swainson's Hawks (*Buteo swainsonii*) entirely avoided suitable nesting habitat within a turbine area (Garvin et al. 2011), indicating that this species may avoid turbine activity.

The cumulative deaths caused by wind energy are much less than those caused by nuclear power stations or fossil fuel plants (Sovacool 2009). Even so, avian mortality attributable to wind power will likely have a cumulative impact on bird populations (Johnson et al. 2002) as opposed to driving bird declines. In cases of small populations or long-lived slow-reproducing species, high collision mortalities could have drastic population-level effects. To assess direct and indirect influences that wind turbines may have on a local avian community, I compared diversity and density metrics before and after turbine construction. The American Wind Energy Association's (AWEA) siting handbook (2008) recommends conducting point count surveys to assess avian use in order to provide information on species composition and to estimate abundance and composition of breeding birds in an area. To meet these criteria, I conducted fixed-point intensive surveys in the breeding season that sampled the breeding birds in the immediate turbine construction area, as well as large-plot broad scale avian surveys (Strickland et al. 2011) that sampled the landscape adjacent to and surrounding the turbine site. My objective was to assess how conditions pre- and post-turbine construction and operation at the SWiFT site influenced seasonal avian abundance and diversity. Surveys were consistent with the guidelines of the Avian and Bat Protection Plan for the Sandia National Laboratories/Texas Tech University Institute for Renewable Energy Wind Test Facility (Sandia National Laboratories 2011).

## METHODS

### *Study Area*

The Reese Technology Center (RTC) is a decommissioned United States Air Force base in Lubbock, Texas. The RTC constitutes the entire study area for breeding bird surveys and the center of the study area for the broadscale point counts. This site is western Lubbock County, Texas, on the Llano Estacado, which covers about 83,000 km<sup>2</sup>. The Llano Estacado is characterized by flat tablelands and is comprised of broad expanses of shortgrass prairie. The site is situated within the North American Central Flyway, a major migratory route. The timing of migration in this area depends largely on biology of different avian species, but generally spans late February to mid-April in spring and late August to early November in this region. Within the 1,012 hectares of the RTC is a 27 hectare plot in the northwest where the testing facility and new turbines are located. The three new test turbines are 225 KW with a tower height of 30 m, a rotor diameter of 27 m, and total height of 44 m (Sandia National Laboratories 2011). The turbines began construction in November 2012 and remain in a “testing” phase. Seven or more additional turbines may be erected on the study area (Slyker 2012). In addition to the three SWiFT turbines, Alstom constructed an industrial-scale turbine at the RTC in approximately February 2011 and Gamesa began construction on another industrial-scale turbine in approximately October 2013.

The average precipitation in the study site is 48.6 centimeters annually and temperatures range from an average winter low of -2.6 degrees Celsius to an average summer high of 32.4 degrees Celsius (NOAA 2014a). The RTC contains a remnant southern shortgrass prairie grassland community characterized by black-tailed prairie dog

colonies, playas, concrete runways, and construction areas. Black-tailed prairie dog colonies are abundant in the SHP and are often associated with playas (Pruett 2004). Nearly 20,000 playas, which are shallow ephemeral prairie wetlands, occur on the Llano Estacado (Bolen et al. 1989). Playas are ecological refuges for many species in the SHP (Bolen et al. 1989), especially amidst the largely agricultural landscape. Common plant species include buffalo grass (*Bouteloua dactyloides*) and blue grama (*Bouteloua gracilis*) intermixed with yucca (*Yucca* spp.) and catclaw (*Acacia greggii*). Surrounding the RTC, land-use practices are predominately agriculture, urban-developed, and herbaceous grassland (USDA 2009). The two main crops in the surrounding agriculture are cotton and grain sorghum.

### *Field Methods*

#### *Breeding bird surveys*

To determine composition, density, and diversity of the breeding bird community, I conducted 49 fixed-radius point count surveys (Fig. 2.1) twice per summer in May and June 2012 – 2014. The points were representative samples of the 7.7 km<sup>2</sup> grassland-runway area within the larger RTC complex where the turbines were constructed. To prevent bias, I spaced points at least 200 m apart to minimize double-counting of birds and did not place points directly on runways or within 75 m of field edges or vegetation changes (Savard and Hooper 1995) to avoid edge effects. I identified all birds detected within a 75 m radius of the point (per Roberts et al. 2012) within a ten-minute window. I estimated the distance from bird to observer at the time of the detection with a laser rangefinder or surveyor experience for subsequent analyses. As suggested by Ralph et al. (1993), I recorded mode of detection for all birds (seen or heard, stationary or flying

through) and also recorded birds flushed upon observer arrival to the survey point. I conducted all breeding bird surveys during the second half of the month between sunrise and three hours after sunrise. I also recorded wind and weather variables with a weather meter; if precipitation or wind conditions over 30 km/h occurred during scheduled surveys, I postponed the survey. It was not reasonable for the wind cutoff to be lower because high, sustained winds are common at the study site (e.g., monthly mean wind speeds of 20 km/h; NOAA 2014b).

*Broad scale point count surveys*

I also conducted large-plot point count surveys year-round, which are often used to assess estimated bird use of wind resource areas (Strickland et al. 2011). Observers surveyed nine 800 m radius general inventory points each month from September 2011 – August 2014. Three points were along the center airstrip at the RTC, one was at a permanent pond on the RTC, and five points surrounded the RTC and sampled areas north, south, east, and west (Fig. 2). I surveyed the five points outside of the RTC to effectively sample the surrounding landscape and thus, the local avian community potentially at risk from turbines each season. The distance between the northernmost point and southernmost point was approximately 9.2 km and between the westernmost point and easternmost point was approximately 6.9 km, covering an area of approximately 63.5 km<sup>2</sup>.

I recorded all birds detected within 20 minutes with the 800 m radius fixed-plot survey area. We also noted the mode of detection (seen, heard, or both; Ralph et al. 1993) and estimated distance from observer to bird with a laser rangefinder to include in density estimates. For birds in flight, we estimated the flight height using clinometers,

rangefinders, and surveyor experience. We gathered additional data on wind and weather variables with a weather meter. We did not conduct point count surveys in times of inclement weather (Ralph et al. 1993) or high winds greater than 30 km/h. We surveyed each point three times per month during the second half of the month. These visits included a morning, mid-afternoon, and evening survey each month to account for various activity levels of birds. Morning surveys began within 30 minutes of sunrise, mid-afternoon surveys evenly spanned the midpoint of the day between sunrise and sunset, and evening surveys terminated within 30 minutes before sunset.

### *Analysis*

In the study area of the Llano Estacado, the breeding season occurs from the end of April through July, fall migration typically occurs from late August to late October, winter residents are established by mid-November, and spring migration occurs from early March through late April. Because I conducted broad scale point counts in the middle of each month, I grouped months into the following biologically relevant seasons: fall – September and October, winter – November through February, spring – March and April, and summer – May through August. At least six survey sessions occurred per season (spring and fall) with a maximum of twelve (winter and summer). Broad scale surveys began in September 2011 but the data collection format was not finalized until December 2011 and one of the nine survey points was not added until May 2012. All surveys are included in reports for and species presence across months, but I only included the eight consistent survey point locations from December 2011 through August 2014 (33 months total) in the analysis of diversity and abundance.

I planned to use a Before-After/Control-Impact (BACI) design for this study (Anderson et al. 1999) to compare avian diversity and abundance data before and after wind turbine construction initiation. This design is commonly used and considered desirable for observational studies of impacts on a treatment site compared with a reference site (Morrison et al. 2008). However, the “impact area” was too small to compare to a control; only one point was within this “impact area.” Thus, I only compared before- and after-construction data for both breeding bird and broad scale surveys. I determined composition, density, and diversity of breeding avian species on the RTC from the small-plot surveys. I determined the same measures for the greater avian community from the large-plot broad-scale data each season.

I examined cumulative avian species richness using EstimateS (Colwell 2013) for both survey methods separately to assess whether the number of surveys adequately sampled the species richness; this can be determined if the estimator reaches an asymptote (Chao et al. 2009). The program fits multiple models to the data using the accumulation of new species per sampling session. For assessment of breeding species accumulation, I included all survey sessions over three breeding seasons (2012 – 2014). I used the mean species abundance for each month of surveys (33 total) for the broad scale community assessment. I used methods described in Magurran (1988) to calculate diversity indices including species richness (S), species evenness (J), Gini-Simpson’s (1-D) and Shannon-Wiener diversity ( $H'$ ) per year for breeding surveys, and per season for broad scale surveys. Each of these measures of diversity describes a different aspect of the community: The Gini-Simpson and Shannon-Wiener diversity indices each give

different weights to species evenness and richness, thus, if both are in agreement, you can make a stronger case with your results.

For breeding bird community assessment, surveys completed in 2012 represented “before construction,” while surveys completed in 2013 and 2014 represented “after construction.” For the broad scale avian community analysis, December 2011 to November 2012 were considered “before construction,” and all subsequent surveys were considered “after construction.” For broad scale survey data, I first standardized the species abundance values for seasons of different length by dividing the raw abundance counts by the number of months surveyed in that season. Even so, one must be cautious comparing diversity indices among seasons because more species accumulate over a longer time period and dividing the abundances by the number of months does not mediate accumulation of species. I pooled the broad scale monthly mean abundances into seasons, then divided the seasonal totals by the number of months to calculate the mean seasonal metrics. I compared diversity ( $H'$ ) across years for both community analyses using Hutcheson’s (1970) robust t-test in order to assess effects of wind turbine construction on avian diversity at the RTC. I performed the Hutcheson’s t-tests in Microsoft Excel. I defined a statistically significant difference at a level of  $\alpha = 0.05$  for all tests.

I used Program DISTANCE 6.2 (Thomas et al. 2010) to calculate species’ densities while incorporating imperfect observer detections using models (Buckland et al. 2001). An avian grassland study found strong observer and species-specific differences in detection rates at distances less than 50 m (Diefenbach et al. 2003); thus I did not assume

a detection probability of close to one for avian surveys despite our small breeding bird search radius in open grassland. To calculate the species' probability of detection and observers' effective detection radius, I assessed four model definitions (uniform cosine, half-normal cosine, half-normal hermite polynomial, and hazard-rate simple polynomial; as recommended by Thomas et al. 2010) for each species. For breeding bird surveys, I calculated each species' detection rates in each year to account for vegetation variation across years. For broad scale surveys, I calculated each species' detection rates in each season when they had adequate detections. I selected the model with the lowest AICc (Akaike's information criterion for small sample sizes; Burnham and Anderson 2002) value along with the best goodness of fit values and applied the appropriate model to each species to calculate density estimates.

I assessed breeding bird species density for five focal grassland species with approximately 30 detections or more in May and June 2012 through 2014: Cassin's Sparrow (*Peucaea cassinii*), Horned Lark (*Eremophila alpestris*), Mourning Dove (*Zenaida macroura*), Western Kingbird (*Tyrannus verticalis*), and Western Meadowlark (*Sturnella neglecta*). For Cassin's Sparrows, I only included singing individuals (representing males with breeding territories) in the analysis (as done by Roberts et al. 2012) because females and juveniles are secretive and difficult to detect. I divided the yearly density estimate by two to determine the mean monthly density because breeding bird surveys were conducted twice per year.

Of all species in each season of the broad scale surveys with enough detections (approximately 30 or more) to conduct analyses, I only analyzed density and abundance

differences for species associated with grasslands. This way, the analyses apply to native species in remnants of their native ecosystem in the area and not to introduced species or urban-associated species. I included both birds perched in and flying through the area to estimate species density. For this analysis, Western Meadowlarks (*Sturnella neglecta*), Eastern Meadowlarks (*Sturnella magna*), and Meadowlarks that weren't identified to species were grouped together due to difficulty of identification without hearing the song. Cackling Geese (*Branta hutchinsii*), Canada Geese (*Branta canadensis*), and unidentified geese were grouped together because of the difficulty in identifying them in flight. I used the same methods for Cassin's Sparrows as in the breeding bird surveys. The density estimate for Cassin's sparrows is representative of the density of singing males per season.

I then divided each month's raw detection counts by the probability of detection to achieve an estimated density per species per month (for breeding surveys) or per year (for broad scale surveys). Next, I calculated the average estimated abundance per species per year in the RTC for breeding birds and in the survey region for the broader avian community. For the broad scale surveys, I compared each species' mean estimated abundance across years using an ANOVA (Buckland et al. 2001) after testing for normality and equal distribution of variance using a Shapiro-Wilk's test (Shapiro and Wilk 1965) and a Bartlett test, respectively. If observations did not meet assumptions, I applied a log-transformation to the data before performing the test. If transformations were not successful to get the data to meet assumptions, I performed a Kruskal-Wallis rank sum nonparametric test. I used a Student's t-test to analyze fall seasons across years (only two years of fall data) and used ANOVAs for all other species and seasons

(Buckland et al. 2001). I defined a statistically significant difference at  $\alpha = 0.05$  in all tests. If there were significant differences among years from the ANOVA, I conducted a Tukey's Honest Significant Differences (Tukey HSD) test to detect where the differences lie. I used Program R (R Core Team 2013) to perform all statistical tests.

## RESULTS

### *Breeding bird surveys*

Observers detected a total of 2474 birds of 37 species over the 2012 - 2014 breeding seasons (Table 2.1; Table 2.2). In 2012 observers detected 620 birds of 26 species, in 2013 observers detected 669 birds of 25 species, and in 2014 observers detected 1185 birds of 25 species. Overall, the most abundant species relative to all detections were Mourning Doves (27.89%), Western Kingbirds (13.62%), Horned Larks (13.5%), Rock Doves (*Columba livia*; 11.52%), Western Meadowlarks (9.38%), and Cassin's Sparrows (7.28%; Table 2.1). Together, these six species made up 83.19% of all individuals detected. I conducted density analyses on all of the above birds except Rock Doves because they are often detection in large groups (meaning few detection events) and are not representative of the grassland ecosystem. The species accumulation curves generated by EstimateS (Colwell 2013; Fig. 2.3) did not yet reach an asymptote, indicating that additional survey sessions are needed in order to adequately sample the species richness in the region. The estimated cumulative abundance curves show that we detected between 77-88% of the species. This is based on comparisons between curves fit to estimated analytical species richness, the abundance coverage-based estimator of species richness (ACE mean), the Michaelis-Menten richness estimator (MM means), and the Cole rarefaction estimator.

Breeding season species richness only varied by one species over the three years (Table 2.2; Fig. 2.4). However, species evenness and diversity (both Shannon-Wiener and Gini-Simpson) decreased as years progressed (Table 2.2; Fig. 2.4). Results from Hutcheson's (1970) t-test across years shows a significant difference in  $H'$  between 2012 and 2014 ( $t=1.97$ ,  $p=0.04$ ; Table 2.3) as well as between 2013 and 2014 ( $t=3.368$ ,  $p=0.0008$ ; Table 2.3).

Analysis of observer effectiveness and detection probability demonstrated that for most species in most years, observers effectively surveyed the entire point count radius out to 75 m (Table 2.4). Mourning Doves were an exception to this general pattern in all years: the estimated probability of detecting individuals within the survey plot was always less than one (Table 2.4). The mean density per species varied per breeding season across the three years (Table 2.4; Fig. 2.5). I applied the probability of detection per species per year to the raw number of individuals detected each month in that year to attain a monthly abundance estimate (Table 2.5).

#### *Broad scale surveys*

Observers made 10,556 detections of 57,814 individual birds (Table 2.6). Of the individuals detected, observers identified 56,180 birds to species, for a total of 109 species detected over the study area. I compiled species detections per month from all surveys at all point count locations, and also divided all species detected into descriptive species groups to describe temporal patterns of occupancy as well as overall relative abundance from raw count data (Table 2.6). Rock Doves and Great-tailed Grackles (*Quiscalus mexicanus*) were the most abundant species (out of 20 species) in the human-associated group, together accounting for approximately 75% of the individuals in that

category (Table 2.6). The open country-associated species with the highest relative abundance out of the 24 species detected were Mourning Doves and Horned Larks, which made up approximately 58% of individual birds in that category (Table 2.6). Among raptors, Red-tailed Hawks (*Buteo jamaicensis*) and American Kestrels (*Falco sparverius*) were the two most abundant species at 21% and 19% relative abundance, respectively; I detected 13 birds of prey on the surveys (Table 2.6). Canada Geese (*Branta canadensis*) and Cackling Geese (*Branta hutchinsii*) dominated the waterfowl category; they comprised about 87% of the relative abundance out of 19 species (Table 2.6). Red-winged Blackbirds (*Agelaius phoeniceus*) and Sandhill Cranes (*Grus canadensis*) occurred in high numbers and together comprised about 72% of individual wetland and waterbird-associated species' detections (out of 24 species detected; Table 2.6). Nine additional migrant species detected were categorized as "other" (Table 2.6).

The cumulative species curve demonstrates that we have nearly reached an asymptote where new species are no longer detected at higher rates (Fig. 2.6); we have detected approximately 95% of species we would expect. Additional sampling sessions would be beneficial in order to adequately assess the avian community. We detected many more species when including the water site point (see Table 2.6) that are not represented in the agricultural-urban-grassland areas but are a part of the avian community.

I compared diversity metrics for each season across years (Table 2.7; Figs. 2.7.1-d). For winter, species richness increased and evenness decreased over the years, with Shannon's ( $H'$ ) and Simpsons diversity not in agreement as to the direction of diversity change (Table 2.7, Fig. 2.7.1). Results from Hutcheson's (1970) t-test showed a

statistically significant difference in  $H'$  among all years (Table 2.8). In the spring, there was a lower avian abundance overall and species richness increased greatly from spring 1 to spring 2. Evenness remained relatively consistent across all years, and Shannon's and Simpson's diversity increased from spring 1 to spring 2 (Table 2.7, Fig. 2.7.2). As with winter, results comparing  $H'$  across years were statistically significant among all years (Table 2.8). Summer diversity analyses yielded nearly equivalent evenness in years 1 and 2 that decreased slightly in year 3. Simpson's index mirrored evenness and decreased slightly in year 3.  $H'$  was the highest in year 2 and much lower in year 3 (Table 2.7, Fig. 2.7.3). Again,  $H'$  varied significantly across all years (Table 2.8). Finally, fall was only compared between year 1 and 2 with a t-test. The species abundance and species richness increased drastically from year 1 to year 2, whereas evenness and both diversity measures decreased (Table 2.7, Fig. 2.7.4). The t-test between years yielded statistically significant results (Table 2.8).

Of species with enough detections to be analyzed across multiple seasons, I conducted analyses for Meadowlark species across all seasons, Horned Larks across winter, spring, and summer, and Mourning Doves in the spring and summer. Northern Harriers (*Circus cyaneus*) and Ferruginous Hawks were only analyzed in the winter, and Cassin's Sparrows, Northern Bobwhite (*Colinus virginianus*), Swainson's Hawks, and Western Kingbirds were only analyzed in the summer. The probability an observer detected birds within the 800 m radius was, predictably, much lower than the detection probability of the 75 m radius breeding bird surveys. All raptors detected, overall, had higher detection probabilities and effective detection radii than the passerine species (Table 2.9). Northern Bobwhites also had relatively high probabilities of detection (0.22 -

0.33), corresponding with effective search radii of over 300 m (Table 2.9). In the summer, Cassin's Sparrows, Horned Larks, and Meadowlark species had relatively high rates of detection compared to other passerines, likely because males make themselves known through song or flight to attract the attention of a mate during the breeding season (Table 2.9). For Horned Larks and Meadowlarks species, these detection probabilities were higher in summer than in all other seasons detected (Table 2.9).

Like in the breeding bird surveys, Mourning Doves showed a drastic increase in density from an estimated 283 doves/km<sup>2</sup> in 2012 to 832 doves/km<sup>2</sup>. Estimated densities of certain species assessed over many seasons indicate great differences; Horned Larks have the highest estimated density in the winter with a lower density in the summer and the lowest in the spring (Table 2.9). Meadowlark species have the highest density in the winter and summer with much lower densities in the fall and spring (Table 2.9.) Based on the probability of detection for each species in a season, I calculated the monthly estimated abundance for each species based on the raw counts from surveys (Table 2.10). I utilized these abundances in species-specific statistical tests comparing density per season across years. Eleven of the 15 ANOVAs comparing grassland-associated species seasonal abundance across years yielded statistically significant results at the  $\alpha = 0.05$  level (Table 2.11). In particular, all raptor species showed significant differences in estimated abundance across years within their season of occurrence (Table 2.11). The post-hoc Tukey HSD test demonstrated among which years the significant differences lie (Table 2.12). Only three species analyzed showed a significant difference between years 1 and 2, or the pre-construction year and the first year post- construction initiation. These species were the Northern Harrier in the winter, Meadowlark species in the spring, and

Cassin's Sparrows in the summer (Table 2.12). Five species demonstrated a significant difference in abundance estimates across years 1 and 3: Meadowlark species and Northern Harriers in the winter and Mourning Doves, Swainson's Hawks, and Western Kingbirds in the summer (Table 2.12).

## DISCUSSION

Based on the results from relative abundance, diversity, density, and detection probability estimates, I conclude that relative raw abundance and diversity metrics per season provided valuable information for the before-after study. The small-plot breeding bird surveys also generated reliable results to compare breeding seasons across years. Issues such as too large of a sample size (an unanticipated problem) for the design of the Hutcheson's t-test and trouble fitting mathematical models to bird observation patterns may affect the interpretation of statistical tests comparing diversity and abundance across seasons from broad scale survey data.

I believe the decline in species evenness and diversity in the final year of the breeding season can be mostly attributed to the dramatic increase in the Mourning Dove population in the summer of 2014. A large increase in abundance of one species would lower the evenness and diversity values for that season. Interestingly, the 2012 breeding season experienced extreme drought yet had the highest diversity and evenness of all the years. Conversely, the Mourning Dove dominated the species relative abundance in 2014, which had the highest precipitation of the three years. It is possible that unfavorable conditions within a landscape actually favor the native species adapted to those conditions, resulting in higher evenness in those times. Or, it may be that the weather conditions in 2012 caused some species with irruptive population cycles to have a low-

abundance phase in that time. These environmental conditions make it difficult to assign the cause of the decline in diversity across seasons, though it appears that weather conditions certainly also influenced breeding bird density and composition.

I suspect that the extremely small p-values resulting from the Hutcheson's analysis of  $H'$  across seasons was influenced by the large sample size. Hutcheson's (1970) t-test comparing  $H'$  was developed before high-powered computers and he did not expect users to do calculations on sample sizes greater than approximately 200. Because variance tends to decrease as sample size increases, the variances for each of my seasonal samples were extremely small. The effect of extremely low variance in an ANOVA is to magnify small differences among samples. I recommend that a resampling technique be applied to future analyses of avian counts with large sample sizes. An additional problem with this survey design is that there are few replicates within each season. While I believe that differences do exist between some of the samples, it is impossible to distinguish a real effect from an exaggerated effect across years from this analysis. I recommend using an alternate method to compare diversity among seasons when sample sizes are high but there are few replicates among groups tested.

Calculating searcher efficiency appeared to effectively estimate actual species abundance for the breeding season surveys. However, it is difficult to assess where the meaningful significant differences are across years when viewing results from the broad scale ANOVA's (Table 2.11). Regardless of the nuances of the statistical test results, I can provide valuable information to the SWiFT project on seasonal species' occurrence (Table 2.6) and relative abundance. In fact, of the ten species of concern listed in the region and the state of Texas, observers detected nine in the broad scale surveys and I

incidentally detected the final species in the survey area, implying a continued need for monitoring and vigilance on the part of the wind energy developers to prevent mortalities of species of concern. In general, this study determined that various raptor species are abundant in the Llano Estacado in different seasons, migrating waterfowl overwinter at the RTC and surrounding landscape in high densities, and grassland-obligate species (mainly passerines) are abundant year-round. Each of these species groups is at risk of negative effects from turbines.

A number of other studies have assessed the collision risk of or impacts of turbines on avian grassland species. Piorkowski (2001) conducted fatality searches in mixed-grass prairie in Oklahoma and Miller (2008) conducted post-construction carcass searches along the caprock escarpment in the Texas Panhandle. Piorkowski (2001) detected only six carcasses across two summers of carcass searches; the avian species detected include a Turkey Vulture, a Northern Bobwhite, a Mourning Dove, a Grasshopper Sparrow, a Lark Sparrow (*Chondestes grammacus*), and an Eastern Meadowlark. Miller's study also detected few carcasses over her one-year search period. Fatalities in her study included Turkey Vultures (36%), Northern Bobwhites (20%), Greater Roadrunners (*Geococcyx californianus* 20%), and one detection each of a Mourning Dove, a Red-tailed Hawk (*Buteo jamaicensis*), a Blue Jay (*Cyanocitta cristata*), a Northern Mockingbird (*Mimus polyglottis*), a Great Horned Owl (*Bubo virginianus*), and an unidentified bird. Miller (2008) found that mortality peaked in the fall. Observers in the RTC study detected every species identified in the fatality collision searches above either in point count surveys or incidental to surveys. Wulff (2010) conducted pre-wind construction avian surveys in the Texas panhandle. She found that

the mean average flight height of 29 species was within the rotor-swept zone and 14 of these species had over 25% of their detected flight heights within the rotor-swept zone. Twenty-one percent of these species were raptors (Turkey Vultures, Swainson's Hawks, and Bald Eagles [*Haliaeetus leucocephalus*]), 50% were waterbird species (Canada Goose [*Branta Canadensis*], Greater White-fronted Goose [*Anser albifrons*], Mallard [*Anas platyrhynchos*], Northern Pintail [*Anas acuta*], Northern Shoveler [*Anas clypeata*], Sandhill Crane [*Grus Canadensis*], and Snow Goose [*Chen caerulescens*]), and 29% were passerines or "other birds" (Common Grackle [*Quiscalus quiscula*], Great-tailed Grackle [*Quiscalus mexicanus*], Mourning Dove, and Red-winged Blackbird [*Agelaius phoenicius*]). All of these species have also been detected at the RTC study site; I also believe raptors and waterfowl have a higher risk of collision compared to other species groups at the study site. Leddy et al. (1999) in Minnesota found that the total bird density was lower in grasslands without turbines compared to sites with turbines and that avian density decreased as distance to the turbine decreased. Piorkowski's (2001) Oklahoma study did not find such clear results, as only four of 27 species assessed showed an avoidance effect of the turbines. Of these four species, only the Western Meadowlark was a grassland-obligate species. Three other open country species showed no effect (Cassin's Sparrows, Horned Larks, and Killdeer [*Charadrius vociferus*]) and six other species showed increased density underneath the turbine.

I did not detect decreases in breeding bird densities from my breeding bird surveys after the turbine construction occurred, but I did not assess localized density with proximity to certain turbines. I recommend that future studies conduct a study of this nature to assess if birds avoid turbines on a smaller scale than this avian survey assessed.

In addition, I recommend that large-plot broad scale survey radii be reduced to approximately 400 m for raptor detections and other large-bodied birds, and reducing the survey radius for passerines and smaller birds to 200 m. This may prevent attaining miniscule detection probabilities as was found in this study.

The wide diversity of avifauna at the RTC in the Llano Estacado appear to be faced with the greatest turbine collision risks depending on species' groups. Relative abundance values indicate that waterfowl are in great abundance during the winter at the site and may be presented with the greatest collision risk then. Raptors are another relatively abundant group (after considering that they occupy areas at relatively low densities) on the RTC, and are probably susceptible to collisions because of their migration patterns and possible associations with playas and prairie dog downs (which are both present at the RTC). Many of the grassland-obligate species persist at the study area year-round with increased density in winter and summer. I detected temporal differences in abundance of many of these passerine species which may indicate a higher risk to passerines in the high-abundance seasons. My conclusions of species groups at greatest risk of turbine impacts are consistent with those of Miller (2008) and Wulff (2010)'s assessments in the Texas Panhandle.

Although wind energy development and production presents the risk of habitat loss and fatality to birds, cumulative deaths from this source must be kept in perspective. Sovacool (2009) estimates that in 2006, wind farms killed approximately 7,000 birds, nuclear power stations killed about 327,000, and fossil-fueled power plants killed around 14.5 million. Many mitigation measures can be employed by wind farms. For example,

management plans can propose options to mitigate fatality impacts including adjusting the operation of their turbines during migration events or specific times of the day or year (Sandia National Laboratories 2011).

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APPENDIX. LIST OF BIRDS OF CONSERVATION CONCERN DETECTED IN THE STUDY AREA

<u>Species</u>	<u>List (source)</u>
Swainson's Hawk	National List (USFWS 2008)
Ferruginous Hawk	Federal species of concern (TPWD 2014)
Peregrine Falcon	National List (USFWS 2008); Federal species of concern (TPWD 2014)
Prairie Falcon	BCR Region 18 list (USFWS 2008)
Solitary Sandpiper	National List (USFWS 2008)
Upland Sandpiper	BCR Region 18 list (USFWS 2008)
Lesser Yellowlegs	National List (USFWS 2008)
Burrowing Owl	BCR Region 18 list (USFWS 2008); Federal species of concern (TPWD 2014)
Loggerhead Shrike	National List (USFWS 2008)
Wood Thrush	National List (USFWS 2008)

Table 2.1. Species detections from pooled May and June breeding bird surveys per year, and species relative abundance from raw count data.

Species	2012	2013	2014	Total	Rel. Abund.
American Avocet	0	0	1	1	0.04%
American Kestrel	2	1	3	6	0.24%
Ash-throated Flycatcher	1	0	0	1	0.04%
Barn Swallow	18	1	10	29	1.17%
Bullock's Oriole	0	0	1	1	0.04%
Burrowing Owl	0	4	3	7	0.28%
Cassin's Sparrow	47	74	59	180	7.28%
Chimney Swift	1	1	0	2	0.08%
Chipping Sparrow	1	0	0	1	0.04%
Cliff Swallow	0	8	16	24	0.97%
Common Grackle	1	0	0	1	0.04%
Eurasian Collared-dove	0	0	8	8	0.32%
European Starling	0	1	3	4	0.16%
Grasshopper Sparrow	12	0	1	13	0.53%
Great Blue Heron	1	0	0	1	0.04%
Great-tailed Grackle	31	18	56	105	4.24%
Horned Lark	84	172	78	334	13.50%
House Finch	9	2	6	17	0.69%
House Sparrow	0	0	3	3	0.12%
Killdeer	2	0	8	10	0.40%
Lark Bunting	25	33	0	58	2.34%
Lark Sparrow	1	0	0	1	0.04%
Mallard	0	7	18	25	1.01%
Mourning Dove	142	54	494	690	27.89%
Northern Bobwhite	3	3	15	21	0.85%
Northern Mockingbird	2	3	1	6	0.24%
Rock Dove	50	9	226	285	11.52%
Red-winged Blackbird	10	9	0	19	0.77%
Ring-necked Pheasant	3	5	3	11	0.44%
Scaled Quail	0	1	0	1	0.04%
Scissor-tailed Flycatcher	8	4	4	16	0.65%
Swainson's Hawk	1	7	6	14	0.57%
Turkey Vulture	1	0	0	1	0.04%
Western Kingbird	89	149	99	337	13.62%
Western Meadowlark	75	94	63	232	9.38%
White-winged Dove	0	5	0	5	0.20%
Yellow-rumped Warbler	0	4	0	4	0.16%
Yearly total	620	669	1185	2474	

Table 2.2. Diversity indices from breeding bird surveys 2012 - 2014 including species richness (S), evenness (J), Shannon-Wiener diversity index (H'), and Gini-Simpson's diversity index (1-D).

Year	n	S	J	H'	var H'	1-D
2012	620	26	0.729	2.374	0.0098	0.874
2013	669	25	0.682	2.196	0.0083	0.842
2014	1185	25	0.613	1.974	0.0043	0.771
Total	2474	37				

Table 2.3. Results of Hutcheson's t-test comparing H' across breeding bird survey years. Statistical significance was set at  $\alpha=0.05$ .

Comparison	df	<i>t</i>	P
2012-2013	1271	1.323	0.186
2013-2014	1338	1.974	0.049
2012-2014	1167	3.368	0.0008

Table 2.4. Mean density estimates from breeding bird surveys within the RTC. Observations were pooled each year to estimate the probability of detection which was used to calculate mean density. Four-letter species codes can be found in Pyle and DeSante (2012).

Year	Name	No. detections	Probability of detection	Effective detection radius	Mean density/km <sup>2</sup>	Density lower CI	Density upper CI	Density CV
1	CASP 2012	32	1.00	75.0	4.1	2.5	6.6	24.5
2	CASP 2013	55	1.00	75.0	6.6	4.4	9.8	20.6
3	CASP 2014	46	0.60	58.1	9.9	5.2	18.7	32.9
1	HOLA 2012	67	1.00	75.0	10.7	7.9	14.4	15.3
2	HOLA 2013	130	0.60	58.2	27.7	19.5	39.5	18.2
3	HOLA 2014	60	1.00	75.0	8.2	5.8	11.7	17.8
1	MEAD 2012	69	1.00	75.0	9.6	7.2	12.7	14.6
2	MEAD 2013	92	1.00	75.0	9.5	7.4	12.2	13.0
3	MEAD 2014	59	1.00	75.0	6.7	4.7	9.4	17.8
1	MODO 2012	89	0.21	34.4	75.5	41.3	137.9	31.2
2	MODO 2013	38	0.56	56.0	10.2	5.6	18.3	30.4
3	MODO 2014	307	0.49	52.7	101.9	71.8	144.5	17.9
1	WEKI 2012	77	1.00	75.0	11.3	8.3	15.5	15.9
2	WEKI 2013	117	1.00	75.0	15.1	11.6	19.5	13.2
3	WEKI 2014	81	1.00	75.0	10.5	7.9	13.8	14.2

\*CASP densities represent densities of singing males

Table 2.5. Conversion of raw breeding bird detections to estimated abundance within the survey plot area. I divided the number of individuals detected by the probability of detection (P) to achieve the estimated actual abundance. Four-letter species codes can be found in Pyle and DeSante (2012).

Species	Month	2012			2013			2014		
		no. indivs. detected	P	Estim. abund.	no. indivs. detected	P	Estim. abund.	no. indivs. detected	P	Estim. abund.
CASP	May	28	1.00	28.0	21	1.00	21.0	17	0.60	28.3
	June	4	1.00	4.0	44	1.00	44.0	36	0.60	60.0
HOLA	May	39	1.00	39.0	72	0.60	120.0	47	1.00	47.0
	June	45	1.00	45.0	99	0.60	165.0	23	1.00	23.0
MODO	May	58	0.21	276.2	16	0.56	28.6	158	0.49	322.4
	June	84	0.21	400.0	38	0.56	67.9	336	0.49	685.7
WEKI	May	25	1.00	25.0	78	1.00	78.0	65	1.00	65.0
	June	64	1.00	64.0	71	1.00	71.0	34	1.00	34.0
WEME	May	45	1.00	45.0	46	1.00	46.0	36	1.00	36.0
	June	30	1.00	30.0	48	1.00	48.0	24	1.00	24.0

\*CASP densities represent densities of singing males

Table 2.6. Avian species observed during broadscale point count surveys from September 2011 – August 2014. Percent composition per species group was calculated using the number of individuals detected. Four-letter species codes can be found in Pyle and DeSante (2012).

Category, description	Months of occurrence												No. Individuals.	No. detections	Composition/ species group
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec			
<i>Human-associated</i>	<i>Species typically associated with human edifices/ disturbance, at least regionally</i>														
AMRO			x	x	x	x	x	x		x		x	116	46	0.7%
BARS				x	x	x	x	x	x	x			490	171	2.9%
BHCO	x						x	x				x	20	11	0.1%
BLJA	x				x	x		x	x	x	x		17	16	0.1%
BRBL				x		x						x	7	3	0.0%
CHSP					x	x			x	x			35	7	0.2%
CHSW							x	x	x				37	16	0.2%
CLSW					x	x	x	x	x				19	12	0.1%
COGR							x		x		x	x	425	6	2.5%
DEJU	x		x	x									26	3	0.2%
ECDO	x		x	x	x	x	x	x	x	x	x	x	937	406	5.5%
EUST	x		x	x	x	x	x	x	x	x	x	x	754	88	4.4%
GTGR	x		x	x	x	x	x	x	x	x	x	x	6005	862	35.0%
HOFI	x		x	x	x	x	x	x	x	x	x	x	337	118	2.0%
HOSP	x		x	x	x	x	x	x	x	x	x	x	568	118	3.3%
NOMO	x		x		x	x	x	x	x				57	50	0.3%
PUMA								x					12	4	0.1%
RODO	x		x	x	x	x	x	x	x	x	x	x	6844	752	39.9%
WCSP			x	x	x							x	84	15	0.5%
WWDO	x		x	x	x	x	x	x	x	x		x	355	55	2.1%
Unk swallow													1	1	0.0%
Unk swift													2	1	0.0%

Table 2.6, Continued

Category, description	Species	Months of occurrence												No. Indivs.	No. detections	Composition/ species group
		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec			
														17148		
<i>Open Country</i>																
																<i>Native grassland/brushland species</i>
	BRTH									x				1	1	0.0%
	BUOR								x					6	1	0.0%
	BUOW				x	x	x	x	x	x	x	x	x	122	108	1.0%
	CASP					x	x	x	x					200	194	1.6%
	CBTH					x		x						2	2	0.0%
	CCSP					x				x				7	4	0.1%
	CHRA					x	x				x	x	x	584	33	4.7%
	EAME										x		x	2	2	0.0%
	GRSP						x			x				2	2	0.0%
	HOLA	x		x	x	x	x	x	x	x	x	x	x	2725	788	22.1%
	LARB	x		x	x	x			x	x	x	x	x	1406	59	11.4%
	LARS					x	x	x	x					73	17	0.6%
	LOSH								x	x			x	6	5	0.0%
	MODO	x		x	x	x	x	x	x	x	x	x	x	4388	1743	35.6%
	NOBO					x	x	x	x	x				129	125	1.0%
	NOFL	x				x					x	x	x	16	15	0.1%
	RNPH	x				x	x	x	x		x			18	18	0.1%
	SATH					x	x							2	2	0.0%
	SAVS	x		x	x					x	x		x	68	31	0.6%
	SCQU						x	x	x					14	12	0.1%
	STFL						x	x		x				11	10	0.1%
	VESP	x					x			x	x	x	x	38	27	0.3%
	WEKI					x	x	x	x	x				1272	854	10.3%
	WEME	x		x	x	x	x	x	x	x	x	x	x	1027	904	8.3%

Table 2.6, Continued

<i>Category, description</i>	Species	Months of occurrence												No. Indivs.	No. detections	Composition/ species group
		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec			
Unk meadowlark														201	200	1.6%
														12320		
<i>Raptors</i>	<i>All birds of prey</i>															
	AMKE	x	x	x	x	x	x	x	x	x	x	x	x	220	211	19.4%
	COHA											x		3	4	0.3%
	FEHA	x	x	x	x					x	x	x	x	102	106	9.0%
	HASH									x				1	1	0.1%
	MERL											x		2	2	0.2%
	MIKI					x	x	x	x					119	79	10.5%
	NOHA	x	x	x	x				x	x	x	x	x	140	139	12.3%
	PRFA	x	x	x							x	x		10	10	0.9%
	RLHA			x									x	3	3	0.3%
	RTHA	x	x	x	x	x	x		x	x	x	x	x	236	222	20.8%
	SWHA				x	x	x	x	x	x				139	111	12.3%
	TUVU				x	x	x	x	x	x				86	63	7.6%
	WTKI												x	1	1	0.1%
	Unk buteo													47	47	4.1%
	Unk raptor													25	25	2.2%
														1134		
<i>Waterfowl</i>	<i>Geese and ducks</i>															
	AMCO					x				x	x			18	10	0.1%
	AMWI	x	x	x	x	x					x	x	x	112	17	0.5%
	BUFF											x		5	4	0.0%
	BWTE				x	x	x				x	x		268	35	1.2%
	CACG	x	x						x				x	2459	12	10.9%
	CAGO	x	x						x		x	x	x	17055	161	75.9%

Table 2.6, Continued

<i>Category, description</i>	Species	Months of occurrence												No. Indivs.	No. detections	Composition/ species group	
		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec				
	CANV		x										x		3	3	0.0%
	GADW				x	x	x							x	13	7	0.1%
	GWFG													x	1	1	0.0%
	GWTE	x	x	x	x	x							x	x	112	21	0.5%
	LESC	x	x	x									x	x	121	16	0.5%
	MALL	x	x	x	x	x	x	x	x	x	x	x	x	x	1449	153	6.5%
	NOPI		x	x		x							x	x	14	8	0.1%
	NSHO		x	x	x	x				x			x	x	268	26	1.2%
	REDH		x			x				x	x		x	x	37	8	0.2%
	RNDU	x	x	x	x	x							x	x	267	27	1.2%
	RUDU	x	x	x	x	x	x						x	x	122	24	0.5%
	SNGO	x	x										x	x	28	6	0.1%
	WODU									x					3	3	0.0%
	Unk duck														106	8	0.5%
															22461		
<i>Wetland/ Shorebirds</i>	<i>Species associated with wetlands and shoreline areas</i>																
	AMAV				x	x	x	x	x				x		68	32	2.0%
	BCNH				x	x	x	x	x						253	27	7.3%
	BEKI					x									1	1	0.0%
	BNST				x	x	x						x		59	36	1.7%
	CAEG												x		4	1	0.1%
	COYE						x								1	1	0.0%
	EAGR				x										2	2	0.1%
	GBHE												x		1	1	0.0%
	GREG												x		2	1	0.1%
	GRYE				x										1	1	0.0%

Table 2.6, Continued

Category, description Species	Months of occurrence												No. Indivs.	No. detections	Composition/ species group
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec			
KILL		x	x	x	x	x	x	x	x	x	x	x	327	187	9.4%
LBDO				x	x								21	4	0.6%
LEYE				x				x					6	4	0.2%
PBGR									x	x			11	8	0.3%
PESA					x								1	1	0.0%
RWBL	x	x	x	x	x	x	x	x	x	x	x	x	1593	177	45.8%
SACR	x	x	x							x	x		898	38	25.8%
SPSA				x	x			x	x				42	27	1.2%
UPSA								x					5	3	0.1%
WFIB				x				x					8	3	0.2%
WILL								x					2	2	0.1%
WIPH				x	x								124	5	3.6%
WISN			x									x	2	2	0.1%
YHBL				x					x				16	4	0.5%
Unk egret													13	1	0.4%
Unk plover													2	1	0.1%
Unk sandpiper													11	4	0.3%
Unk shorebird													7	3	0.2%
													3481		
<i>Other</i>	<i>Species considered as vagrant, pass-through migrants, and unidentified species without an associated group</i>														
EAPH									x				1	1	0.1%
FISP												x	1	1	0.1%
LBWO						x							1	1	0.1%
MOBL		x	x							x			17	3	1.3%
SOSP			x	x					x			x	8	3	0.6%



Table 2.7. Diversity indices across seasons from winter 2011 surveys through fall 2014 surveys including species richness (S), evenness (J), Shannon-Wiener diversity index (H'), and Gini-Simpson's diversity index (1-D). Standardized n was calculated by dividing n by the number per season.

Season	n	standardized n	S	J	H'	var H'	1-D
Year 1							
Winter	2701	900.3	23	0.603	1.89	0.00000619	0.772
Spring	813	406.5	25	0.648	2.08	0.0000338	0.822
Summer	1789	447.3	34	0.700	2.47	0.0000372	0.872
Fall	689	344.5	21	0.801	2.44	0.00012	0.890
Year 2							
Winter	7319	1829.8	25	0.509	1.25	0.00000125	0.705
Spring	1108	554.0	35	0.697	2.48	0.0000243	0.870
Summer	2475	618.8	41	0.701	2.60	0.0000213	0.888
Fall	2876	1438.0	30	0.653	2.22	0.00000293	0.847
Year 3							
Winter	16589	4147.3	31	0.400	1.37	0.000000222	0.577
Spring	2172	1086.0	33	0.686	2.40	0.00000578	0.876
Summer	4255	1063.8	35	0.613	2.18	0.00000561	0.815
Total	42786		71				

Table 2.8. Results from Hutcheson's t-test comparing H' each season across years. Statistical significance was set at  $\alpha=0.05$ . Winter includes December – February, spring includes March and April, summer includes May, June, July, and August, and fall includes September and October. November was excluded from winter analyses because the first November survey in year one occurred before methods were standardized.

Seasons Compared	df	<i>t</i>	P
<b>Winter</b>			
1-2	1830	93.8	<0.0001
2-3	4131	220.0	<0.0001
1-3	4131	204.6	<0.0001
<b>Spring</b>			
1-2	871	51.7	<0.0001
2-3	825	14.6	<0.0001
1-3	551	49.9	<0.0001
<b>Summer</b>			
1-2	892	17.7	<0.0001
2-3	951	82.2	<0.0001
1-3	586	44.3	<0.0001
<b>Fall</b>			
1-2	245	19.5	<0.0001

Table 2.9. Mean density estimates from broadscale point count surveys. Observations were pooled within seasons to estimate the probability of detection, but the density estimate is an average of the pooled months' means. Four-letter species codes can be found in Pyle and DeSante (2012).

Season	Name	No. indivs.	Probability of detection	Effective detection radius (m)	Mean density/ km <sup>2</sup>	Density lower CI	Density upper CI	Density CV
Winter	FEHA winter 1	18	1.00	620	1.89	1.08	3.29	9.60
	FEHA winter 2	12	1.00	500	1.76	0.89	3.49	11.93
	FEHA winter 3	43	0.31	441	6.25	3.19	12.27	11.75
	HOLA winter 1	23	0.01	61	1219.17	442.13	3361.87	18.10
	HOLA winter 2	115	0.03	95	4247.88	2588.00	6972.38	8.54
	HOLA winter 3	155	0.06	95	2382.74	1486.00	3820.62	8.13
	MEAD winter 1	50	0.07	147	112.50	60.38	209.64	10.78
	MEAD winter 2	57	0.09	139	126.41	75.51	211.62	8.88
	MEAD winter 3	147	0.04	139	444.12	265.03	744.23	8.91
	NOHA winter 1	18	0.49	491	3.01	1.39	6.54	13.48
	NOHA winter 2	30	0.06	183	29.98	10.80	83.25	17.90
	NOHA winter 3	37	0.33	437	5.52	3.15	9.67	9.64
Spring	HOLA spring 1	41	0.08	101	8.74	4.82	15.87	30.94
	HOLA spring 2	36	0.02	90	10.99	5.65	21.38	34.60
	HOLA spring 3	52	0.13	104	9.30	5.25	16.47	29.64
	MEAD spring 1	96	0.01	58	44.77	6.58	304.57	124.15
	MEAD spring 2	77	0.07	160	5.31	3.41	8.27	22.77
	MEAD spring 3	103	0.04	116	11.46	6.92	18.95	26.02
	MODO spring 1	137	0.03	145	23.04	12.52	42.39	31.77
	MODO spring 2	36	0.04	120	6.04	2.04	17.88	58.58
	MODO spring 3	137	0.03	145	23.04	12.52	42.39	31.77
Summer	CASP summer 1	35	0.43	207	12.98	7.28	23.12	7.47
	CASP summer 2	60	0.13	255	11.97	7.22	19.82	6.52
	CASP summer 3	72	0.27	288	11.98	6.95	20.64	7.06
	HOLA summer 1	56	0.19	100	107.54	66.08	175.02	6.29
	HOLA summer 2	88	0.13	92	183.30	115.60	290.67	5.95

Table 2.9, Continued

Season	Name	No. indivs.	Probability of detection	Effective detection radius (m)	Mean density/ km <sup>2</sup>	Density lower CI	Density upper CI	Density CV
	HOLA summer 3	140	0.20	112	191.15	126.09	289.77	5.36
	MEAD summer 1	94	0.14	226	29.60	21.97	39.86	3.81
	MEAD summer 2	170	0.15	190	61.79	39.84	95.83	5.65
	MEAD summer 3	240	0.09	182	100.10	73.42	136.46	3.97
	MODO summer 1	277	0.03	142	282.66	215.33	371.06	3.48
	MODO summer 2	210	0.03	98	350.81	98.47	1249.84	17.95
	MODO summer 3	691	0.04	144	831.91	628.75	1100.71	3.58
	NOBO summer 1	20	0.25	378	3.18	1.30	7.75	11.57
	NOBO summer 2	48	0.22	327	5.96	3.38	10.50	7.32
	NOBO summer 3	51	0.33	344	5.68	3.34	9.64	6.86
	SWHA summer 1	17	0.48	554	0.82	0.38	1.78	10.09
	SWHA summer 2	39	0.45	534	2.12	1.24	3.64	6.95
	SWHA summer 3	18	0.07	205	5.63	0.92	34.50	26.31
	WEKI summer 1	191	0.02	111	305.78	197.72	472.90	5.61
	WEKI summer 2	324	0.05	174	204.63	143.74	291.31	4.53
	WEKI summer 3	219	0.12	164	125.00	92.93	168.13	3.80
Fall	MEAD fall 1	58	0.10	130	2.91	1.51	5.59	0.17
	MEAD fall 2	95	0.05	148	4.74	2.68	8.38	0.15

\*CASP densities represent densities of singing males

Table 2.10. Conversion of raw broadscale point count bird detections to estimated abundance within survey plot area. I divided the number of individuals detected by the probability of detection (P) to achieve the estimated actual abundance. Four-letter species codes can be found in Pyle and DeSante (2012).

Season	Species	Month	2012			2013			2014		
			no. indivs. detected	P	Estim. abund.	no. indivs. detected	P	Estim. abund.	no. indivs. detected	P	Estim. abund.
Winter	FEHA	Dec	10	1.00	10.0	2	1.00	2.0	8	0.31	25.6
		Jan	1	1.00	1.0	6	1.00	6.0	12	0.31	38.5
		Feb	7	1.00	7.0	2	1.00	2.0	8	0.31	25.6
	HOLA	Dec	61	0.01	4075.4	450	0.03	12991.9	201	0.06	3563.6
		Jan	69	0.01	4609.9	138	0.03	3984.2	155	0.06	2748.1
		Feb	22	0.01	1469.8	87	0.03	2511.8	140	0.06	2482.1
	MEAD	Dec	16	0.07	223.6	14	0.09	155.2	69	0.04	1579.5
		Jan	15	0.07	209.7	40	0.09	443.5	76	0.04	1739.7
		Feb	31	0.07	433.3	25	0.09	277.2	96	0.04	2197.6
NOHA	Dec	8	0.49	16.2	6	0.06	102.3	10	0.33	30.2	
	Jan	4	0.49	8.1	10	0.06	170.4	9	0.33	27.2	
	Feb	6	0.49	12.2	8	0.06	136.4	11	0.33	33.2	
Spring	HOLA	Mar	30	0.08	370.5	39	0.02	2216.7	42	0.13	335.3
		Apr	34	0.08	419.9	17	0.02	966.2	39	0.13	311.4
	MEAD	Mar	61	0.01	8742.6	35	0.07	497.7	54	0.04	1530.8
		Apr	43	0.01	6162.8	53	0.07	753.7	70	0.04	1984.4
	MODO	Mar	11	0.03	328.4	36	0.04	888.9	176	0.03	5254.7
		Apr	78	0.03	2328.8	57	0.04	1407.5	259	0.03	7732.7
Summer	CASP*	May	22	0.43	51.0	24	0.13	180.4	21	0.27	76.7
		Jun	10	0.43	23.2	14	0.13	105.2	36	0.27	131.6
		Jul	7	0.43	16.2	15	0.13	112.8	15	0.27	54.8
		Aug	0	0.43	0.0	12	0.13	90.2	7	0.27	25.6
	HOLA	May	25	0.19	133.2	67	0.13	497.7	42	0.20	208.5
		Jun	26	0.19	138.5	20	0.13	148.6	83	0.20	412.0
		Jul	7	0.19	37.3	23	0.13	170.8	35	0.20	173.8

Table 2.10, Continued

Season	Species	Month	2012			2013			2014		
			no. indivs. detected	P	Estim. abund.	no. indivs. detected	P	Estim. abund.	no. indivs. detected	P	Estim. abund.
		Aug	13	0.19	69.3	12	0.13	89.1	25	0.20	124.1
	MEAD	May	36	0.14	254.1	49	0.15	321.2	59	0.09	682.4
		Jun	39	0.14	275.3	40	0.15	262.2	82	0.09	948.4
		Jul	18	0.14	127.0	44	0.15	288.4	78	0.09	902.1
		Aug	2	0.14	14.1	43	0.15	281.9	26	0.09	300.7
	MODO	May	93	0.03	2939.3	81	0.03	2608.3	255	0.04	6739.5
		Jun	97	0.03	3065.7	135	0.03	4347.1	401	0.04	10598.2
		Jul	113	0.03	3571.4	150	0.03	4830.2	502	0.04	13267.5
		Aug	136	0.03	4298.4	181	0.03	5828.4	553	0.04	14615.4
	NOBO	May	5	0.25	20.3	12	0.22	55.0	7	0.33	21.3
		Jun	12	0.25	48.6	7	0.22	32.1	15	0.33	45.6
		Jul	6	0.25	24.3	18	0.22	82.4	28	0.33	85.0
		Aug	0	0.25	0.0	15	0.22	68.7	1	0.33	3.0
	SWHA	May	2	0.48	4.2	7	0.45	15.7	5	0.07	74.7
		Jun	4	0.48	8.3	16	0.45	35.9	6	0.07	89.6
		Jul	7	0.48	14.6	11	0.45	24.7	3	0.07	44.8
		Aug	7	0.48	14.6	11	0.45	24.7	4	0.07	59.7
	WEKI	May	48	0.02	2506.8	160	0.05	3201.5	98	0.12	805.1
		Jun	84	0.02	4387.0	117	0.05	2341.1	87	0.12	714.8
		Jul	117	0.02	6110.4	158	0.05	3161.5	144	0.12	1183.1
		Aug	16	0.02	835.6	141	0.05	2821.3	36	0.12	295.8
Fall	WEME	Sept	27	0.09 7979	275.6	31	0.05	595.0			
		Oct	36	0.09 7979	367.4	125	0.05	2399.1			

\*CASP densities represent densities of singing males

Table 2.11. Results from statistical tests comparing seasonal density per species across years. Statistical significance was set at  $\alpha=0.05$ . November was excluded from winter analyses because the November survey in year one occurred before methods were standardized. Winter includes December – February, spring includes March and April, summer includes May – August, and fall includes September and October. Four-letter species codes can be found in Pyle and DeSante (2012).

Season	Species	df	Test statistic	P
Winter	FEHA	2,6	F 7.11	0.026
	MEAD	2,6	F 23.39	0.0015
	HOLA	2,6	F 0.76	0.51
	NOHA	2,6	F 68.54	<0.0001
Spring	HOLA	2,3	F 11.37	0.040
	MEAD	2,3	F 23.17	0.015
	MODO	2,3	F 10.62	0.044
Summer	CASP	2,9	F 7.34	0.013
	HOLA	2,9	F 2.53	0.13
	MEAD*	2	$\chi^2$ 8.65	0.013
	MODO	2,9	F 16.93	0.0009
	NOBO	2,9	F 1.87	0.21
	SWHA	2,9	F 22.31	0.0003
	WIKI	2,9	F 7.27	0.013
Fall	MEAD**	1.08	<i>t</i> -1.86	0.30

\*I could not achieve normality and equal variances for MEAD in summer, so I conducted a Kruskal-Wallis nonparametric test.

\*\*I conducted a student's t-test for MEAD in fall

Table 2.12. Tukey's Honest Significant Differences Test for multiple comparison of means for ANOVAs with significant results. Four-letter species codes can be found in Pyle and DeSante (2012).

Season	Species	years	diff	P
Winter**	FEHA	3-2	2.33	0.03
		MEAD	3-1	1.9
	NOHA	3-2	1.92	0.0024
		2-1	2.43	0.000061
		3-1	0.94	0.0097
		3-2	-1.49	0.00096
Spring	HOLA	3-2	-1.51	0.044
	MEAD	2-1	-6826.5	0.016
		3-2	-5695.5	0.027
Summer	MODO	-	-	NS
	CASP	2-1	99.5	0.01
		3-1	1.15	0.001
	SWHA	3-2	0.94	0.004
		3-1	56.77	0.00031
		3-2	41.92	0.0027
	WIKI	3-1	-1.41	0.024
3-2		-1.45	0.021	

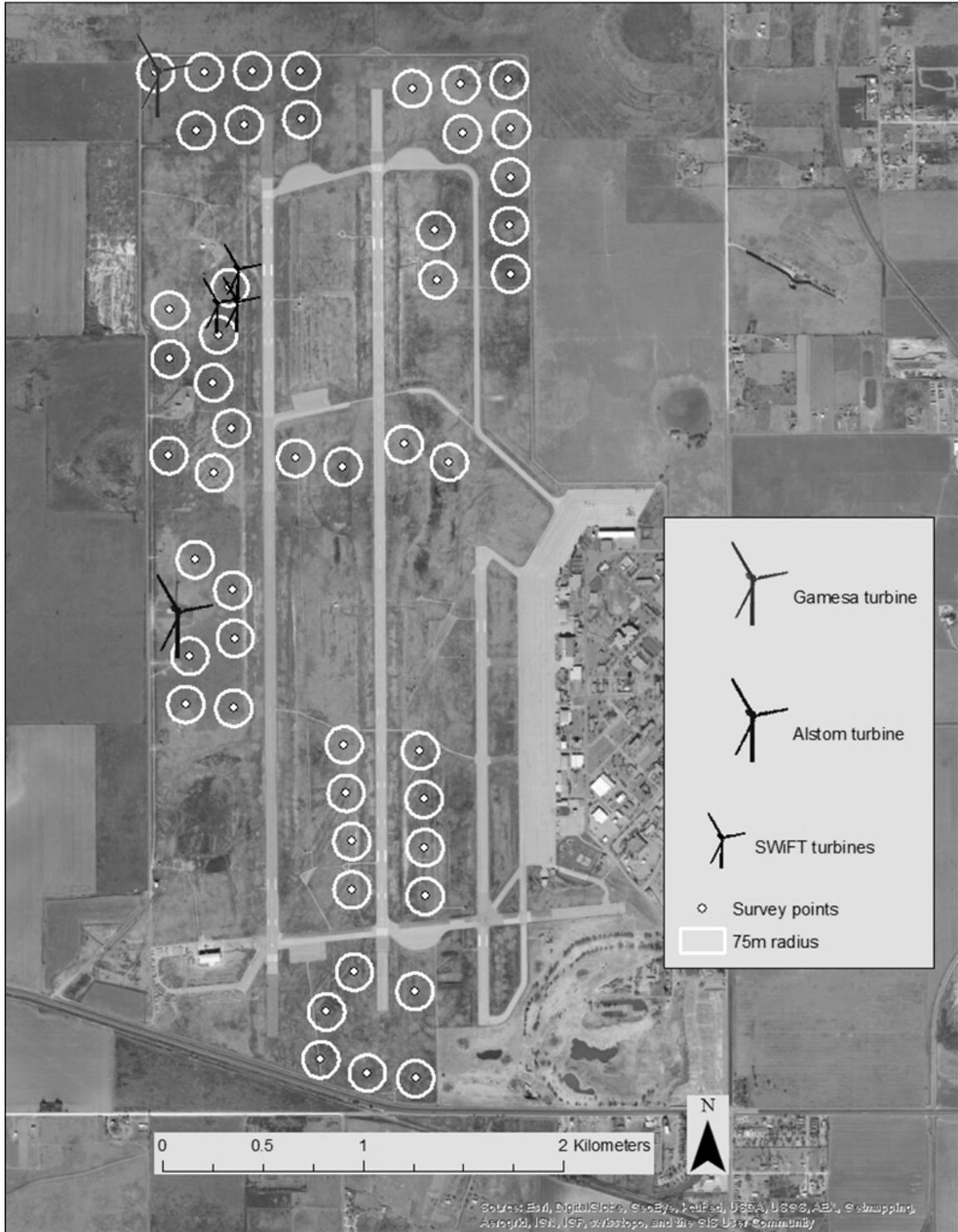


Figure 2.1. RTC site map depicting 75 m radius breeding bird survey point locations and turbine locations.

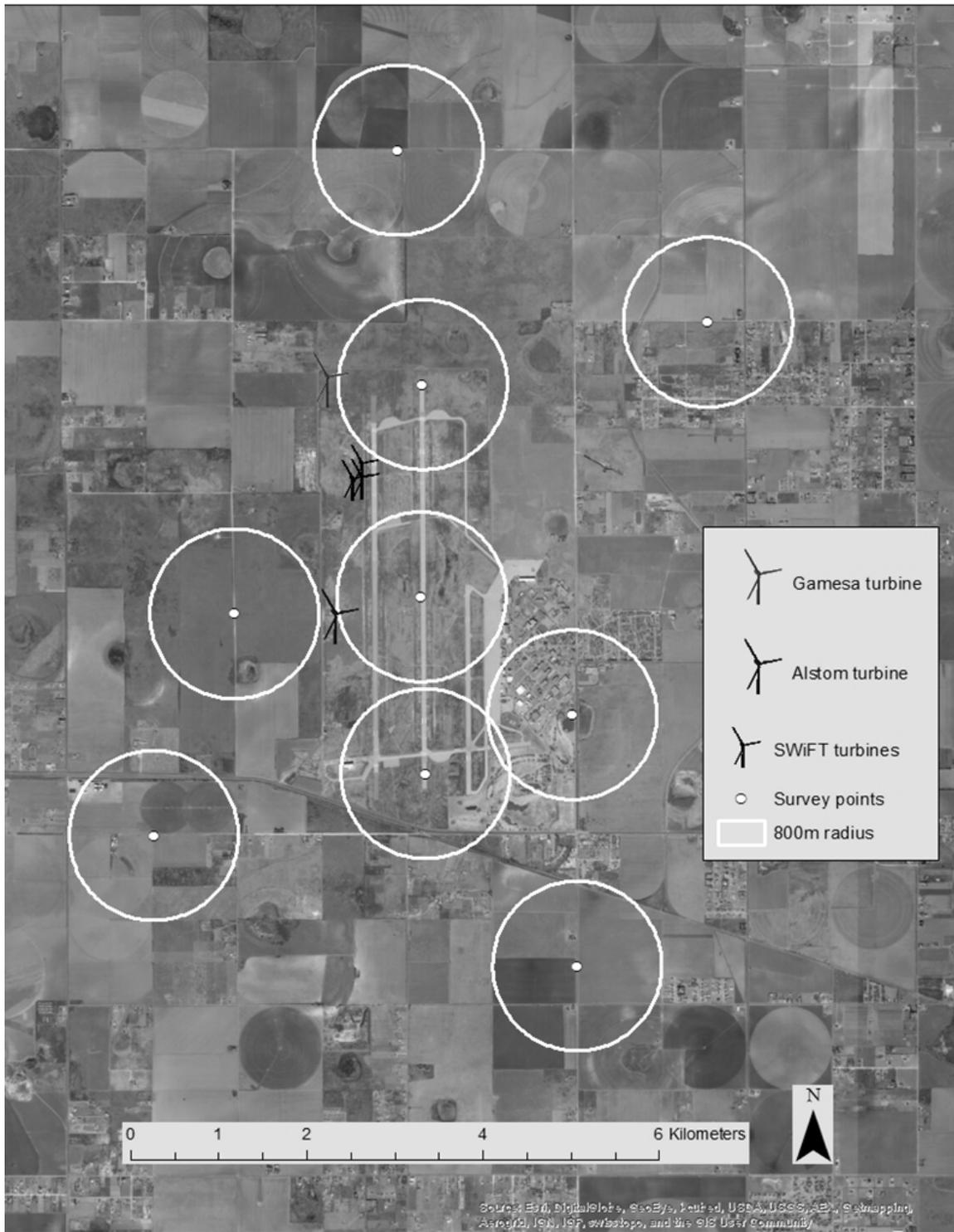


Figure 2.2. RTC site vicinity map depicting large-plot 800 m radius broad scale point counts and turbine locations.

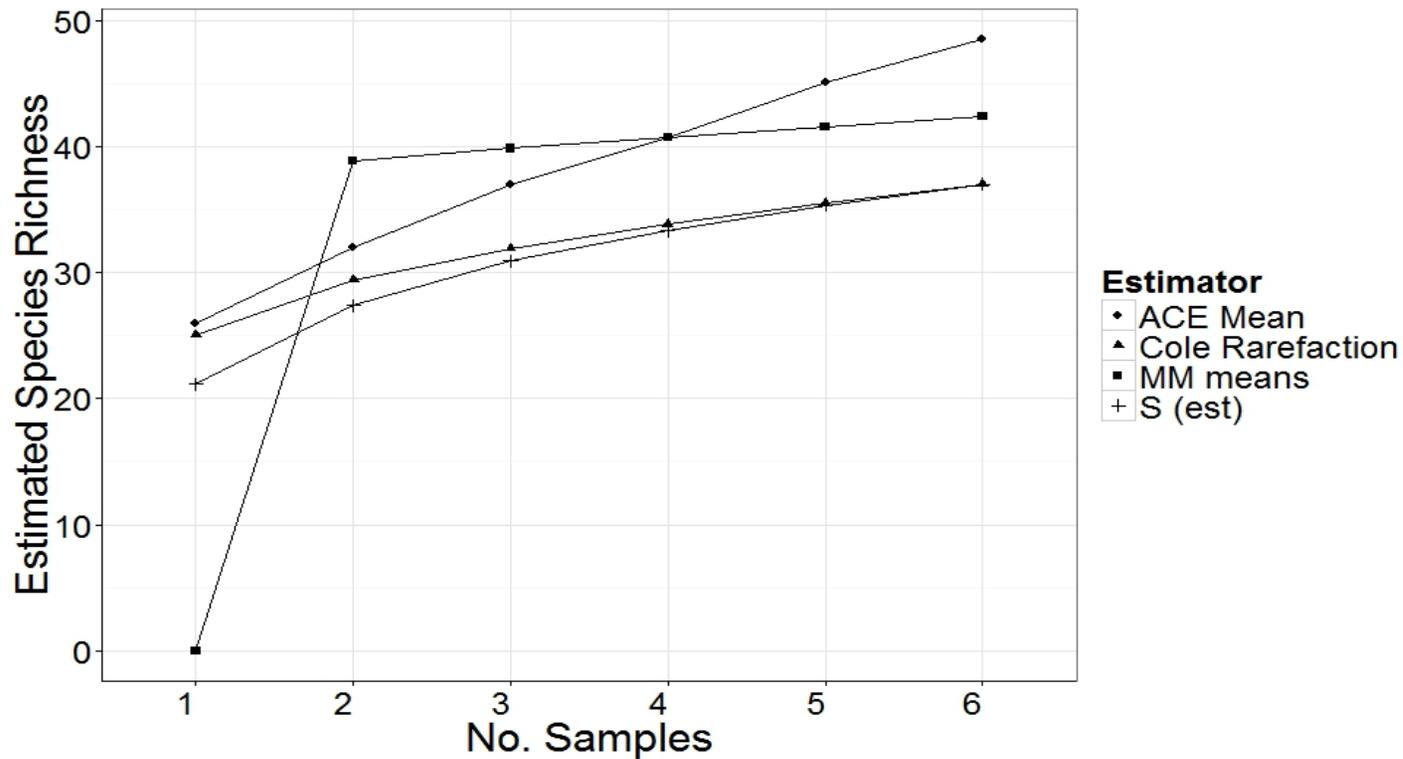


Figure 2.3. Species accumulation curves for breeding bird community assessment. The estimated species richness (y axis) increases with more survey samples (x axis). Each survey (each month) was equal to one sample. The rate of estimated accumulation of new species depends on the richness estimator (legend). The ACE Mean is the abundance coverage-based estimator (mean among runs), Cole Rarefaction represents the Coleman rarefaction estimator (number of species expected in pooled samples, assuming individuals distributed at random among samples), MM means is the Michaelis-Menten richness estimator where the estimators are averaged over randomizations, S (est) is the expected number of species in pooled samples, given the reference sample (analytical) (Colwell 2013).

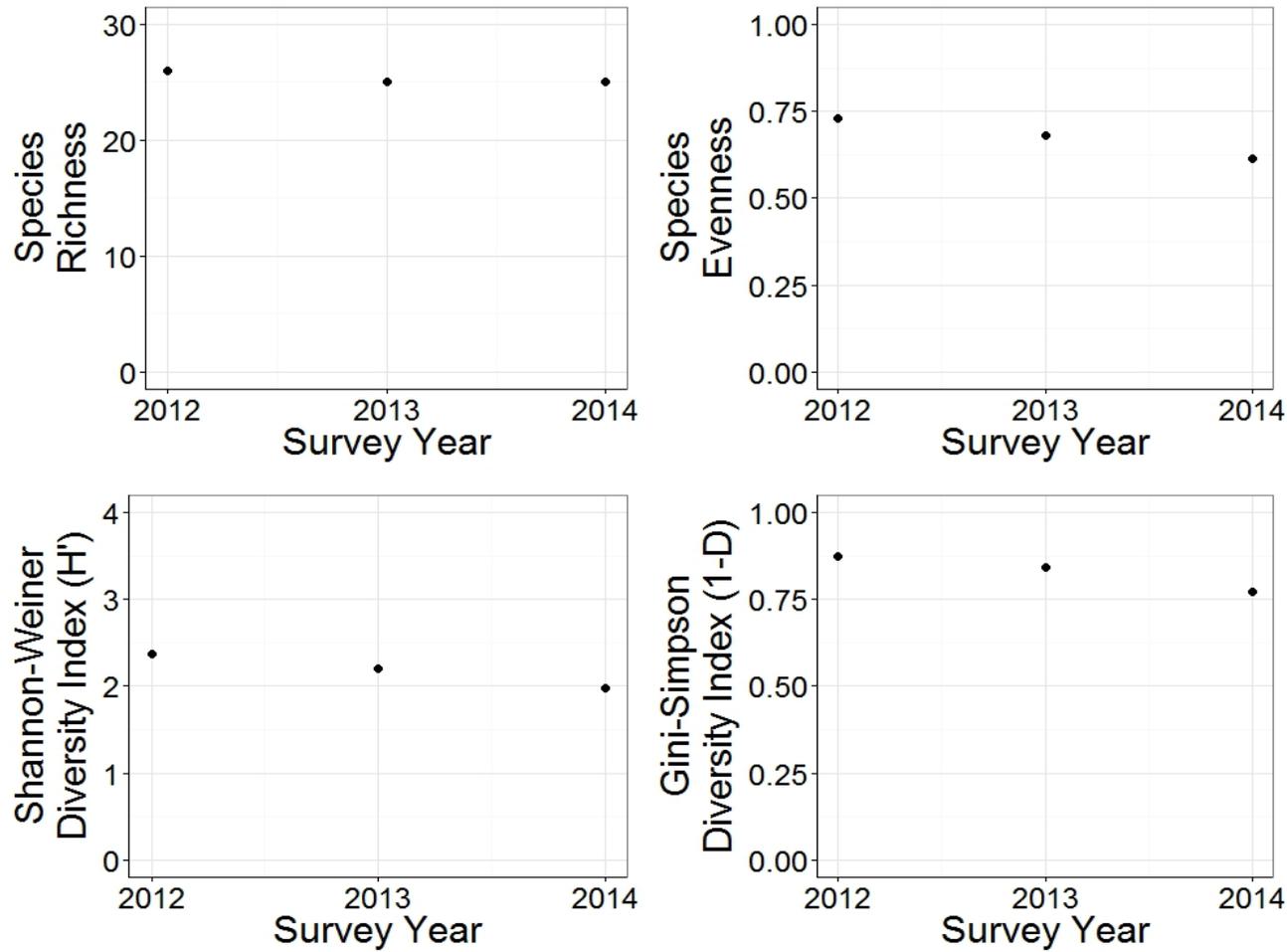


Figure 2.4. Diversity metrics for breeding bird surveys across years. Each point represents the diversity value of each year based on pooled breeding survey months.

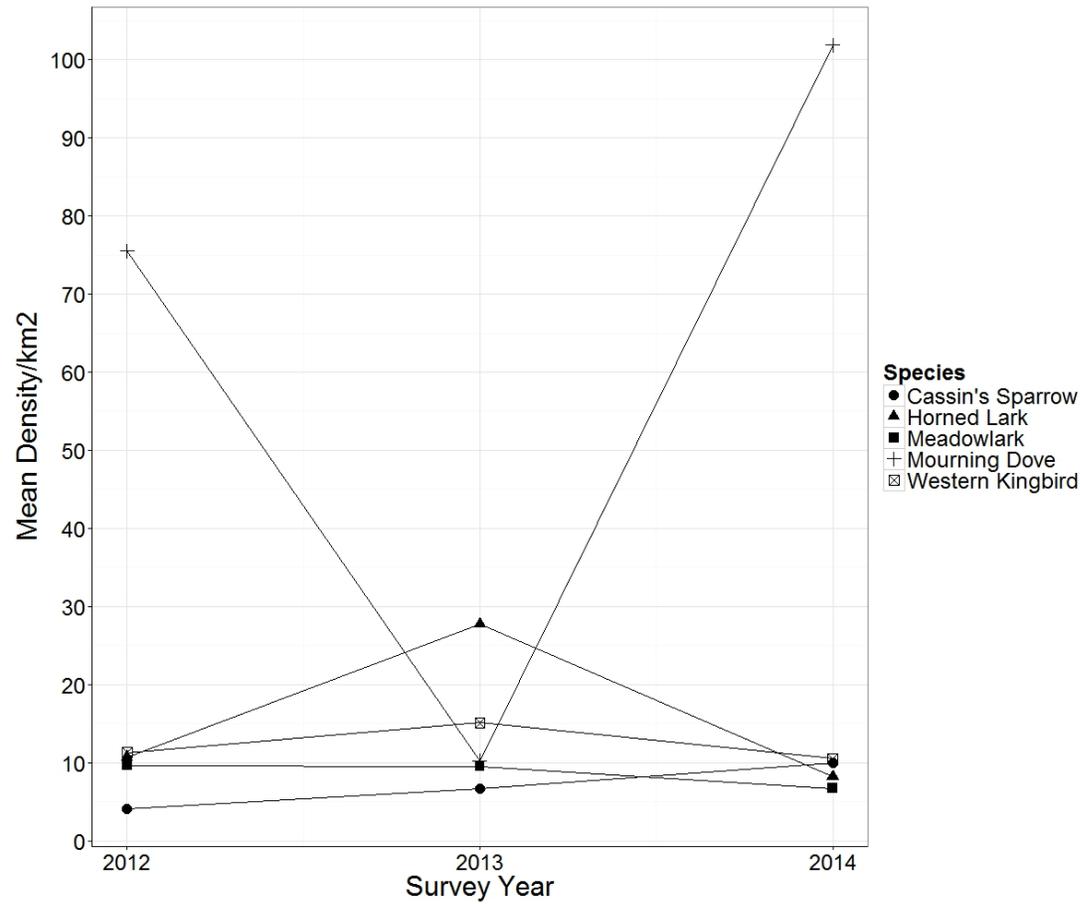


Figure 2.5. Estimated density from the five most common species detected in breeding bird surveys per year. Points represent the mean density of May and June surveys in their respective year.

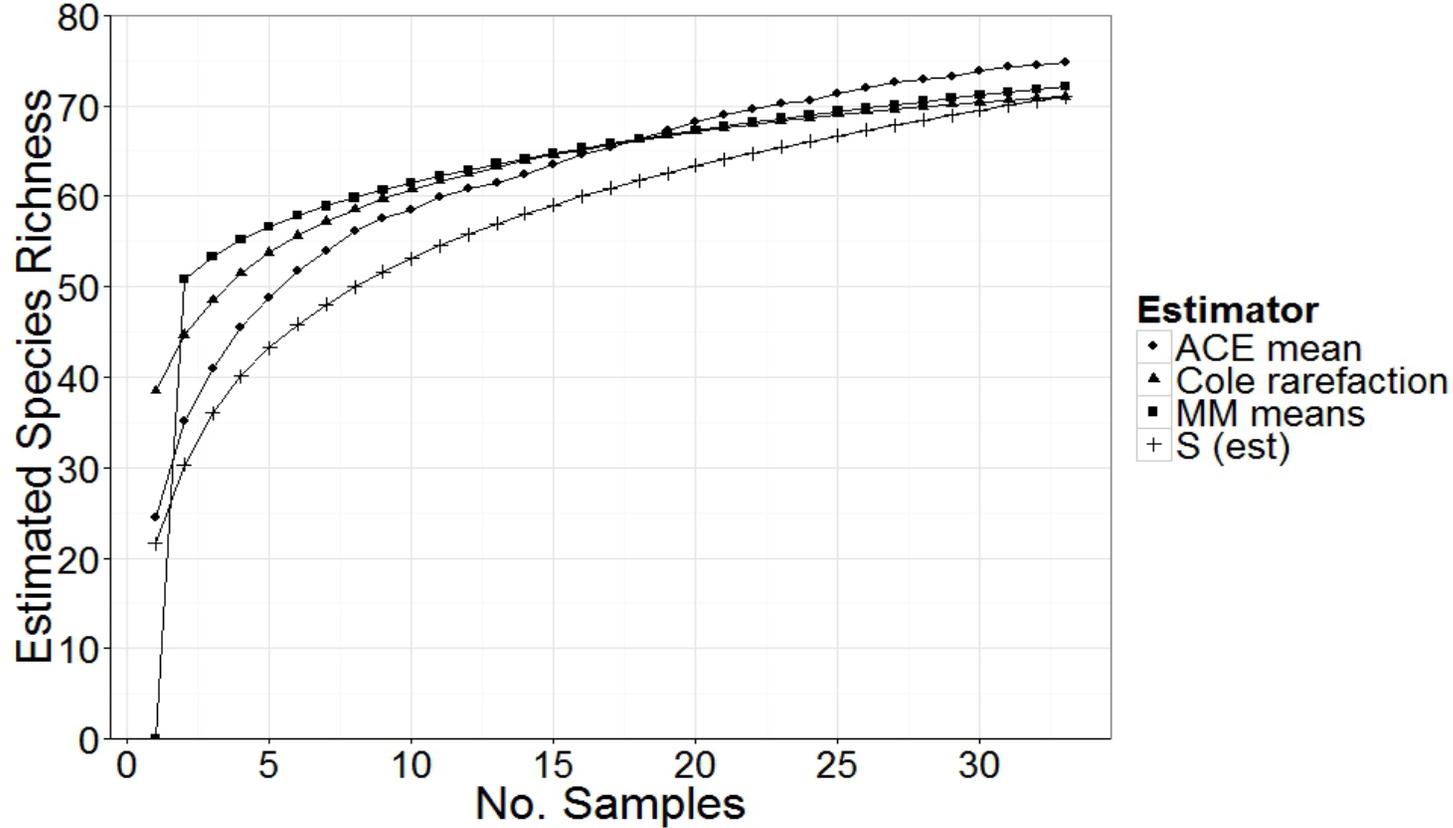


Figure 2.6. Species accumulation curves for broad scale point count surveys. The estimated species richness (y-axis) increases with more survey samples (x-axis). Each survey session (each month) was equal to one sample. The rate of estimated accumulation of new species depends on the richness estimator (legend). The ACE Mean is the abundance coverage-based estimator (mean among runs), Cole Rarefaction represents the Coleman rarefaction estimator (number of species expected in pooled samples, assuming individuals distributed at random among samples), MM means is the Michaelis-Menten richness estimator where the estimators are averaged over randomizations, S (est) is the expected number of species in pooled samples, given the reference sample (analytical) (Colwell 2013).

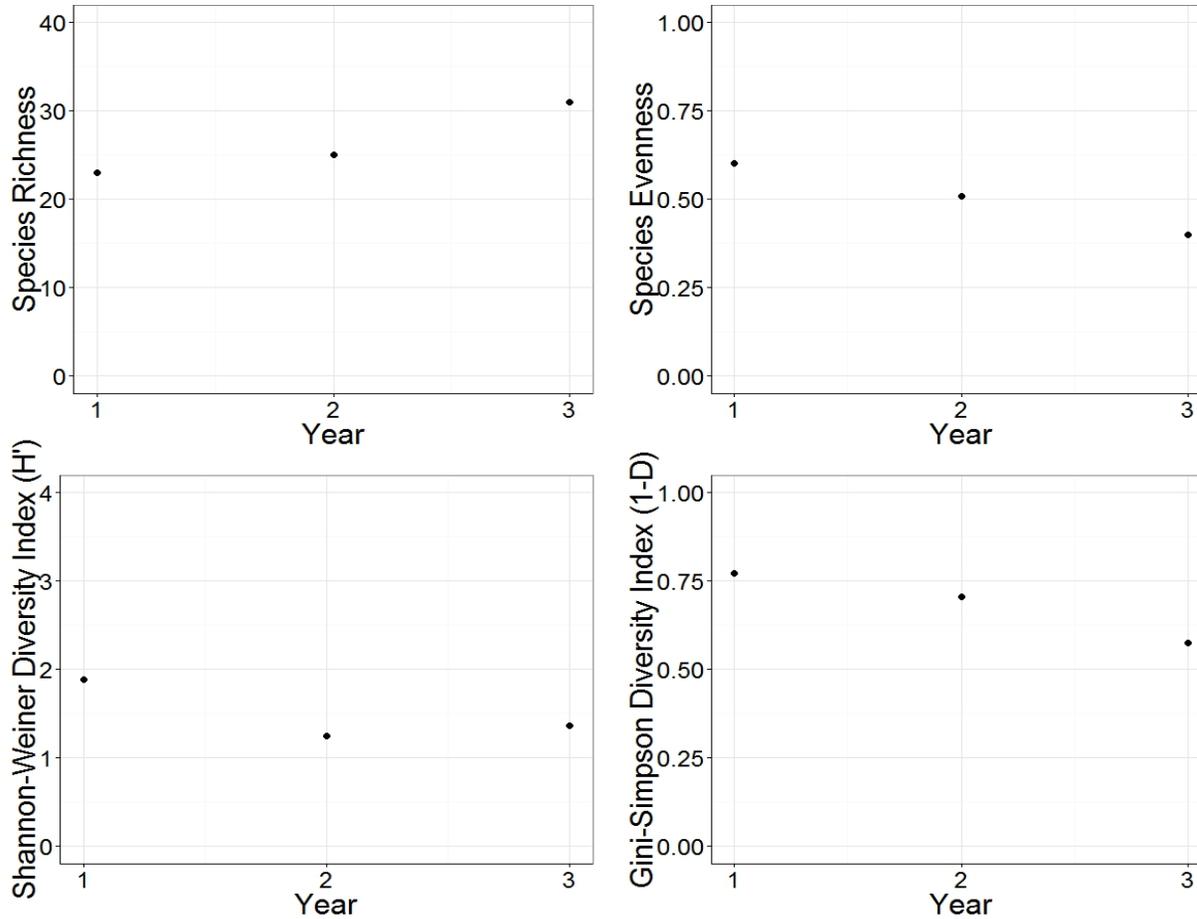


Figure 2.7.1. Comparison of winter diversity indices among years. On the x-axis, year 1 is 2012, year 2 is 2013, and year 3 is 2014. The points represent the values of the different diversity metrics (denoted on the y-axis) in the winter of each survey year. The diversity values were calculated from the mean species' abundances per winter season (n=4 months each season, or 12 surveys total, except for 2012 where only 3 months were included in winter, or 9 surveys total).

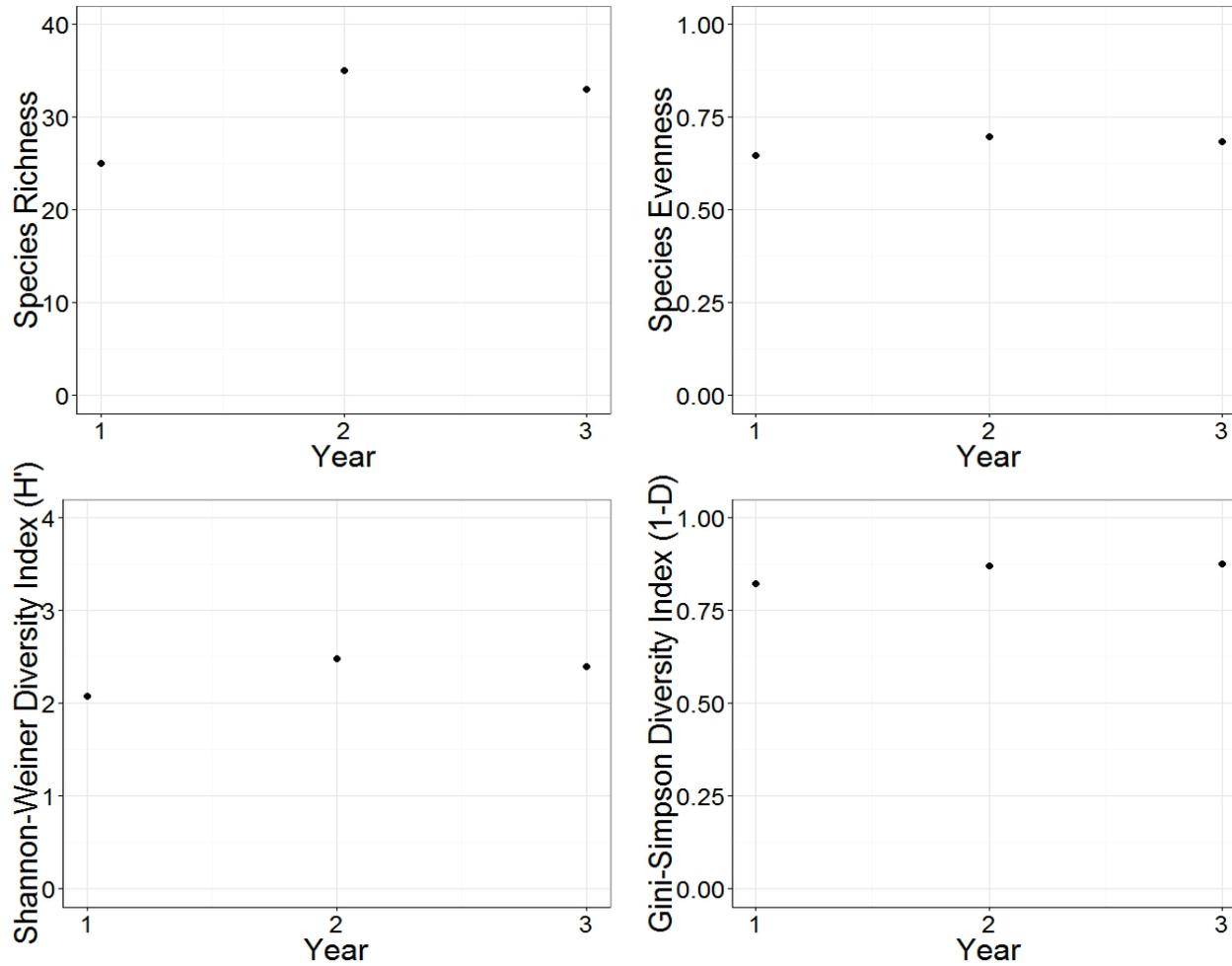


Figure 2.7.2. Comparison of spring diversity indices among years. On the x-axis, year 1 is 2012, year 2 is 2013, and year 3 is 2014. The points represent the values of the different diversity metrics (denoted on the y-axis) in the spring of each survey year. The diversity values were calculated from the mean species' abundances per spring season (n=2 months each season, or 6 surveys total).

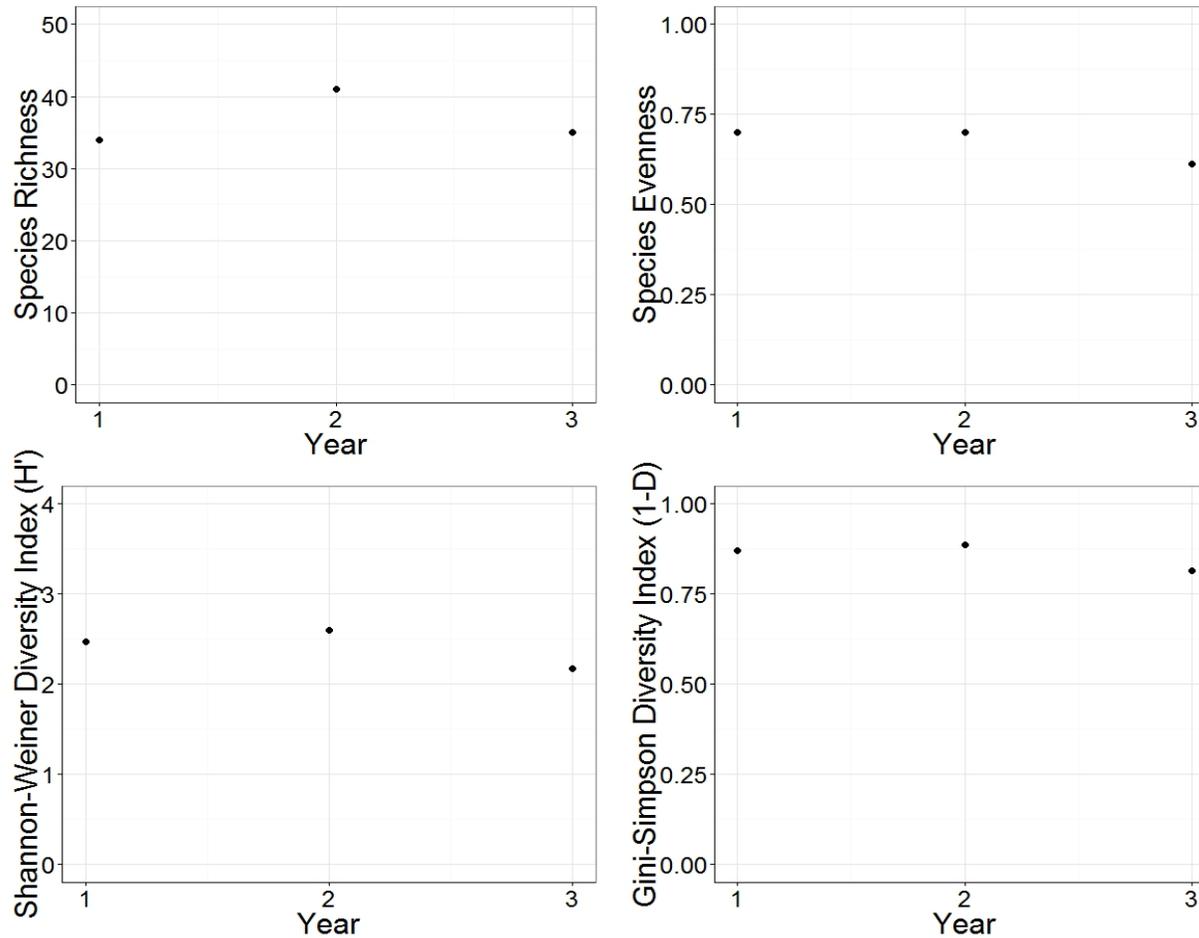


Figure 2.7.3. Comparison of summer diversity indices among years. On the x-axis, year 1 is 2012, year 2 is 2013, and year 3 is 2014. The points represent the values of the different diversity metrics (denoted on the y-axis) in the summer of each survey year. The diversity values were calculated from the mean species' abundances per summer season (n=4 months each season, or 12 surveys total).

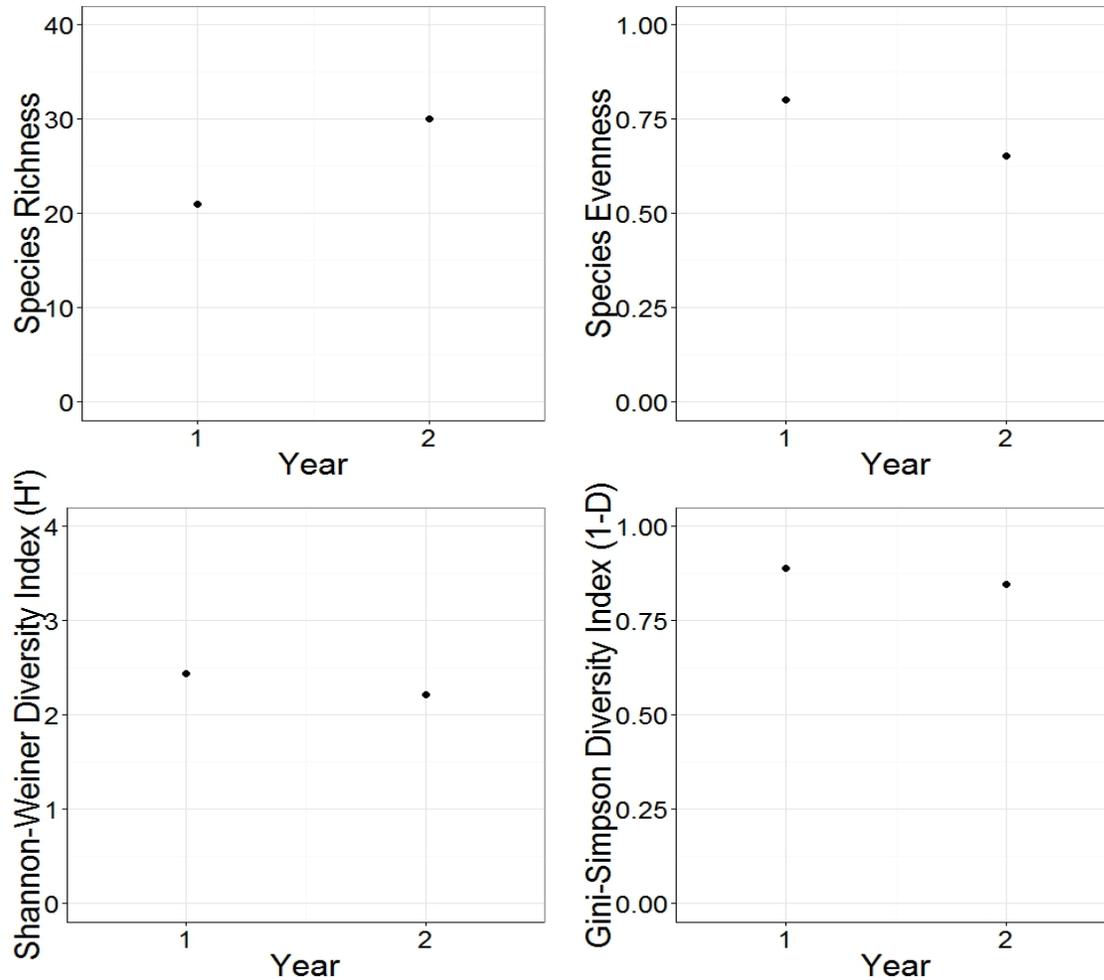


Figure 2.7.4. Comparison of fall diversity indices among years. On the x-axis, year 1 is 2012, and year 2 is 2013. The points represent the values of the different diversity metrics (denoted on the y-axis) in the fall of each survey year. The diversity values were calculated from the mean species' abundances per fall (n=2 months each season, or 6 surveys total).

## CHAPTER III

### DETERMINING PATTERNS OF LAND-USE AND PLAYA ASSOCIATION TO ESTIMATE WIND ENERGY DEVELOPMENT RISK TO RAPTORS ON THE LLANO ESTACADO

#### INTRODUCTION

The Llano Estacado is a primary area for wind energy development (Combs 2008) due to its commonly high wind speeds and flat landscape. The Llano Estacado is a high, flat mesa within the Southern High Plains (SHP) that covers northwest Texas and northeast New Mexico (Leatherwood 2010). Although wind energy has benefits as a clean source of energy, it also can negatively affect the fauna in its surroundings (de Lucas et al. 2007, Kuvlesky et al. 2007, Strickland et al. 2011). Raptors are known to be especially susceptible to collisions with wind turbines (Nelson and Curry 1995, Smallwood and Thelander 2005, National Research Council 2007), and numerous bird of prey species have historically occupied the Llano Estacado (Stevenson 1942, Allan and Sime 1943, Seyffert 2001). The Panhandle region of Texas, which includes much of the Llano Estacado, experiences some of the state's highest-quality winds (Combs 2008); thus this region appears to be attractive to both raptor species and wind energy interests. When raptor landscape preferences overlap with those that maximize wind energy output, their populations could be affected by collisions with rotor blades, disturbance, or land-use changes that lead to habitat loss (National Research Council 2007, Strickland et al. 2011).

Three distinct components mentioned above have unique aspects in the Llano Estacado: wind energy development, raptor occurrence, and landscape characteristics. Each component has a history of research with each of the other components – raptor

risks associated with, and response to, wind energy development (McIsaac 2001, de Lucas et al. 2008, Garvin et al. 2011), landscape characteristics ideal for wind energy development (Elliot 1977, Heister and Pennel 1981), and raptor habitat association on small and large scales (Andersen et al. 1985, Garner and Bednarz 2000, Canavelli et al. 2003). Studies have also looked at all three components (Hoover and Morrison 2005, Fielding et al. 2006, Madders and Whitfield 2006). Assessing the relationship and interaction among all three components in the Llano Estacado may predict risk to raptors from future wind energy development based on raptor land-use selection.

In regards to wind energy development, the state of Texas has more than double the installed wind capacity of any other state (AWEA 2014) at nearly 13,000 megawatts total. Texas continues to be top state for wind energy construction activity as well (AWEA 2014) and it is expected that much development will occur on the Llano Estacado. Heister and Pennel (1981) advise that when prospecting for wind resource areas, candidate sites experience usable winds and fulfill land-use and accessibility criteria. Ideal site attributes include high mean annual wind speeds, low turbulence, and remoteness from human habitation (Jenkins 1993). Terrain, especially, is an important consideration: locating turbines in flat terrain generally results in lower costs for development and operation because wind variability and extreme wind events are reduced (Heister and Pennel 1981) and allows for smooth, undisturbed windflow (Jenkins 1993). The SHP is the epicenter of a broad, flat region with among the highest average wind power available for extraction in the lower 48 states (Elliot 1977). Clearly the relationship between first two components of my study, landscape and wind energy development, is important in determining wind energy risk to raptors.

Raptors, the third component, have a higher risk of turbine collisions compared to other birds (Smallwood et al. 2009b). There is potential for high collision rates to influence species' populations because raptors occur in low densities and have long lifespans, delayed maturity, and generally low reproductive output. Smallwood and Thelander's (2008) study at the Altamont Pass Wind Resource Area (APWRA) shows that ongoing operations kill large numbers of raptors and other birds based on mortality counts and resulting fatality estimates. The most common fatalities they found were Red-tailed Hawks (*Buteo jamaicensis*), Burrowing Owls (*Athene cunicularia*), American Kestrels (*Falco sparverius*), and Golden Eagles (*Aquila chrysaetos*). Smallwood et al. (2007) believe that APWRA might act as an ecological sink to Burrowing Owls because they estimated that between one-fifth and twice the number of owls in the site are killed annually. However, nearly all collision rates estimated at other turbine mortality studies' collision rates are low, and raptor populations were likely not impacted by turbine collisions (Kuvlesky et al. 2007).

APWRA supports a high density of resident and migrant raptors, which may help explain the high fatality rates. Kuvlesky et al. (2007) believed the reason behind APWRA's high rate of collision fatalities is the site's location within a migration route in California. Orloff and Flannery (1992) assert that locating wind farms along migratory routes could result in higher collision rates. Approximately 71% of avian collisions in one study were categorized as migrants (as opposed to 20% breeding species and 9% residents; Johnson et al. 2002) and collision risk is higher in seasons of migration (Kuvlesky et al. 2007). According to Drewitt and Langston (2006), collision risk is dependent on many factors: the design of the wind farm, characteristics of turbines,

weather conditions, topography, specific species characteristics, abundance, and avian behavior. Johnson et al. (2002) also attribute many collisions to inclement weather; they estimated that up to 93% of turbine collision mortalities were associated with storms and fog. Smallwood et al. (2009b) hypothesize that raptor collision risk increases in high winds because he observed the highest proportion of American Kestrels flying within 50 m of wind turbines during high winds. One recent study asserts that abundance is not a good indicator of raptor fatality risk, but that topological factors and species behavior are good indicators (de Lucas et al. 2008). Species behavior is most often assessed by analyzing flight heights within the rotor-swept zone and flight activities near turbines. Osborn et al. (1998) observed that 31% of American Kestrels flew at heights within the rotor-swept zone and 67% flew within 15 m of wind turbines, indicating that this species in particular may be at a higher risk to turbine collisions. Wulff (2010) found that, along with waterfowl, raptors had the greatest risk of collision as indicated by their flight heights in a wind farm in the Texas Panhandle. In the APWRA, American Kestrels and Red-Tailed Hawks were the species that most often flew through the rotor zone and they experienced high fatality rates compared to other birds. In the same study, Burrowing Owls (which also have relatively high fatality rates) and kestrels were often observed interacting with other birds near the rotor-swept zones (Smallwood et al. 2009b). One physiological factor is visual acuity: raptors' frontal visual field is not in high resolution and they often employ lateral vision to detect conspecifics or prey during flight, meaning that they may not detect seemingly conspicuous objects such as wind turbines in their flight path (Martin 2011).

In addition to turbine collisions and mortality risk, there are indirect ways that wind energy can impact birds of prey. Wind energy facility construction and maintenance can alter the local area in multiple ways including vegetation clearing, soil disruption, and erosion (National Research Council 2007). Besides localized habitat disturbance, Garvin et al. (2011) found that raptor abundance decreased from pre- to post-construction of a wind farm, suggesting a displacement effect. Although abundance did not decrease in another study, nesting habits of locally breeding raptors were influenced by turbine presence. In this study, Usgaard et al. (1997) searched within the wind resource area and only found raptor nests in areas absent of turbines although adequate nesting vegetation was present in the turbine areas. However, the current general consensus is that, of direct and indirect effects from turbines, collision mortalities have greater population-level effects on raptors than do displacement or disturbance (Kuvlesky et al. 2007).

The direct impact of turbine collisions and indirect effects of habitat loss or range displacement on raptor populations will vary depending on the species. Two examples of potentially large raptor population effects due to wind turbine collisions are of Golden Eagles at the APWRA (Hunt 2002) and Red Kites (*Milvus milvus*) in Germany (Bellebaum et al. 2013). Radio-tagged and nesting Golden Eagles were studied in the APWRA and the resulting estimation of 40-60 turbine strike deaths per year is substantial. Hunt (2002) found that dangerous factors were turbine designs with lattice towers, the close proximity of turbines, the low sweep of the rotor blades near the ground, and the high prey density of squirrels. Their demographic analysis yielded a model that indicates the population is failing to maintain individuals to buffer the eagle population when conditions are poor. Models of annual collision rates for the Red Kite in Germany

represent approximately a 3% fatality rate of the population of about 3,000 post-breeding birds per year (Bellebaum et al. 2013). They calculated the mortality threshold to be 4% using the potential biological removal method, which could be reached in the near future as turbine numbers increase.

With its characteristically high winds and landscape ideal for wind energy development, the Llano Estacado may represent a region of comparatively higher risk for raptor turbine collisions. The period of highest wind power in Texas is in the spring (Elliot 1977), which coincides with raptor migration through the area. Miller (2008) conducted the first wind farm carcass searches in the Texas Panhandle; in her study, raptors comprised 47% of mortalities, of which Turkey Vultures (*Cathartes aura*) were 36%. Wulff (2010) studied avian flight heights in proximity to turbines in the central Texas Panhandle and found that a small number of raptors made up 21% of species that often flew into the rotor-swept zone, indicating high risk of collision. Raptor surveys in relation to wind energy development is relatively new; the history of raptor surveys on the Llano Estacado spans from Stevenson's (1942) bird records and Allan and Sime's (1943) West Texas highway surveys to Oberholser et al.'s statewide (1974) and Seyffert's (2001) Panhandle compilations of bird observations. Recently, Behney et al. (2012) surveyed for raptors in the western part of the Texas Panhandle and found Swainson's Hawks to be the most abundant raptor in the summer, with a much more diverse composition of raptors in the fall and winter. Additionally, Merriman et al. (2007) surveyed raptors in Lubbock County, Texas to determine patterns of abundance in areas occupied and unoccupied by prairie dogs and found species-specific relationships. The most abundant raptor reported by many surveys on the Llano Estacado year-round is the

American Kestrel. The common breeding raptors include American Kestrels, Swainson's Hawks (*Buteo swainsoni*), Burrowing Owls, and Mississippi Kites (*Ictinia mississippiensis*). Red-tailed Hawks, Ferruginous Hawks (*Buteo regalis*), and Northern Harriers (*Circus cyaneus*) are the most abundant wintering birds in the study area.

Although the same landscape may be attractive to both wind energy developers and to raptors, raptors certainly have motivations other than wind speed for selecting quality habitat. For example, the height or structure of vegetation can influence raptor distribution because it affects the abundance and availability of prey. Red-tailed hawks responded to prey abundance by selecting for or against certain areas with different rodent densities (Garner and Bednarz 2000). Canavelli et al. (2003) examined Swainson's Hawk habitat use at the landscape level over two provinces in Argentina and found that the availability of food (insect outbreaks) appeared to drive the hawks' association with agricultural field types. Vegetation structure that facilitates prey capture in combination with an abundance of the prey influences time spent at foraging patches (Preston 1990) and increases raptor densities (Baker and Brooks 1981). Perch sites are also known to affect raptor distribution and density; foraging areas may be selected partially based on the availability of perches (Preston 1990, Garner and Bednarz 2000). American Kestrels will select foraging patches with perches and short vegetation despite actual prey density in plots (Sheffield et al. 2001). The density of perch sites is integral in Red-tailed Hawk selection of a nest territory for effective foraging (Janes 1984), and Swainson's Hawks will compete with Red-tailed Hawks for nesting areas with moderate densities of perches (Janes 1994). Raptors are somewhat species-specific in their perch site preferences (Worm et al. 2013, Bobowski et al. 2014).

For breeding birds, the potential for nest sites is also an important influence in landscape characteristic associations. The amount of open space can be important: American Kestrels nest in cavities within open areas (Smallwood and Bird 2002) and nest in higher frequencies in large patches of contiguous open space than small patches (Smallwood et al. 2009a). Some wintering raptor species in open space grasslands in Colorado were highly sensitive to landscape urbanization and were scarce in areas of development (Berry et al. 1998).

A unique landscape characteristic of the Llano Estacado is the presence of playas, the primary surface water in the region (Dvoracek 1981). Playas are shallow, circular depressions that serve as ephemeral water bodies and are a prominent feature in the SHP (Haukos and Smith 1992). Nearly 22,000 playas exist in the Llano Estacado and are havens for floral and faunal diversity including insects (Bolen et al. 1989) which are a common prey item of American Kestrels and Swainson's Hawks (Sherrod 1978; Giovanni et al. 2007). Black-tailed prairie dogs (*Cynomys ludovicianus*) in the Llano Estacado are strongly associated with playas, and Burrowing Owls are found in higher abundance when associated with prairie dog colonies (Pruett 2004). Pruett et al. (2010) detected different small mammal species compositions on colonized and non-colonized sites; certain small mammals may be important for raptor species. Pruett (2004) also noted Red-tailed Hawks and Swainson's Hawks associating with playas. Prairie dogs are a main prey item of Ferruginous Hawks (Cully 1991), and their abundance was found to be positively correlated with proximity to prairie dog colonies (Berry et al. 1998; Merriman et al. 2007).

Because of their trophic level, delayed maturity, low breeding densities, and low reproductive output, wind power-related effects may have population-level consequences for raptor species. To gain an understanding of raptor presence and habitat associations in the context of potential wind energy development, I conducted road transect surveys from December 2012 to November 2013. I assessed raptor density and abundance, species richness, evenness, and diversity per season in the Llano Estacado to evaluate within-year trends. I also evaluated selection of land-use types and association with plays in the region. I studied raptor associations with land-use types rather than specific habitat characteristics because wind energy siting is based on land-use and ownership. I do not believe birds of prey select habitat based on the land-use categories that I delineated, but rather they are selecting a combination of desirable habitat characteristics that coincide with the land-use category.

If patterns of raptor use can be found among such broad a scale as land-use categories, information gained may be useful for wind energy developers in the context of minimizing potential wind energy-wildlife conflict. I collected data by conducting a road transect survey that representatively sampled different land-use categories (as done with land cover; Andersen et al. 1985) on the Llano Estacado. The null hypothesis was that there would be no significant difference between the expected and observed land-use frequencies, implying that raptors select habitat in accordance with the proportion it is available. My alternative hypothesis was that the expected and observed frequencies would be significantly different, implying that raptors are selecting certain land-use types more or less than they are available. The information gained fulfills the objective of assessing the relationship among wind energy, landscape characteristics, and raptor

occurrence in this region. I identified landscape components (land-use type, playas) where wind energy, if developed, would pose a comparatively greater risk to raptors.

## METHODS

### *Study Area*

The study was conducted in Lubbock and Hockley Counties, Texas, in the Llano Estacado region of the SHP. This area is characterized by flat tablelands and located in the greater southern shortgrass prairie. The region is classified as semi-arid, with an average rainfall in the study site of between 48.6 centimeters and 50.4 centimeters annually and temperature averages ranging from an average winter low of -2.6 degrees Celsius to an average summer high of 33.1 degrees Celsius (NOAA 2014a, NOAA 2014b). Land-use practices are predominately agriculture (primarily cotton and grain sorghum), urban-developed, and herbaceous grassland (which includes grazed lands, Conservation Reserve Program land, and native grasslands; USDA 2009). Over 20,000 playas mark the land surface in the Texas Panhandle (Bolen et al. 1989). There are small concentrations of wind turbines in Lubbock and Hockley County, but currently no large-scale wind farms. The Reese Technology Center (RTC) in Lubbock County is a focal area along my survey due to its recent wind energy development. The decommissioned Air Force base currently has three scaled test turbines and two industrial-sized turbines; seven or more additional turbines may be erected on the RTC in the near future (Slyker 2012). The RTC is a native grassland (over 1,000 hectares) amidst concrete runways. Black-tailed prairie dog colonies are ubiquitous within the RTC and often where playas are present.

### *Data Collection and Management*

When assessing raptor densities, Fuller and Mosher (1987) consider line transects to be one of the best techniques for surveys. Road transect counts, despite inherent biases, are the best option for sampling raptor populations in open vegetation (Millsap and LeFranc 1988). I used this sampling method because of its practicality in an agricultural-grassland dominated landscape with abundant rural roads. Allen and Sime (1943) assert that the study area offers excellent hawk-watching opportunities because of the broad expanses of land without obstruction. Surveys methods were based on techniques suggested by Fuller and Mosher (1987).

I conducted a road transect survey that representatively sampled different land-use categories (as done with land cover; Andersen et al. 1985) from December 2012 to November 2013 in Lubbock and Hockley Counties, Texas. Two observers experienced with raptor identification made observations of raptors from a vehicle (as done in Andersen et al. 1985) along a 154 km by 1.6 km line transect (half-width of 800 m; see Fig. 3.1). The total area within the surveyed strip out to 800 m equaled 239.3 km<sup>2</sup>. The survey route was not selected randomly or broken into parallel survey sections due to constraints of available roads and the main objective of representative sampling of land-use types. The final survey route consists of north, west, south, and east transects that cover the north-south spans of Lubbock and Hockley Counties as well as the east-west area between urban centers (Lubbock, TX, to the east and Levelland, TX, to the west) which I intentionally avoided. The road survey samples the five land-use categories at an approximate proportion to their availability on the landscape as done by Andersen et al. (1985).

I broke the 154 km survey into two approximately equal-length sections (78 km and 76 km) to minimize searcher fatigue; these two survey halves were usually conducted on consecutive days. The survey was completed in the first ten days of each month. Both observers scanned the landscape for raptors and determined identification using binoculars or a spotting scope. Each survey began within one half-hour of sunrise and continued until the allotted route was finished. Although time of day affects the numbers of certain raptor species seen on roadside surveys (Bunn et al. 1995), I began all surveys in the morning to maintain consistency and for comparability. I alternated the starting points of the survey sections in order to minimize the time-of-day effects (Garner and Bednarz 2000). The vehicle was driven by the other observer at an average speed of 30 km/h. I also recorded time of day of sightings and weather conditions; surveys were not conducted in inclement weather (i.e., sustained wind speed over 30km/h or precipitation). During the surveys, I collected the GPS location of the observer, the perpendicular distance to the bird from the transect, and whether the bird was in movement (in accordance with distance sampling protocol; Buckland et al. 2001). To compare land-use associations I also collected bearing from the transect to the point and land-use types within a 100 m radius of the observation site in order to verify land-use in ArcGIS.

To assess land-use associations, I drove along the survey route and visually ground-truthed all land parcels within the 1.6 km wide transect area (see Fig. 3.1) every three months. These three-month increments represented a season in relation to raptor phenology and agricultural practices: December, January, and February – winter; March, April, and May – spring; June, July, and August – summer; and September, October, and November – fall. I began with Common Land Unit files (USDA-FSA) in ArcGIS and

edited the feature class along the route out in the field, adjusting the land units as crops rotated over the seasons. I grouped land-use types into five categories: cotton, grain crops (e.g., sorghum, winter wheat), grazed grassland (e.g., rangeland), non-grazed grassland (e.g., Conservation Reserve Program) and other (e.g., residential areas, fallow fields); the agriculture crops were further subdivided into standing and harvested (Table 3.1). I demarcated the categories based on land-use description rather than vegetation structure because while vegetation structure and community may be informative for high-quality raptor foraging habitat (Sheffield et al. 2001, Preston 1990), the intentions of this study involve wind energy prospects and the landscapes they would develop. As such, wind energy interests would view and/or assess land cover based on ownership and use.

It is important to obtain actual land-use data in the field rather than only using GIS layers because agricultural field conditions and crop stages will change depending on the season. These seasonal differences have an effect on vegetation structure, which influences the prey base and raptors' access to that prey, potentially affecting their habitat associations. In addition, in center-pivot irrigated agricultural fields, the corners outside of the circular area receiving irrigation are often a different cover type than the field, which is not accounted for in most land-cover layers. These portions may represent a significant amount of different habitat used by raptors and were drawn into ArcGIS for this analysis.

### *Analysis*

Diversity measures are commonly seen as robust indicators of the well-being of an ecosystem (Magurran 1988). I calculated raptor species richness, evenness, and

Shannon-Wiener ( $H'$ ) and Gini-Simpson (1-D) diversity indices according to Magurran (1988) per season using raw observation counts, only including raptors that were identified to species. I assessed two measures of diversity because the Gini-Simpson index is influenced more by species evenness, whereas Shannon-Wiener's index is influenced more by species richness. Comparing the two allows for a comprehensive assessment of diversity. In order to evaluate if the number of surveys adequately captured the species richness of the study area, I created species accumulation curves using the program EstimateS (Colwell 2013) as done by Verble and Yanoviak (2013).

In order to assess abundance and density, I first modeled the probability of bird detections by observers (Thomas et al. 2010). For line-transect surveys, detection is modeled as a function of distance from the line (Thomas et al. 2010) and explains an inverse relationship: as observations increase in distance from the survey route, probability of detection decreases. This applies to my 800 m half-width line transect survey; I am certain that observers could not accurately detect all raptors out to 800 m. The risk in this case is that density and abundance values would be underestimated. I estimated the detection function for each raptor species using program DISTANCE 6.2 (Thomas et al. 2010) and employing four key functions with adjustment terms based on recommendations by Thomas et al. (2010). I pooled observations within a species to calculate the effective strip width (hereafter ESW). I selected the appropriate detection function and series adjustment based on lowest AICc and best goodness-of-fit values. The ESW indicates the effective area surveyed (Thomas et al. 2002), which is the distance beyond the survey transect within which observers are reliably accurate in detecting samples and beyond which observers are significantly less accurate in making

detections. Many raptors perch on utility lines directly adjacent to the survey route, which biases the estimated ESW's to a lower distance that is not representative of observer visual detection. So I left-truncated the data to exclude observations of perched birds within 15 m of the route (excepting Northern Harriers, which are rarely seen perching immediately adjacent to the survey route) for this estimation.

I assessed raptor abundance as number of raptors seen per kilometer of survey using raw counts per season divided by the length of the survey route (154 km) in order to be comparable to other raptor studies. However, because detection function and repeat observations per season are not accounted for in this method, I calculated density using the species-specific detection model in Program DISTANCE (Thomas et al. 2010). I applied these density estimates to attain seasonal and annual abundance per species as well. I am confident that the abundance values calculated using the ESW's in Program DISTANCE are more accurate than the values calculated with the raw counts because it incorporates searcher effectiveness.

For the land-use selection analysis, I adjusted GPS points taken in the field by the bearing and distance and imported them into ArcGIS (Fig. 3.1). I overlaid raptor location points onto land-use polygons and only used the points within the ESW in analyses (e.g., Fig. 3.2). I placed a 100 m radius buffer around each raptor observation point such that it "punches out" a circle of raptor use from the land-use layer (e.g., Fig. 3.3). This method was utilized to prevent assigning only one land-use type to a raptor where it was observed, when the raptor may have been perched at the edge of one type of land-use but foraging in an adjacent plot, as suspected in Garner and Bednarz (2000). Although

raptors may utilize a larger foraging radius, a 100 m radius allows for a reasonable area to assess land-use composition at the raptor location. I generated an "available land-use" layer for each species per season by clipping it by the ESW with an additional 100 m so that the entire foraging buffer was included. I calculated the proportional availability of each land-use type and the proportional use of land-use types per season. I then converted the proportion availability and use to frequencies by multiplying the proportions by the sample size of the raptor for the season analyzed. Thus, the frequencies per available land-use category represented their expected frequency of use by raptors. This process was done separately for each raptor species and each season analyzed. Detailed steps are outlined in Appendix 3.1.

We assessed land-use selection by comparing use vs. availability with a chi-squared test (Neu et al. 1974). The level of significance was set at  $\alpha=0.05$ . Some of the "expected use" categories had frequencies of less than five, however Roscoe and Byers (1971) assert that having an average expected value of five or more across all categories (after dividing sample size by the number of categories) is sufficient. These data fulfilled those recommendations. Following this test, I evaluated the magnitude of difference between available and used land-use categories. I conducted the analysis for American Kestrels in all seasons; Red-tailed Hawks in winter, spring, and fall; and for Swainson's Hawks in spring, summer, and fall. I pooled Ferruginous Hawks observations and pooled Northern Harriers observations across fall, winter, and spring, due to low within-season sample sizes. Although Burrowing Owls were abundant in the summer, I did not analyze land-use selection of this species because individual observations are not independent; Burrowing Owls nest in colonies. I removed observations of fledglings in

summer and multiple sightings of adults on a nest from the analysis. I gauged potential wind energy risks based on which land-use types are associated with the highest proportional use by raptors.

I examined distribution of raptor species in relation to playas on the landscape using the methods of Pruett (2004), who studied the distribution of prairie dog colonies in relation to playas. I first obtained a digital layer of playas (USDI 2013) in the Llano Estacado and measured the distance of each raptor observation to the edge of the nearest playa using the "Near" tool in ArcGIS. For each species in the analysis, I generated an equal number of random points as observations of that species within its ESW using the Geospatial Modeling environment (Beyer 2012) and used the Near tool in ArcGIS to measure the distance from each point to the edge of the nearest playa in the digital layer. I calculated the mean and 95% confidence interval for observed distances and random distances and performed statistical tests using program R (R Core Team 2013). The null hypothesis tested was that the mean distance of raptor observations to the nearest playa was equal to the mean distance of random points to the nearest playa. The alternative hypothesis was that the mean of observation distances was less than the mean of random point distances; consequently, I used a one-sided Student's t-test.

Because I did not know wherein lies the importance of the playa to raptors - in the ephemeral water source itself (implying all playas are important to raptors) or in the native remnant wetland community (implying that only "intact" playas are important raptors) - I performed two separate analyses per species. I visually categorized playas as either "intact" (i.e., grassland vegetation present with little disturbance apparent) or

"modified" (e.g., a historical playa that has been plowed over for agricultural purposes) out to approximately 1.5 miles beyond the raptor survey route. For "intact" grasslands, I did not distinguish CRP, non-native, or native grass species; these affect playa hydrology and function differently (Smith et al. 2011). Thus, some information may be lost by only categorizing playas into the two groups. First, I compared the mean of observed to random distances to "intact playas," and second, I compared the mean of observed to random distances to all playas (including both intact and modified playas). Though playas may alter in importance depending on if they are wet or dry, at the time of each survey I did not record whether playas held water or not. I hypothesized that raptors would associate with intact playas, especially grassland raptor species such as the Ferruginous Hawk and the Northern Harrier. This result would indicate that the prairie-grassland aspect of the playa (as opposed to simply an intermittent source of surface water) is important to raptors.

I pooled all observations for each raptor species for analysis and only used observations located within the ESW. Prior to conducting the t-test to compare observed versus random points, I assessed each sample for normality using a Shapiro-Wilk's test (Shapiro and Wilk 1965) and equal variance using a box plot. I employed a square root transformation to meet assumptions for Ferruginous Hawks and Northern Harriers in both analyses, and for Red-tailed Hawks and Swainson's Hawks only in the "intact" playa analyses. American Kestrel data for all analyses and Red-tailed Hawk and Swainson's Hawk data for "all playa" analyses violated the normality and equal variance assumptions despite transformations, so I performed a one-sided Wilcoxon rank sum test. I could not

analyze Burrowing Owls because the samples are not independent (they nest in colonies) but still compared mean distances and CI's to random points.

## RESULTS

I detected 831 raptors of 15 species on my road surveys from December 2012 to November 2013 (Table 3.2). Monthly raptor detections varied from a low of 38 birds in January to a high of 106 in July (Table 3.2). American Kestrels were the only raptor detected year-round and I observed the highest number from July through December (Table 3.2, Fig. 3.4). Ferruginous Hawks, Red-tailed Hawks, and Northern Harriers were frequently detected in all seasons except summer (Table 3.3, Fig. 3.4). Mississippi Kites and Swainson's Hawks breed in the study region and were often observed in the summer (Table 3.3; Fig. 3.4). The number of species detected was consistently higher in spring and fall seasons than in winter or summer (Table 3.3), but the highest raptor abundance overall was detected in late summer months and the fall season (Table 3.3; Fig. 3.5). Surprisingly, I observed Great Horned Owl fledglings in July 2013 in the survey area, which infers an exceptionally late breeding attempt for that species (Houston et al. 1998).

The six most numerous species detected were the American Kestrel, Swainson's Hawk, Red-tailed Hawk, Burrowing Owl, Ferruginous Hawk, and Northern Harrier (Tables 3.2, 3.3). Over the entire year, these six species represented 90.1% of all raptor detections (Table 3.3). Winter detections were dominated by Red-tailed Hawks and American Kestrels (34.7% and 35.3% relative abundance, respectively; Table 3.3). The highest occurring raptor in spring was the Swainson's Hawk (35.3% relative abundance; Table 3.3). In summer, American kestrels, Swainson's Hawks, and Burrowing Owls

shared similar relative abundance values (33.3%, 27.9%, and 26.1%, respectively; Table 3.3). In fall, detections for American Kestrels yielded the highest relative abundance (32.7%, Table 3.3).

Measures of diversity were averaged over all surveys conducted within the three months making up each season. The greatest species richness occurred in spring and fall (Table 3.6). Species evenness and diversity (Shannon-Wiener and Gini-Simpson) also increased in the spring and fall but the increase was much less pronounced (Table 3.4; Fig. 3.6). All of the metrics demonstrated a rising and falling pattern that increased in migration seasons and decreased in winter and summer (Fig. 3.6), and all metrics had the highest value in the fall (Table 3.4). The cumulative species abundance estimators all depicted a plateau during the end of our sampling period (Fig. 3.7). This asymptote estimates the true species richness at the site so when the species richness found from sampling approaches this asymptote, it means that observers performed a sufficient number of surveys to adequately sample the expected assemblage in the study area (Chao et al. 2009).

For density and abundance estimates, I assessed the fit for various detection models for the six raptor species with the greatest sample sizes (see above), and nearly every species utilized a different detection model adjustment (Table 3.5). This is important because the method affects the way in which density is calculated. The results from program DISTANCE (Thomas et al. 2010) show that observers' effective distance for detecting Swainson's Hawks was the smallest, followed closely by American Kestrels and Burrowing Owls (132 m, 149 m, and 151 m, respectively; Table 3.5). The Northern

Harrier and Ferruginous Hawk had slightly larger distances for the ESW (177 m and 190 m, respectively) whereas the Red-tailed Hawk had the greatest ESW distance (295 m; Table 3.5). In the same procedure, program DISTANCE calculated the density for each species for the entire survey period along with the upper and lower confidence limits (Table 3.6). I converted this value into the approximate density per month to account for a few factors. First, there was a possibility of repeat counts of the same individuals each month. Second, I wanted to account for species with high densities during short periods of time (e.g., Burrowing Owls) as opposed to species with relatively consistent densities over long periods of time (e.g., American Kestrels). To illustrate, the estimated density for Burrowing Owls for the survey year was 1.9 birds/km<sup>2</sup>, which is less than the American Kestrel density over the year (2.5 birds/km<sup>2</sup>; Table 3.6). However, the Burrowing Owl density per month of occurrence is higher than that of American Kestrels (0.238 birds/km<sup>2</sup> and 0.208 birds/km<sup>2</sup>, respectively; Table 3.6). Swainson's Hawks had the highest density in the Llano Estacado study area during its season of occurrence and per month (2.7 birds/km<sup>2</sup> and 0.386 birds/km<sup>2</sup>, respectively; Table 3.6). Red-tailed Hawks are abundant in the winter but they were detected in 11 months in this study; thus its density per month was lower than that of Northern Harriers (0.109 birds/km<sup>2</sup> and 0.111 birds/km<sup>2</sup>, respectively; Table 3.6), probably because Northern Harriers' period of occurrence was shorter.

The methods described above utilizing detection function models to estimate density and abundance yielded very different results than my calculations using conventional abundance index methods. Detection function models estimated species abundances of two to five times conventional raptor abundance index estimates (Table

3.7) to account for imperfect detection of birds away from the survey line. For example, I estimated an abundance of 0.139 kestrels/km per month using conventional sampling, but I estimated an abundance of 0.324 kestrels/km per month using the detection function methods (Table 3.7). Estimates of monthly species abundance during their season of occurrence for conventional estimates were 0.292 birds/km for Ferruginous Hawks, 0.351 birds/km for Northern Harriers, 0.584 birds/km for Burrowing Owls, 0.883 birds/km for Red-tailed Hawks, 1.091 birds/km for Swainson's Hawks, and 1.662 birds/km for American Kestrels (Table 3.7). In contrast, these values from detection-function methods were 0.932 birds/km for Ferruginous Hawks, 1.554 birds/km for Northern Harriers, 2.952 birds/km for Burrowing Owls, 1.865 birds/km for Red-tailed Hawks, 4.196 birds/km for Swainson's Hawks, and 3.885 birds/km for American Kestrels, illustrating large differences in abundance estimates between the two methods (Table 3.7).

For land-use selection analyses, sample sizes allowed for chi-squared analyses of five species: American Kestrels in all seasons; Swainson's Hawks in Spring, Summer, and Fall; Red-tailed Hawks in Fall, Winter, and Summer; and Ferruginous Hawks and Northern Harriers each in a Fall+Winter+Spring combined season of occurrence. There was apparent raptor selection across seasons and species: each species analyzed did not use the land-use classes in proportion to their availability on the landscape in many seasons (Table 3.8, 3.9). The data reveal a general pattern in which nongrazed grassland was used more than expected and grazed grassland slightly more than expected (Table 3.8). In contrast, textile agriculture was used substantially less expected based on availability across seasons, whereas grain agriculture use appeared to be season-specific (Table 3.8). Chi-squared test results demonstrate that certain species are particularly

selective in which land-use types they associate with (Red-tailed Hawks, Northern Harriers, Ferruginous Hawks; Table 3.8) whereas others only express selection in certain seasons (American Kestrel and Swainson's Hawk; Table 3.9).

American Kestrels exhibited statistically significant selection of certain land-use categories only in the winter ( $\chi^2=13.2$ ,  $df=4$ ,  $p=0.011$ ). Two patterns of land-use were consistent for this species across seasons, regardless of the significance of the test: they always utilized cotton less than it was available and always utilized the "other" category (which includes residential development and plots of trees) more than expected based on availability (Fig. 3.8.1). In winter and spring, American Kestrels utilized harvested grain fields at a greater proportion than available, then in spring and summer American Kestrels associated with nongrazed grassland in higher proportions than it was available (Fig. 3.8.1). Swainson's Hawks consistently used cotton in a much smaller proportion than it was available (Fig. 3.8.2). Unlike American Kestrels, Swainson's Hawks dependably used nongrazed grasslands in greater proportion than they were available (Fig. 3.8.2). There were no patterns evident for the other land-use categories. Swainson's Hawks showed a statistically significant difference between use and availability of land-use categories in spring ( $\chi^2=17.5$ ,  $df=4$ ,  $p=0.0015$ ) and summer ( $\chi^2=19.4$ ,  $df=4$ ,  $p=0.0007$ ). Red-tailed Hawks showed the same pattern as Swainson's Hawks, except more pronounced (Fig. 3.8.3). This species selected land-use types in a significantly different magnitude than they were available in all seasons (winter,  $\chi^2=38.4$ ,  $df=4$ ,  $p<0.0001$ ; spring,  $\chi^2=20.0$ ,  $df=4$ ,  $p=0.0005$ ; fall,  $\chi^2=18.1$ ,  $df=4$ ,  $p=0.0012$ ): they avoided grain crops and cotton crops, especially, while strongly selecting for nongrazed grassland (Fig. 3.8.3). The final two species analyzed, Ferruginous Hawks and Northern Harriers,

both demonstrated statistically different usage of land-use types than they were available in their season of occurrence ( $\chi^2=19.4$ ,  $df=4$ ,  $p=0.0007$  and  $\chi^2=10.5$ ,  $df=4$ ,  $p=0.033$ , respectively). They displayed the same patterns of association with land-use types (Fig. 3.8.4 and Fig. 3.8.5, respectively) with each other and with Red-tailed Hawks and Swainson's Hawks: a strong selection for nongrazed grassland patches in tandem with using cotton at a much lower proportion than would be expected. Ferruginous Hawks also appeared to avoid associating with grain agriculture (Fig. 3.8.4).

I categorized approximately 400 playas surrounding the survey route into “intact” and “modified” groups. About 190 of these playas were considered to be “intact” or not obviously modified; these were characterized by densely vegetated playa borders containing much native vegetation. Thus, at least 53% of playas in the study area have been obviously modified by agricultural activity, heavy grazing, or other apparent modifications. In every analysis except for one, the mean distance of observed raptors to the nearest playa edge was less than the mean distance of random points to the nearest playa (Table 3.10; Fig. 3.9). American kestrels show no association with intact playas but produced a nearly significant test result for all playas ( $W=0.10$ ,  $p=0.10$ ; Table 3.10). Northern Harriers showed a nearly significant result for mean distance comparisons for intact playas as well as all playas combined ( $t=-1.55$ ,  $df=65.1$ ,  $p=0.063$  and  $W=-583$ ,  $p=0.13$ , respectively; Table 3.10). Swainson's Hawks, interestingly, did not show a significantly lower distance to intact playas than random points had to intact playas but did have a significantly lower distance to all playas ( $W=3411$ ,  $p=0.0097$ ; Table 3.10). The strongest species association with intact playas appears to be that of Burrowing Owls: mean distance to intact playas equaled 377 m, whereas mean distance of random

points to the nearest playa equaled 782 m (Table 3.10; Fig. 3.9). Because of the nature of Burrowing Owl colonies, no significance tests were performed. Red-tailed Hawks and Ferruginous Hawks had the strongest associations with intact playas along the survey route, with significantly lower mean distances to the nearest intact playa than of random points ( $t=-4.15$ ,  $df=206.2$ ,  $p<0.0001$  and  $t=-1.98$ ,  $df=70.5$ ,  $p=0.026$ , respectively; Table 3.10; Fig. 3.9). Yet for both of these species, when the analysis was done on all playas inclusively, the means to the nearest playas were not significantly lower than the random point means to the nearest playa (Table 3.10). It seems that raptor associations with playas are species-specific and sometimes dependent on the characteristics of the playas in their proximity.

## DISCUSSION

Very little has changed in the seasonal occurrence of common species in the High Plains compared to earlier studies. Just as Stevenson (1942) reported, American Kestrels are a common resident of the region whereas Mississippi Kites and Swainson's Hawks are regular breeders. However, I did not detect as many Turkey Vultures in my survey as Stevenson reported. Common winter species he listed included Red-tailed Hawks, Ferruginous Hawks, and Northern Harriers, although Stevenson reported that Northern Harriers were the most abundant wintering raptors. In my study, Red-tailed Hawks and American Kestrels were the most abundant winter raptors detected. Stevenson (1942) also described that Prairie Falcons and Rough-legged Hawks were numerous in his study near Palo Duro Canyon; I detected very few individuals of these species in Lubbock and Hockley Counties. I detected one Merlin (*Falco columbarius*) in the winter, as well as a few Cooper's Hawks (*Accipiter cooperii*) and Sharp-shinned Hawks (*Accipiter striatus*)

in the fall and winter, in contrast to Stevenson's records of accipiters being summer residents. Results from my study were similar to Allan and Sime's (1943) hawk census in the Texas Panhandle. I noted many of the same yearly population trends: Northern Harrier abundances peak in the fall and spring, Turkey Vultures are only resident in summer and fall, Mississippi Kites are only present during their short breeding season, Ferruginous Hawks occupy the area in the winter and disappear in summer months. However, this region demonstrates some differences from historical reports in that I detected much higher abundances of American Kestrels year-round and an occupancy of the Swainson's Hawk prior to fall migration. Also, Allen and Sime (1943) reported their months of peak abundance in February and in August through October whereas I recorded the peak raptor abundance occurring in April, and then August through the winter in the Llano Estacado (Table 3.2; Fig. 3.5).

Compared to the Merriman et al.'s (2007) recent raptor point counts at prairie dog colonies in Lubbock County, Texas, I detected comparable levels of most raptor species. Merriman et al. (2007) detected a greater abundance of Ferruginous Hawks, probably because the surveys focused on a prey base of the hawk. Merriman et al. (2007) also detected fewer American Kestrels. Behney et al. (2012) detected the same suite of species in Lesser Prairie Chicken (*Tympanuchus pallidicinctus*) habitat areas in the SHP as in this study, except they detected occasional Golden Eagles in the winter (*Aquila chrysaetos*) but not American Kestrels at his site. The most abundant species detected by Behney et al. (2012) were Swainson's Hawks, Northern Harriers, and Red-tailed Hawks.

An influx of migrant species in spring and fall caused higher species richness, evenness, and diversity values in this study in those seasons. The high species richness in spring and fall may have had an influence on the respective evenness and diversity metrics (Fig. 3.4). The rarefaction curves generated by EstimateS (Colwell 2013) illustrated that the sampling effort of this study probably detected nearly all species that are present in the region, as evidenced by the plateau (Fig. 3.7).

The detection function models account for observer predisposition to miss detections of individuals as distance from the observer increases (Buckland et al. 2001), so that estimated densities can be applied to the rest of the study area or region. The difference in the effective strip width and model detection functions with series adjustments for each raptor species indicates dissimilarity in detection of raptor species. Millsap and LeFranc (1998) also concluded that a number of factors may cause variation in raptor counts, including differences in detectability across cover types, season, and distance of observations. These detection differences could be manifested in raptor behaviors such as the tendency to forage from perches (Widen 1994, Sheffield et al. 2001, Bobowski et al. 2014), or the tendency to associate with variable vegetation structure (Anderson et al. 1985, Preston 1990), and are probably worth incorporating into calculations. At the very least, I recommend that survey strips be narrowed to widths lower than historical recommendations (Craighead and Craighead 1969) and common practice (Bauer 1982, Anderson et al. 1985, Garner and Bednarz 2000) in order to improve the accuracy of detections within the strip. The largest effective search distance calculated in this study was 295 m for Red-tailed Hawks; therefore, it may be prudent to

limit search widths to 300 m or less in future raptor road surveys conducted in open grassland.

The species abundance estimates according to the detection function model versus the conventional relative abundance index show vast differences in this study. The conventional method calculates abundance as the number of raptors per a distance, and is useful because of its simplicity and ease of calculation based on raw survey counts. However the detection model function method incorporates observer imperfection into the calculations, which is very applicable to road surveys conducted in moving vehicles. Because power lines often parallel roads used for raptor survey transects and many raptor species select foraging locations based on perch site availability (Bildstein 1978, Widen 1994, Thiollay 1981), detection function methods may overestimate raptor abundances of certain species because of the presence of these poles. In reality, the true abundance for species that perch on utility lines is probably somewhere between both conventional and model estimates. If one can assess the effects of power pole presence using it as a covariate, this issue may be ameliorated and detection function models could be used without reservation. Yet even for Burrowing Owls and Northern Harriers that do not normally utilize power poles for perching, the detection models estimated four to five times higher abundances than the conventional abundance index methods (Table 3.7).

This study demonstrated that land-use is indeed applicable when assessing raptor habitat selection and, though coarse, can be used with success as a proxy for higher-quality foraging habitat. Raptors in the Llano Estacado exhibited species-specific and season-specific patterns of association with land-use types. Broad conclusions may be

drawn from two patterns that were displayed by all species in nearly every season: that cotton, no matter what stage of growth, is consistently selected against and that nongrazed grassland is usually selected in greater proportion than available. These patterns were strongest for Red-tailed Hawks, Swainson's Hawks, Northern Harriers, and Ferruginous Hawks. Two possible explanations for these patterns are their prey base and their accessibility to prey exhibited by different land-use types (Sheffield et al. 2001, Preston 1990). Nongrazed grasslands may harbor adequate rodent and insect populations, whereas cotton likely provides little nutritional value for rodents or insects and also has dense cover (and thus lower prey accessibility to raptors) in the growing season. The use of grain agriculture was season-specific. American Kestrels used it in greater proportion than its availability in winter and spring (when it was harvested) but Red-tailed Hawks and Ferruginous Hawks displayed an avoidance of grain agriculture in their occupancy season. Grain agriculture, when harvested, may provide grain residue as food for rodents and other prey. While the crop is standing, raptors may be less successful in detecting prey in the dense greenery. That American Kestrels used land-use categories in proportion to their availability in most seasons may be explained by their generalist strategy. They used the "other" category in a greater proportion than it was available, probably because American Kestrels commonly associate with urban and developed areas (Smallwood and Bird 2002).

As hypothesized, Ferruginous Hawks and Northern Harriers showed associations with playas in the study region. The statistically significant mean distance to nearest playa shown by both Red-tailed Hawks and Ferruginous Hawks (compared to random point distances to the nearest playa) did not hold true when all playas were assessed. This

finding strengthens the argument that “intact playas,” as I delineated them, may be meaningfully different from the “modified playas.” Prairie dogs, which associate with playas in the region (Pruett 2004), comprise a large proportion of prey biomass for Ferruginous Hawks (Giovanni et al. 2007), and this hawk's abundance is strongly linked to local availability of prairie dogs (Cully 1991). Interestingly, Ferruginous Hawks and Northern Harriers are only present during migration seasons and in great abundance in the winter; playas are usually dry at this time (Johnson et al. 2011). It may be that playas in a dry state as opposed to an inundated state are more beneficial to species that associate with playas than is currently recognized.

That American Kestrels and Swainson’s Hawks showed an association with all playas but not with “intact playas” was remarkable. Because the “all playas” category includes both “modified” and “intact” playas, and there was no statistically significant association with “intact” playas, it means that these two species are actually associating strongly with the “modified” playas. The American Kestrel is somewhat adapted to landscapes with more development, which may characterize the location where some “modified” playas occur in contrast to the “intact” ones. I suspect that the invertebrate community and small mammal community may differ between the two coarse types of playas. Swainson’s Hawks and American Kestrels are both generalists whose diets include a large proportion of invertebrates (Sherrod 1978; Giovanni et al. 2007). Many “modified” playas were defined as such due to alteration by cultivation or the reduction of a vegetation buffer around the playa. This changes the drainage pattern and water quality (Bolen et al. 1989) and could provide habitat for a different invertebrate community (as has been documented in Odonata by Reece and McIntyre 2009).

Potentially, the altered invertebrate community is more attractive to American Kestrels and Swainson's Hawks. Additionally, Pruett et al. (2010) found a variety of small mammals at playa sites; perhaps these communities are comprised differently at "intact" and "modified" sites such that prey at the latter sites are more attractive for American Kestrels and Swainson's Hawks.

Also as predicted, Burrowing Owls showed a strong association with both "intact" and all playas, though in this case one must be cautious of the result for two reasons. First, intact playas were included in the "all playas" dataset and the effects might be carried over. Second, Burrowing Owl observations are not independent of one another. It would be more appropriate to analyze one representative point for each colony and test that distance to the nearest playa, but we did not detect enough separate colonies to perform such an analysis. However, because Burrowing Owls are obligate nesters in prairie dog burrows, and prairie dogs are associated with playas on the SHP (Pruett 2004), it naturally follows that Burrowing Owls are also associated with playas.

The implications for wind energy planning in the Llano Estacado can be based on raptor land-use selection patterns. The assessment of the interactions of the three components -- wind energy, landscape characteristics, and raptors -- can be compared to known causes of increased turbine collision risk to mitigate those effects. One attribute of the Llano Estacado that could represent increased collision risk is its characteristic high springtime winds (Elliot 1977), which overlap with a migration season. I found that raptor species richness, evenness, and diversity of raptor species increase during migration seasons, although overall raptor abundance in spring was low (Table 3.3).

Additionally, each species analyzed in this region significantly selected land-use types in different proportions than they were available on the landscape (Table 3.9). In general, they associated more strongly with nongrazed grassland than expected and used cotton less than expected based on its availability (Figs. 3.8.1-3.8.5). From a wind energy development perspective, one could assign greater collision risk or displacement risk to the land-use types raptors associate with strongly. Conversely, siting wind farms within agricultural fields that include cotton in their crop rotation or managing the land-use parcels surrounding turbines with farming pursuits would conceivably decrease collision risk in this area.

Another landscape component that could increase risk of raptor collisions is the presence of playa wetlands. The playas occur at high densities and this research shows that many raptor species prone to collisions associate with playas. Depending on the species of conservation interest, more specific recommendations based on their proportional land-use categories and associations with playas will apply. I recommend that further research on the Llano Estacado pursue the reason behind the patterns of association with playas.

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## APPENDIX. DATA MANAGEMENT AND ARCGIS STEPS TO ASSESS RAPTOR USE VERSUS AVAILABILITY

Availability was assessed using an Estimated Strip Width (ESW) and Use was determined with a foraging buffer

Land-use category *use*:

1. Ground-truthed Common Land Unit files by adding a “Land-Use Category” field and updating land plots into categories.
2. Saved each land-use season as a separate geodatabase file.
3. Created ESW buffer by buffering the survey route by the ESW distance for the species of interest, dissolved the buffer (steps below apply to same species).
4. Overlaid raptor observation points onto seasonal land-use file of interest.
5. Clipped points to the ESW buffer (used these points in future steps).
6. Placed a 100 m buffer around each raptor point and did not dissolve these buffers.
7. Intersected the 100 m point buffers with the land-use file for the season of interest to obtain “use of land-use categories.”
8. In the attribute table for the intersected land-use layer, summarized “Land-use Category” by the sum of the “Area” field. This gives you the amount of land per land-use category for the season of interest.
9. Converted categorical land-use values to proportions.
10. Multiplied land-use category proportions by the sample size of the raptor in the season of interest to obtain frequency of use.

Land-use category *availability*:

1. Created a new ESW buffer that will be able to include the entire foraging buffer in the analysis; use same method as (3) but add 100 m to species’ ESW.
2. Clipped entire land-use file for season of interest by new ESW+100 m buffer to obtain “available land-use.”
3. In the attribute table for the intersected land-use layer, summarized “Land-use Category” by the sum of the “Area” field. This gives you the amount of land per land-use category for the season of interest.
4. Converted categorical land-use values to proportions.
5. Multiplied land-use category proportions by the sample size of the raptor in the season of interest to obtain frequency of use.

Table 3.1. Description of land-use types used to categorize all land parcels for the raptor use-versus-availability analysis.

Land-use type	Description
Cotton	<i>The most commonly farmed crop in the area. Two sub-categories: growing/standing and harvested</i>
Grain agriculture	<i>The other main crop in the area; e.g., Grain Sorghum, winter wheat also included. Two sub-categories: growing/standing and harvested</i>
Grazed grassland	<i>Determined by the presence of domestic grazers, an intact border fence, or obviously grazed vegetation</i>
Non-grazed grassland	<i>Obviously non-grazed vegetation; e.g., CRP grassland or open plots of native species</i>
Other	<i>Human structures or development, vertical woody vegetation, industrial land-use, fallow fields</i>

Table 3.2. Occurrence of raptor species per month in the Llano Estacado, Dec 2012 - November 2013 based on raw counts.

Species	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Total
Turkey Vulture ( <i>Cathartes aura</i> )					1	1	3		3	4	7		19
Northern Harrier ( <i>Circus cyaneus</i> )	6	3	5	5	11	2				4	9	9	54
Mississippi Kite ( <i>Ictinia mississippiensis</i> )						3	3	2	8				16
Sharp-shinned Hawk ( <i>Accipiter striatus</i> )	2										1		3
Cooper's Hawk ( <i>Accipiter cooperii</i> )	2										2	1	5
Unidentified Accipiter ( <i>Accipiter</i> spp.)						1							1
Harris's Hawk ( <i>Parabuteo unicinctus</i> )										1			1
Swainson's Hawk ( <i>Buteo swainsonii</i> )					36	25	15	17	30	35	10		168
Red-tailed Hawk ( <i>Buteo jamaicensis</i> )	23	18	17	21	2	4	1	1		2	12	35	136
Ferruginous Hawk ( <i>Buteo regalis</i> )	9	3	7	7	1						6	12	45
Rough-legged Hawk ( <i>Buteo lagopus</i> )				1									1
Unidentified Buteo ( <i>Buteo</i> spp.)	4	2	3	4	2		1		1			2	19
Merlin ( <i>Falco columbarius</i> )				1									1
American Kestrel ( <i>Falco sparverius</i> )	32	10	17	9	16	10	13	29	32	30	36	22	256

Table 3.2, Continued

Species	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Total
Prairie Falcon ( <i>Falco mexicanus</i> )		1	1								2	1	5
Great Horned Owl ( <i>Bubo virginianus</i> )								2					2
Burrowing Owl ( <i>Athene cunicularia</i> )				3	1	4	4	24	30	18	6		90
Unidentified Owl							1						1
Unidentified Raptor		1	1			2			2	1		1	8
Total monthly detections	78	38	51	51	70	52	41	75	106	95	91	83	831
No. species	6	5	5	7	7	7	6	6	5	7	10	6	15

Table 3.3. Seasonal species detections (n) and relative abundances from raw counts from raptor surveys.

Species	Winter n	Winter rel. abund	Spring n	Spring rel. abund.	Summer n	Summer rel. abund.	Fall n	Fall rel. abund.	Total n per species	Overall rel. abund.
Turkey Vulture	0	0.0	2	1.2	6	2.7	11	4.1	19	2.3
Northern Harrier	14	8.4	18	10.4	0	0.0	22	8.2	54	6.5
Mississippi Kite	0	0.0	3	1.7	13	5.9	0	0.0	16	1.9
Sharp-shinned Hawk	2	1.2	0	0.0	0	0.0	1	0.4	3	0.4
Cooper's Hawk	2	1.2	0	0.0	0	0.0	3	1.1	5	0.6
Unidentified Accipiter	0	0.0	1	0.6	0	0.0	0	0.0	1	0.1
Harris's Hawk	0	0.0	0	0.0	0	0.0	1	0.4	1	0.1
Swainson's Hawk	0	0.0	61	35.3	62	27.9	45	16.7	168	20.2
Red-tailed Hawk	58	34.7	27	15.6	2	0.9	49	18.2	136	16.4
Ferruginous Hawk	19	11.4	8	4.6	0	0.0	18	6.7	45	5.4
Rough-legged Hawk	0	0.0	1	0.6	0	0.0	0	0.0	1	0.1
Unidentified Buteo	9	5.4	6	3.5	2	0.9	2	0.7	19	2.3
Merlin	0	0.0	1	0.6	0	0.0	0	0.0	1	0.1
American Kestrel	59	35.3	35	20.2	74	33.3	88	32.7	256	30.8
Prairie Falcon	2	1.2	0	0.0	0	0.0	3	1.1	5	0.6
Great Horned Owl	0	0.0	0	0.0	2	0.9	0	0.0	2	0.2
Burrowing Owl	0	0.0	8	4.6	58	26.1	24	8.9	90	10.8
Unidentified Owl	0	0.0	0	0.0	1	0.5	0	0.0	1	0.1
Unidentified Raptor	2	1.2	2	1.2	2	0.9	2	0.7	8	1.0
Season total	167		173		222		269		831	
No. species	7		10		7		10		15	

Table 3.4. Diversity metrics per season of raptor surveys, including species richness (S), evenness (E), Shannon-Wiener diversity index (H'), and Gini-Simpson diversity index (1-D) per season.

Season	S	E	H'	(1-D)
Winter	7	0.707	1.38	0.695
Spring	10	0.747	1.72	0.772
Summer	7	0.736	1.43	0.726
Fall	10	0.809	1.86	0.805

Table 3.5. Raptor observations used in analysis after truncation using the effective strip width.

Species	Observations included	Detection model adjustment	Effective strip width (m)	Probability of detection
American Kestrel	218	Hazard Rate cosine	149	0.29
Swainson's Hawk	108	Hazard Rate simple polynomial	132	0.19
Burrowing Owl	38	Uniform cosine	151	0.43
Red-tailed Hawk	111	Uniform cosine	295	0.37
Ferruginous Hawk	36	Half-normal cosine	190	0.32
Northern Harrier	37	Hazard Rate simple polynomial	177	0.24

Table 3.6. Density estimates with lower and upper confidence intervals per species. Some species are only present seasonally, described in Period of occupancy and number of months. Density values were calculated for the entire occupancy period and per month (only months were included when the species was present).

Species	Period of occupancy	No. months	Density/km <sup>2</sup> /			
			Period of occupancy	Density lower CI	Density upper CI	Density/km <sup>2</sup> / month
American Kestrel	Jan - Dec	12	2.500	2.000	3.100	0.208
Swainson's Hawk	April - Oct	7	2.700	1.100	6.400	0.386
Burrowing Owl	Mar - Oct	8	1.900	1.500	2.500	0.238
Red-tailed Hawk	Oct - Mar	11	1.200	1.000	1.400	0.109
Ferruginous Hawk	Oct - Mar	7	0.600	0.400	0.900	0.086
Northern Harrier	Sept - Apr	9	1.000	0.700	1.400	0.111

Table 3.7. Comparison of two methods of raptor abundance estimation: conventional abundance index methods (in italics) and detection function model abundance estimates. Calculations were based on the period of occupancy number of months detected (Tab. 3.6) per species. Density values per month assume that the raptor is present in that month.

Species	raptors/ km of survey/year		raptors/ km of survey/year		raptors/survey area/year		raptors/survey area/month	
	<i>1.7</i>	3.9	<i>0.1</i>	0.3	<i>256.0</i>	598.3	<i>21.3</i>	49.9
American Kestrel	<i>1.7</i>	3.9	<i>0.1</i>	0.3	<i>256.0</i>	598.3	<i>21.3</i>	49.9
Swainson's Hawk	<i>1.1</i>	4.2	<i>0.2</i>	0.6	<i>168.0</i>	646.1	<i>24.0</i>	92.3
Burrowing Owl	<i>0.6</i>	3.0	<i>0.1</i>	0.4	<i>90.0</i>	454.7	<i>11.3</i>	56.8
Red-tailed Hawk	<i>0.9</i>	1.9	<i>0.1</i>	0.2	<i>136.0</i>	287.2	<i>12.4</i>	26.1
Ferruginous Hawk	<i>0.3</i>	0.9	<i>0.04</i>	0.1	<i>45.0</i>	143.6	<i>6.4</i>	20.5
Northern Harrier	<i>0.4</i>	1.6	<i>0.04</i>	0.2	<i>54.0</i>	239.3	<i>6.0</i>	26.6

Table 3.8. Proportion of land-use categories available and used by each species within their respective effective strip width. Results are separated by season. Values in land-use category columns represent the percent of survey area per category (Available) or the percent of summed foraging buffers (Used). Four-letter species codes can be found in Pyle and DeSante (2012).

Season	Species	Used or Available	n	Growing Grain	Growing Grain*	Growing Cotton	Harvested Cotton	Harvested Cotton*	Grazed Grassland	Nongrazed Grassland	Other
Winter	AMKE	Used	53		19.2		45.2		6.7	14.4	14.4
Winter	AMKE	Available	53		13.6		48.2		7.5	25.5	5.1
Winter	RTHA	Used	50		11.4		9.8		13.1	52.5	13.1
Winter	RTHA	Available	50		13.0		47.5		7.7	26.0	5.8
Spring	AMKE	Used	30		17.9		27.6		10.0	32.8	11.7
Spring	AMKE	Available	30		13.6		46.1		8.2	25.7	6.3
Spring	RTHA	Used	24		6.7		11.8		24.2	47.3	10.1
Spring	RTHA	Available	24		14.3		46.7		8.1	25.3	5.6
Spring	SWHA	Used	40		21.4		15.7		14.4	37.1	11.4
Spring	SWHA	Available	40		13.5		46.0		8.2	25.8	6.5
Summer	AMKE	Used	58	25.2		11.1			11.2	35.6	16.9
Summer	AMKE	Available	58	30.2		24.5			8.2	25.7	11.4
Summer	SWHA	Used	39	18.7		6.6			9.8	53.5	11.4
Summer	SWHA	Available	39	30.2		24.3			8.2	25.7	11.5
Summer	BUOW	Used	44	1.0		7.0			55.8	33.7	2.5
Summer	BUOW	Available	44	30.2		24.5			8.2	25.7	11.4
Fall	AMKE	Used	77	27.3		21.1			13.9	23.2	14.6
Fall	AMKE	Available	77	31.4		24.1			7.7	25.6	11.1
Fall	SWHA	Used	29	29.8		18.2			7.4	35.2	9.4
Fall	SWHA	Available	29	31.5		24.0			7.7	25.7	11.2
Fall	RTHA	Used	37	17.6		6.1			13.4	49.6	13.4
Fall	RTHA	Available	37	31.3		25.1			7.8	25.2	10.7
Combined**	NOHA	Used	37		20.7			19.6	5.9	43.0	10.7

Table 3.8, Continued

Season	Species	Used or Available	n	Growing Grain	Growing Grain*	Growing Cotton	Harvested Cotton	Cotton*	Grazed Grassland	Nongrazed Grassland	Other
Combined**	NOHA	Available	37		19.7			39.6	7.9	25.7	7.1
Combined**	FEHA	Used	36		6.4			24.5	4.9	56.3	7.8
Combined**	FEHA	Available	36		19.8			39.7	7.9	25.7	7.0

\*both growing and harvested "phase" included here

\*\*Ferruginous Hawks and Northern Harriers observations were pooled into a combined "season of occurrence."

Table 3.9. Results of chi-squared tests comparing observed and expected frequencies of raptor land-use categories. Bold font indicates statistical significance which was set at  $\alpha = 0.05$ .

Species	<u>Winter</u>			<u>Spring</u>			<u>Summer</u>			<u>Fall</u>			<u>Winter+Spring+Fall</u>		
	n	$\chi^2_4$	P	n	$\chi^2_4$	P	n	$\chi^2_4$	P	n	$\chi^2_4$	P	n	$\chi^2_4$	P
American Kestrel	53	13.2	0.011	30	5.8	0.21	58	8.7	0.069	77	5.4	0.25			
Swainson's Hawk				40	17.5	0.0015	39	19.4	0.0007	29	1.1	0.90			
Red-tailed Hawk	50	38.4	<0.0001	24	20.0	0.0005				37	18.1	0.0012			
Ferruginous Hawk													36	19.1	0.0007
Northern Harrier													37	10.5	0.033

Table 3.10. Results of analyses comparing distances of observed raptor locations to the nearest playa with distances of random points to the nearest playas. I used student's t-tests for some analyses and Wilcoxon rank sum tests for data that violated the t-test assumption of normality. Statistical significance was set at  $\alpha = 0.05$ . Four-letter species codes can be found in Pyle and DeSante (2012).

Season of occurrence	Species	Playa type	n	Mean distance observed	Mean distance random		Test statistic:			
				(m)	95% CI	(m)	95% CI	t or (W)	df	P
Winter + Spring + Summer + Fall	AMKE	Intact	218	772	703 - 841	750	681 - 819	(24565)	-	0.73
		All	218	420	385 - 455	446	411 - 481	(22094)	-	0.10
Winter + Spring + Fall	RTHA	Intact	111	566	477 - 654	776	690 - 862	-4.15	206.2	<0.0001
		All	111	416	353 - 480	465	409 - 522	(5418)	-	0.060
	NOHA	Intact	37	661	491 - 830	777	628 - 927	-1.55	65.1	0.063
		All	37	373	287 - 458	449	372 - 527	-583	-	0.13
	FEHA	Intact	36	586	447 - 725	777	626 - 929	-1.98	70.5	0.026
		All	36	436	348 - 525	450	371 - 528	-0.43	46.7	0.33
Spring + Summer + Fall	SWHA	Intact	108	713	604 - 822	710	623 - 797	-0.48	181.7	0.31
		All	108	396	345 - 447	494	438 - 550	(3411)	-	0.0097
Summer	BUOW*	Intact	38	377	299 - 455	782	703 - 861			
		All	38	351	301 - 402	493	442 - 544			

\*validates assumption of independent samples because they nest in colonies; did not perform a statistical test

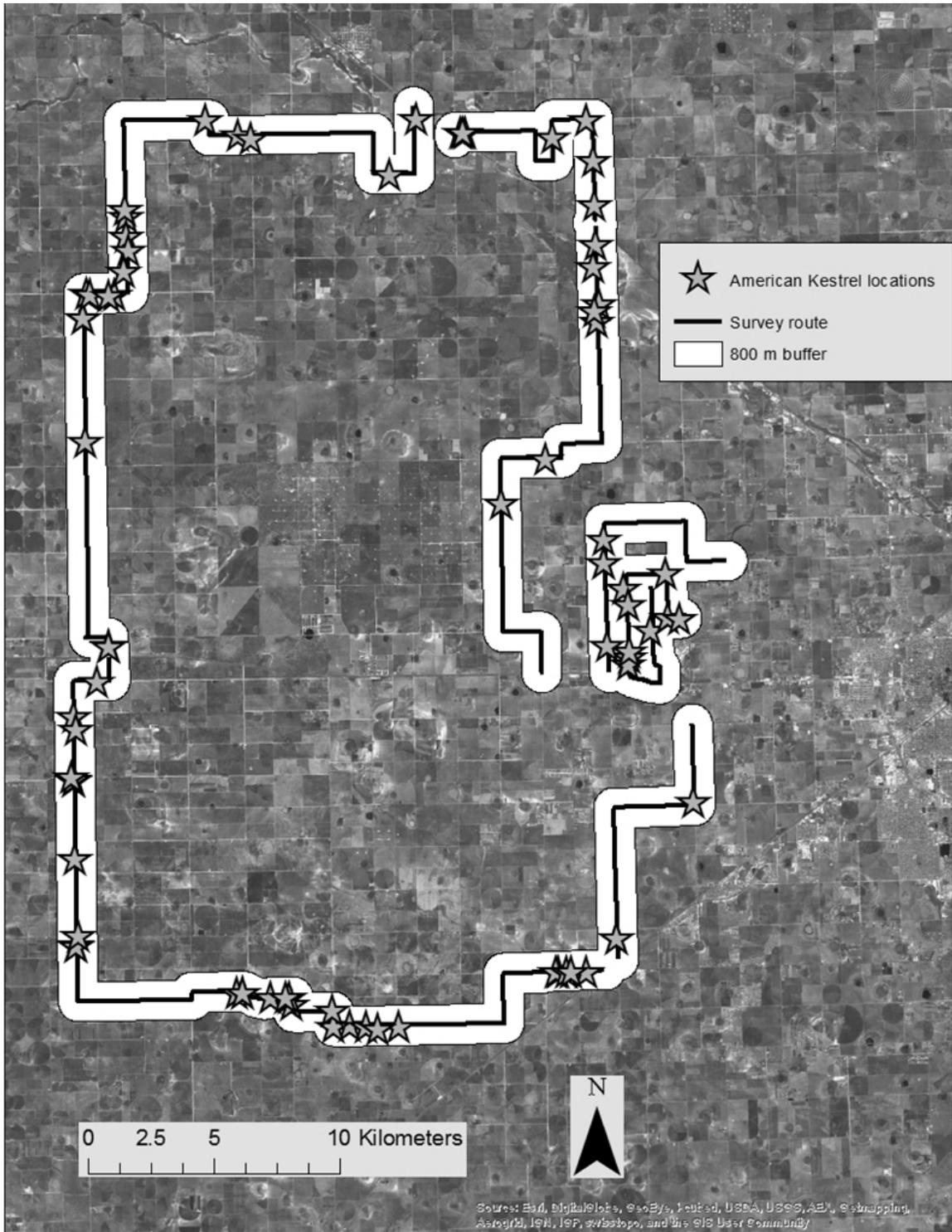


Figure 3.1: Observed American Kestrel locations along the 1600 m wide survey transect in the fall of 2013.



Figure 3.2: Clipped land-use layer (fall 2013) based on Effective Strip Width for American Kestrels: the faded land-use region represents the 1600 m wide ground-truthed survey area (see Fig. 3.1), whereas the bright land-use strip in the center of the survey route represent the clipped layer of land-use available to raptors used in analysis.



Figure 3.3. Demonstration of 100 m “foraging buffers” for assessing a raptor's use of different land-use types. The faded land-use region represents the Effective Strip Width for American Kestrels (see Fig. 3.2), whereas the bright land-use circles are the clipped 100 m foraging buffer that represents the area of raptor use for analysis.

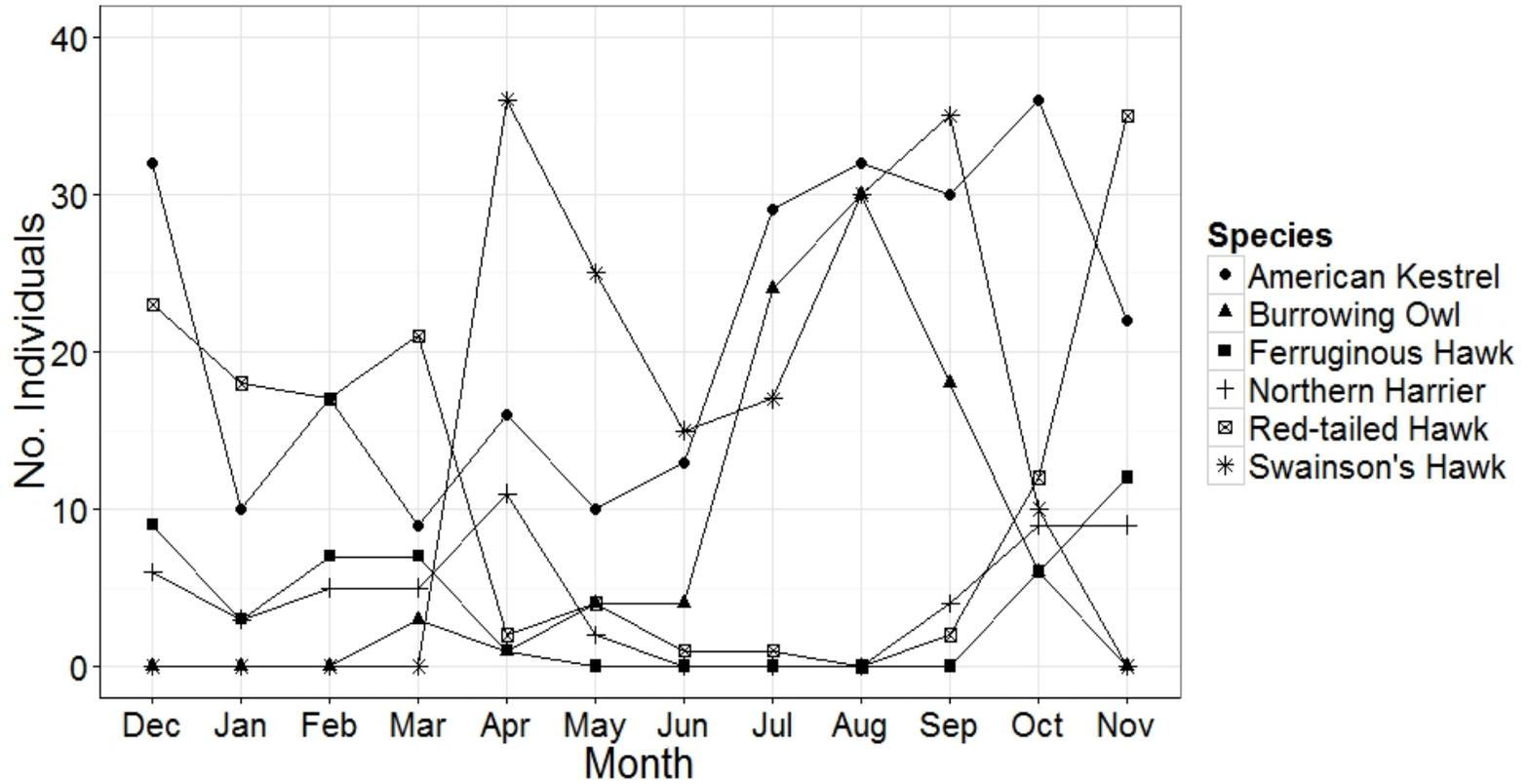


Figure 3.4. Raw abundance (see y-axis) trends per month for the six most common raptor species based on detections from raptor surveys.

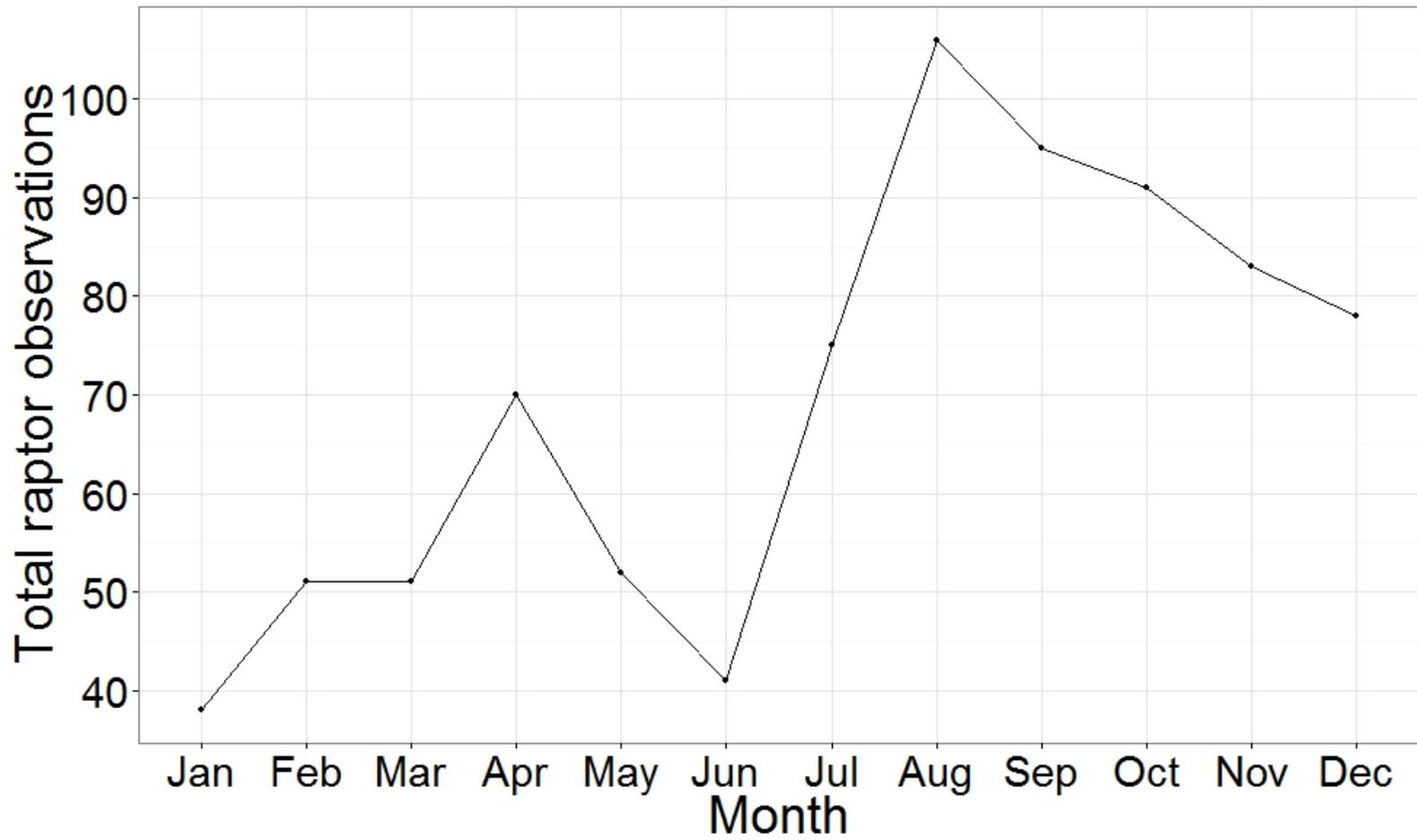


Figure 3.5. Total raptor abundance per month based on raw counts from raptor surveys.

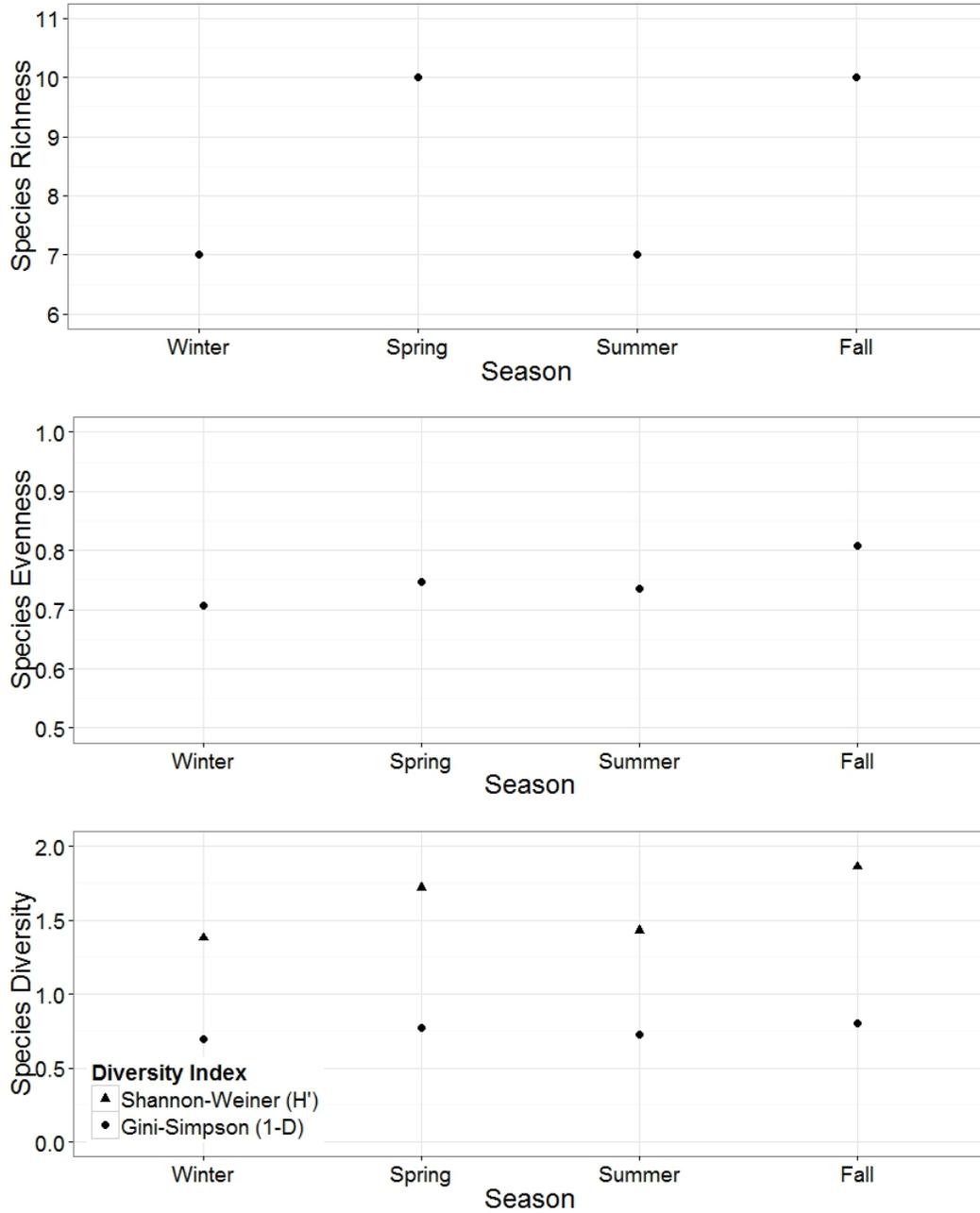


Figure 3.6. Species richness, evenness, and diversity of all raptors detected along the survey route per season.

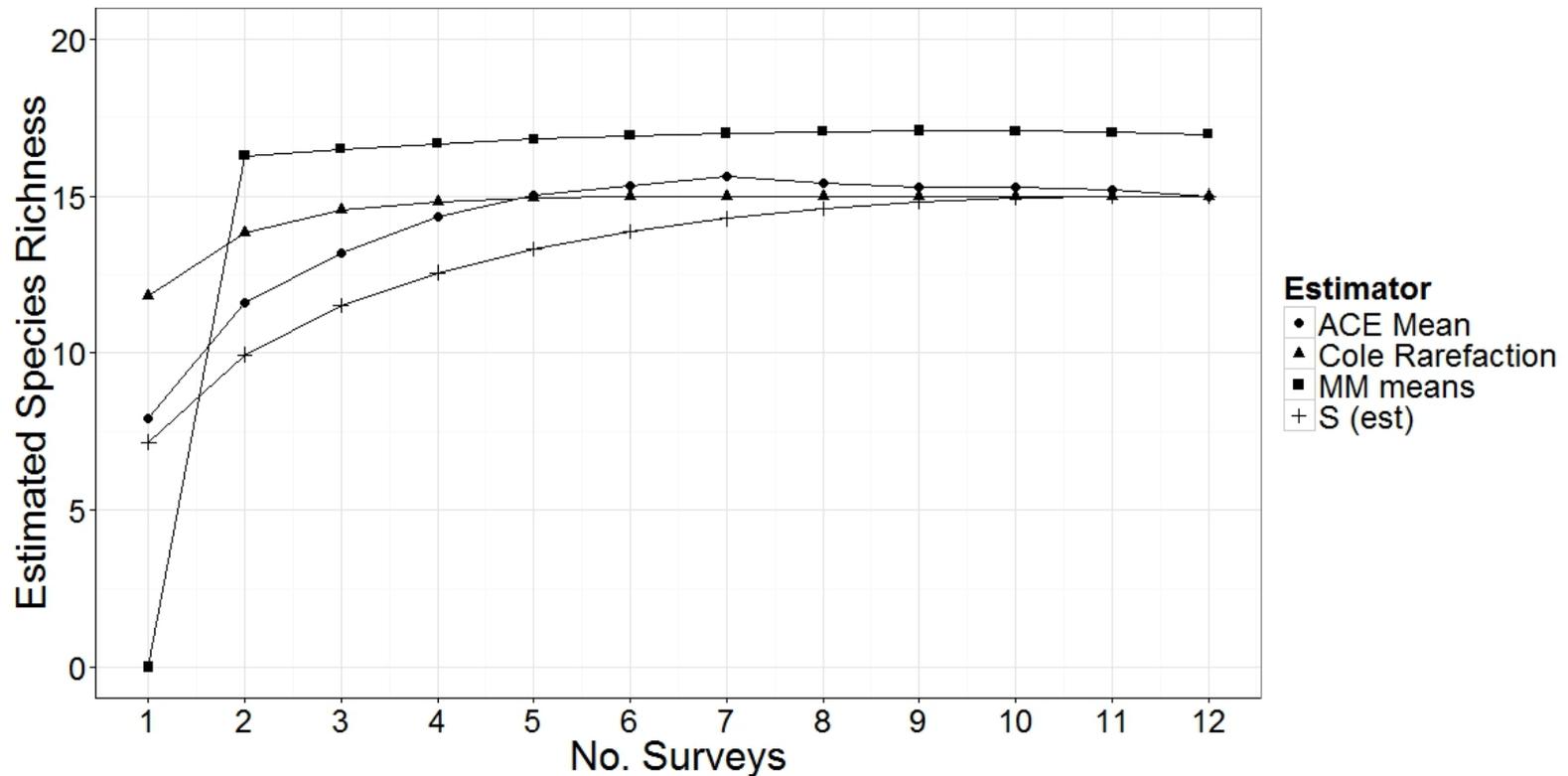


Figure 3.7. Raptor species accumulation curves calculated by EstimateS (Coleman 2013). The estimated species richness (y axis) increases with more survey samples (x axis). Each survey (each month) was equal to one sample. The rate of estimated accumulation of new species depends on the richness estimator (legend). The ACE Mean is the abundance coverage-based estimator (mean among runs), Cole Rarefaction represents the Coleman rarefaction estimator (number of species expected in pooled samples, assuming individuals distributed at random among samples), MM means is the Michaelis-Menten richness estimator where the estimators are averaged over randomizations, S (est) is the expected number of species in pooled samples, given the reference sample (analytical) (Colwell 2013). The species accumulation curve has reached an asymptote.

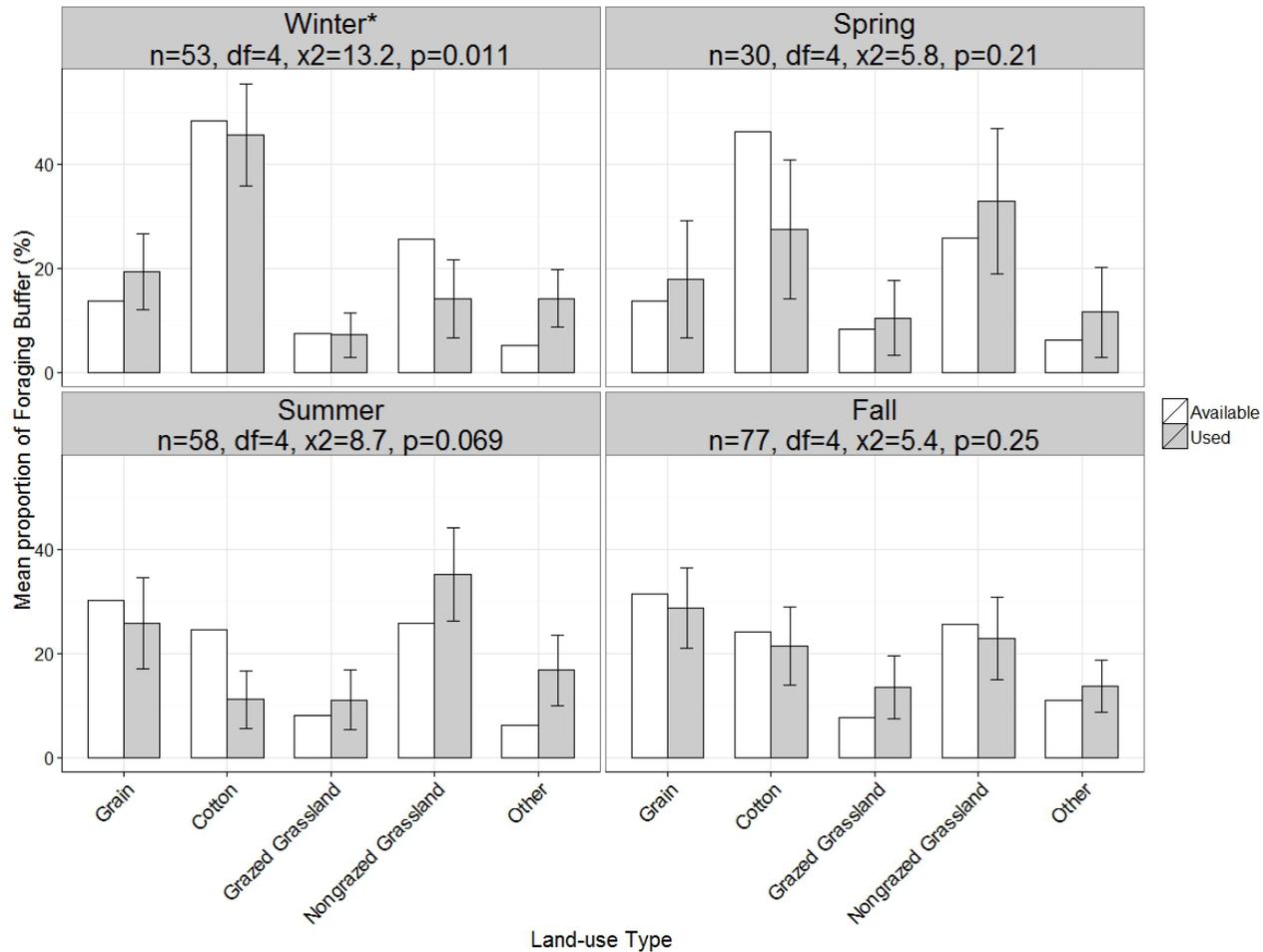


Figure 3.8.1. Proportional use versus availability of land-use categories per season by American Kestrels. Different land-use types are listed on the x-axis, the mean proportion used versus available land-use types within the 100 m “foraging buffer” are shown. In winter and spring, “Grain” and “Cotton” categories represent both growing and harvested agriculture (e.g., growing winter wheat and harvested sorghum stubble included). In summer and fall, “Grain” and “Cotton” represent growing

agriculture only. Sample size is listed under each season. Asterisk indicates a statistically significant difference at the  $\alpha = 0.05$  level between use and availability of land-use types from a chi-squared test. Error bars represent 95% confidence intervals.

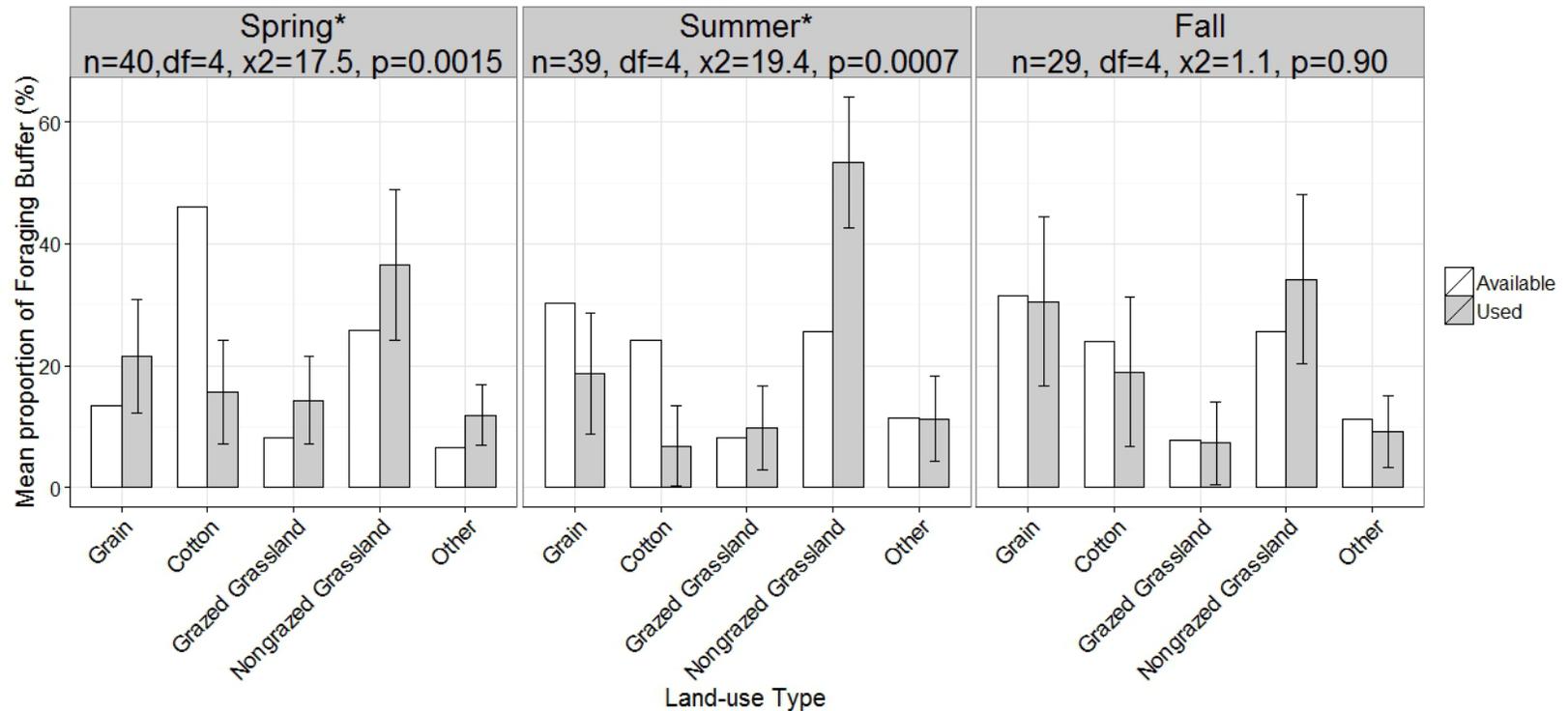


Figure 3.8.2. Proportional use versus availability of land-use categories per season by Swainson's Hawks. Different land-use types are listed on the x-axis, the mean proportion used versus available land-use types within the 100 m "foraging buffer" are shown. In spring, "Grain" and "Cotton" categories represent both growing and harvested agriculture (e.g., growing winter wheat and harvested sorghum stubble included), while "Grain" and "Cotton" in summer represent growing agriculture only. Sample size is listed under season. Asterisk indicates a statistically significant difference at the  $\alpha = 0.05$  level between use and availability of land-use types from a chi-squared test. Error bars represent 95% confidence intervals.

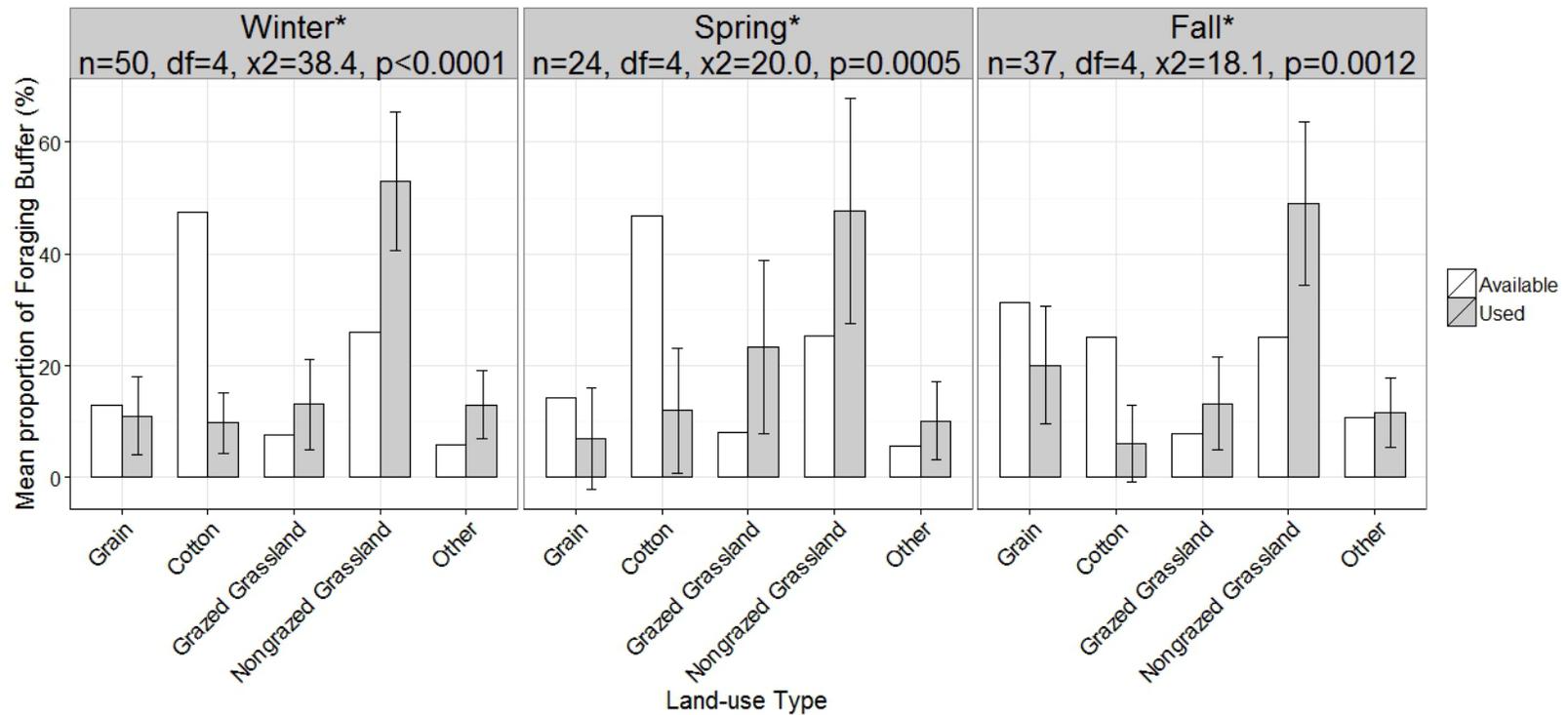


Figure 3.8.3. Proportional use versus availability of land-use categories per season by Red-tailed Hawks. Different land-use types are listed on the x-axis, the mean proportion used versus available land-use types within the 100 m “foraging buffer” are shown. In winter and spring, “Grain” and “Cotton” categories represent both growing and harvested agriculture (e.g., growing winter wheat and harvested sorghum stubble included). In summer and fall, “Grain” and “Cotton” represent growing agriculture only. Sample size is listed under each season. Asterisk indicates a statistically significant difference at the  $\alpha = 0.05$  level between use and availability of land-use types from a chi-squared test. Error bars represent 95% confidence intervals.

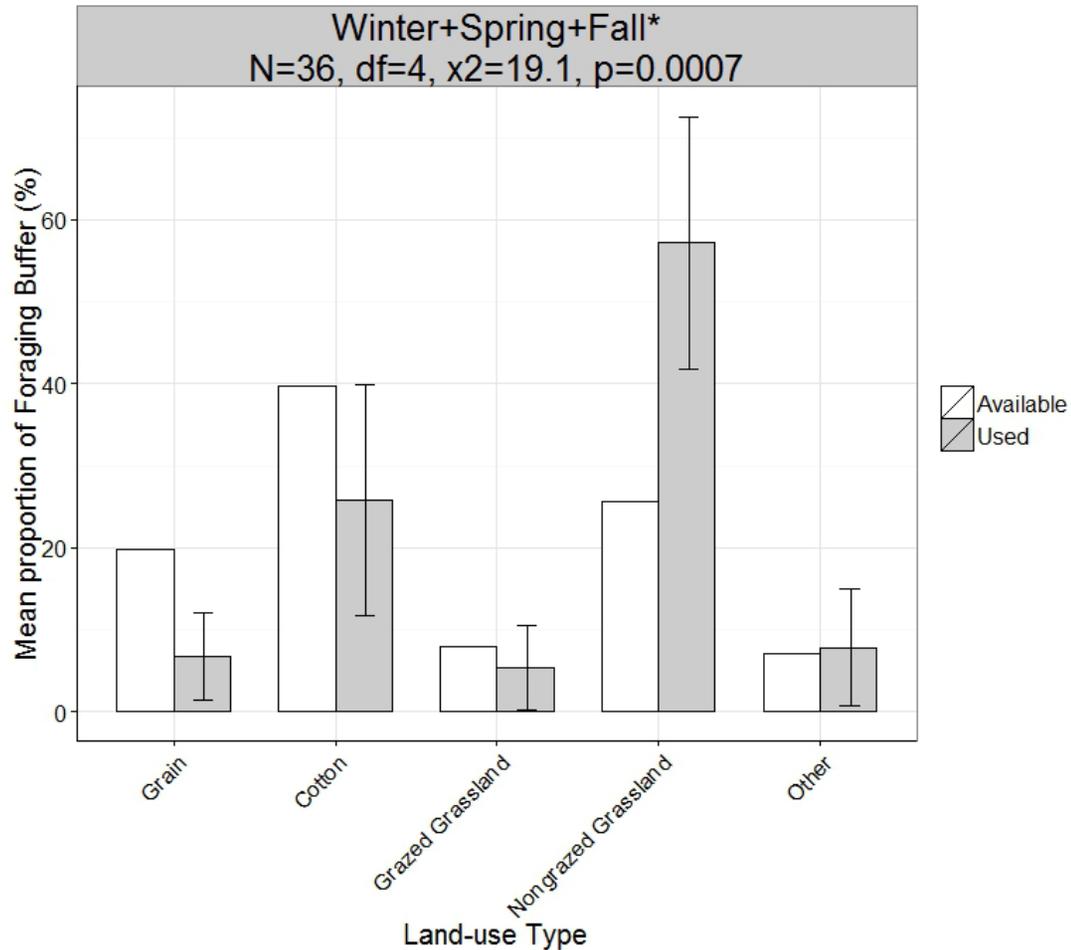


Figure 3.8.4. Proportional use versus availability of land-use categories per season by Ferruginous Hawks. Winter, spring, and fall counts were pooled because of a low sample size. Different land-use types are listed on the x-axis, the mean proportion used versus available land-use types within the 100 m “foraging buffer” are shown. “Grain” and “Cotton” land-uses represent growing and harvested agriculture. Sample size is listed under season. Asterisk indicates a statistically significant difference at the  $\alpha = 0.05$  level between use and availability of land-use types from a chi-squared test. Error bars represent 95% confidence intervals.

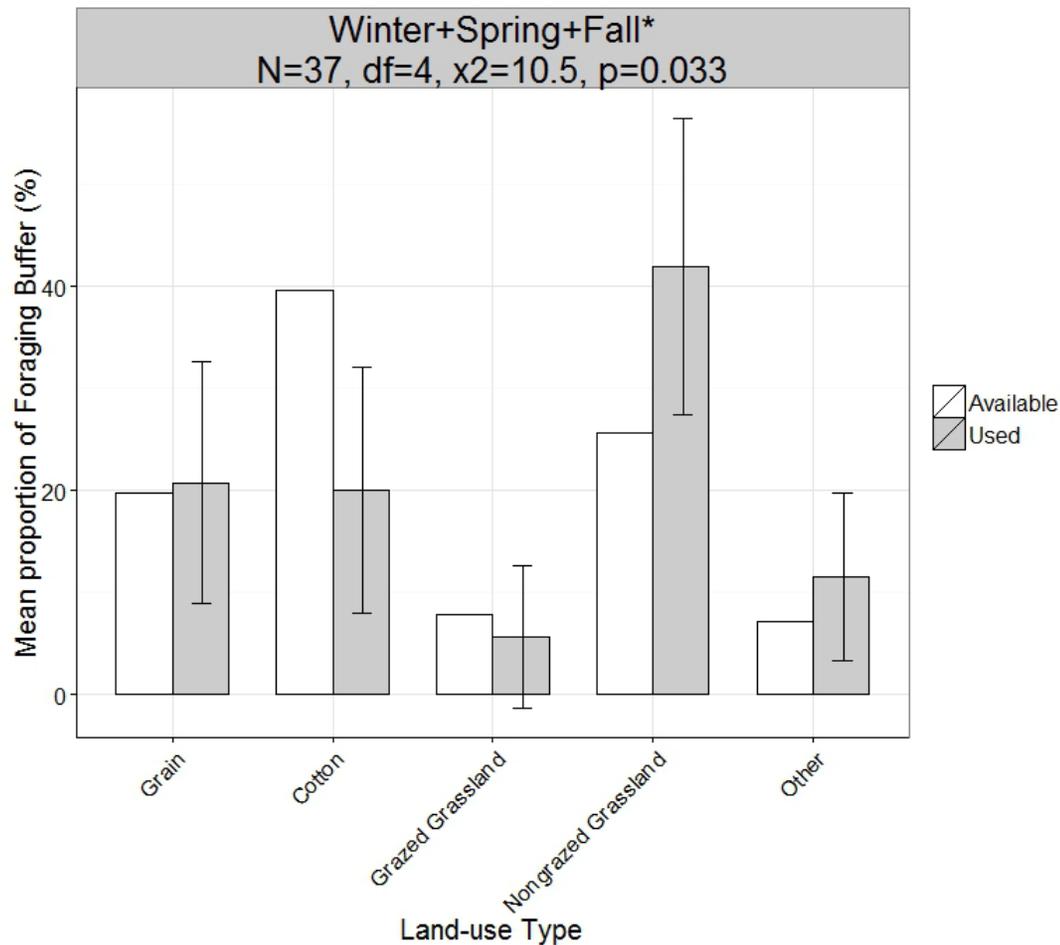


Figure 3.8.5. Proportional use versus availability of land-use categories per season by Northern Harriers. Winter, spring, and fall counts were pooled because of a low sample size. Different land-use types are listed on the x-axis, the mean proportion used versus available land-use types within the 100 m “foraging buffer” are shown. “Grain” and “Cotton” land-uses represent growing and harvested agriculture. Sample size is listed under season. Asterisk indicates a statistically significant difference at the  $\alpha = 0.05$  level between use and availability of land-use types from a chi-squared test. Error bars represent 95% confidence intervals.

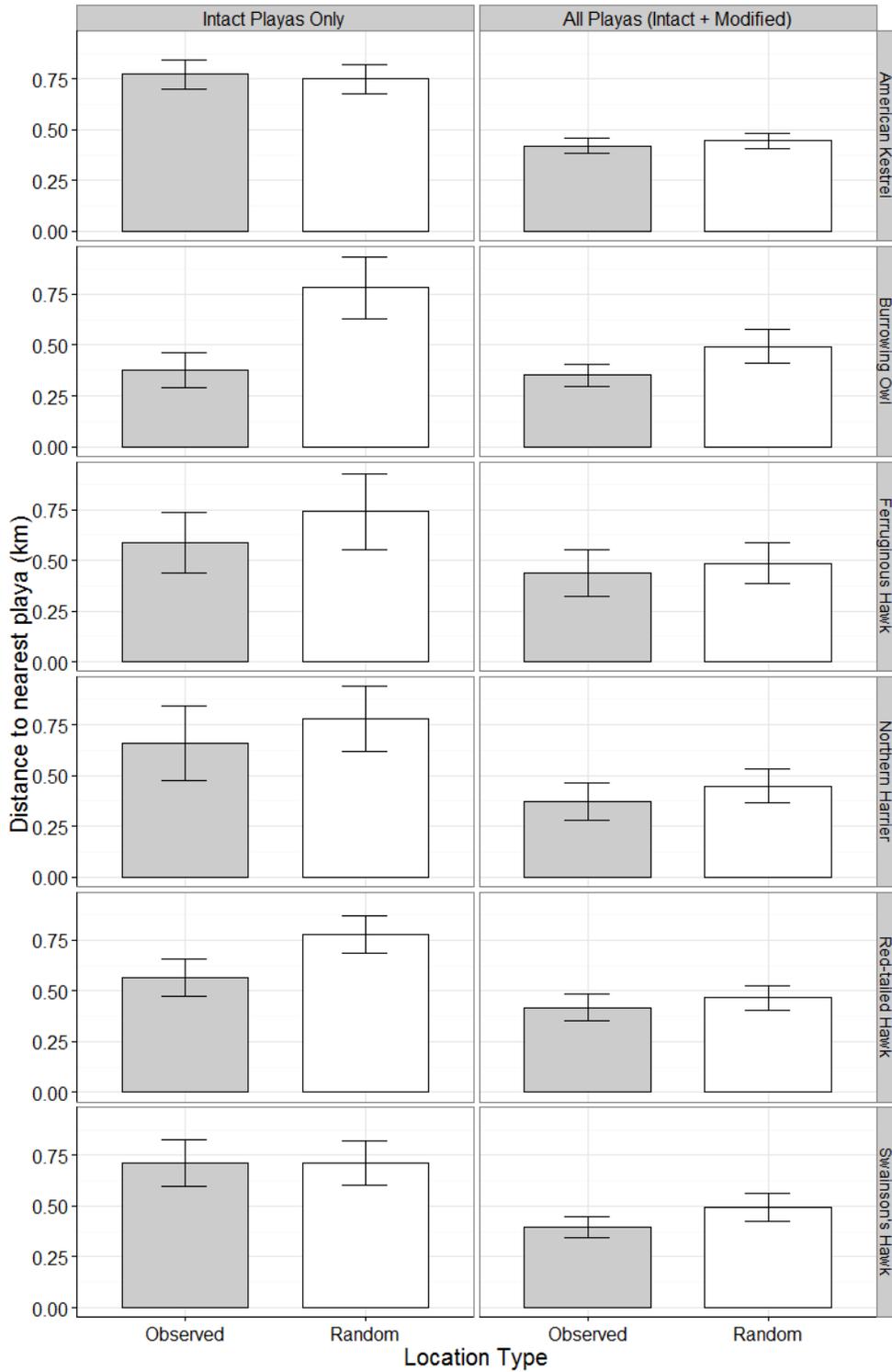


Figure 3.9. Comparison of mean distance (y-axis) and 95% confidence intervals from observed raptor locations and random points (x-axis) to nearest playa. Analysis was done separately per species (right panel labels) and type of playa (top panel labels).

## CHAPTER IV

### AMERICAN KESTREL BREEDING ECOLOGY IN THE LLANO ESTACADO INTRODUCTION

American Kestrels (*Falco sparverius*; hereafter kestrels) are one of the most abundant diurnal raptors in North America (Smallwood and Bird 2002), however, they are experiencing significant range-wide population declines. Three illustrative data sets demonstrate this negative trend. First, a synthesis of eight long-term nest-box monitoring programs revealed significant kestrel declines (Smallwood et al. 2009a). Second, regional seasonal migration counts show a long-term decline in the Northeast and more recent declines in the Midwest and Western United States (Farmer and Smith 2009). Third, yearly breeding bird survey trends across Bird Conservation Regions indicate that kestrels are significantly declining in 13 regions, while only two show a significant increase (AKP 2012a). Many hypotheses (or a combination thereof) are proposed as to the long-term population decline, such as increased Coopers Hawk predation (Farmer et al. 2008), poisoning from various pesticides (Wiemeyer and Sparling 1991, Hunt et al. 1991), climate change (Steenhof and Peterson 2009; Heath et al. 2012), the rise of West Nile Virus (Medica and Bildstein 2009), and land-use changes (Farmer and Smith 2009). Kestrel collisions with conspicuous structures such as wind turbines and associated infrastructure (Erickson et al. 2001) or aircraft (Dolbeer and Wright 2008; Garland et al. 2009) are not proposed as a cause for population decline but could be a compounding factor in areas of high kestrel density and turbine activity.

Despite rangewide declines, kestrels are increasing in the Southern High Plains of Texas (Bird Conservation Region 18, AKP 2012a) and the cause behind the trend is

unknown. Thus, additional research on kestrels in the area is warranted. Kestrels are common nationwide (Smallwood and Bird 2002) and the Llano Estacado, a high tableland within the Southern High Plains, is no exception: kestrels nest in the region in the summer, migrants move through in the fall and spring, and a kestrel population resides there in the winter (Allan and Sime 1943; Merriman et al. 2007). My recent raptor surveys (see Chapter III) confirm that there is a relatively high density of kestrels year-round.

Kestrels in the Llano Estacado especially face potential threats from increasing wind energy development. The Llano Estacado of Texas possesses some of the state's greatest expanses of high winds (Combs 2008) in addition to a flat agricultural-grassland community that is prime for wind energy development and has landscape characteristics that are attractive to kestrels (e.g., open grassland; Smallwood and Bird 2002). In fact, Texas is the leading state in installed wind energy capacity and construction (AWEA 2014).

Compared to other avian species, and even among raptors, kestrels are especially susceptible to impacts from wind turbines. A summary of existing wind turbine studies revealed that kestrels are one of the most commonly observed birds that use and experience mortality at wind energy sites (Erickson et al. 2001; Thelander et al. 2003). Smallwood et al. (2009b) showed that, of birds that fly through the rotor-swept zone, kestrels have a comparatively high collision rate with rotor blades. Certain behaviors may be to blame. For example, kestrels often fly at heights that are within rotor-swept zones (Wulff 2010, Smallwood et al. 2009b, Garvin et al. 2011) or fly near to operating turbines

(Thelander et al. 2003, Smallwood et al. 2009b). Foraging patterns may play a role; a study of kestrel collisions with aircraft showed collisions peak during periods in the day that corresponded with active foraging periods near the end of the breeding season (Garland et al. 2009). Late in the breeding season is also the post-fledging period for young kestrels. Thus, kestrels breeding near sources of collision may place their fledglings at risk. Even physiologically, kestrels were found to have lower visual acuity than previously assumed of diurnal raptors (McIsaac 2001). Raptors' frontal visual field is not in high resolution, and using lateral vision to detect conspecifics or prey may be more important (Martin 2011) meaning they may not detect seemingly conspicuous objects in their flight path like turbines.

Indirect impacts are also documented. Kestrels were one raptor species that decreased in abundance the most from pre- to post-construction of a wind farm, representing a possible displacement effect (Garvin et al. 2011). In contrast, kestrels could be caught in an ecological trap if they use wind turbine construction areas in that they could introduce new dangers while mirroring quality habitat cues. Kestrels typically inhabit open or semi-open landscapes with short vegetation cover (Smallwood and Bird 2002), especially less than 25 cm in height (Smallwood 1987; Ardia and Bildstein 1997). In nature, these habitat conditions would be represented by grazed grasslands or habitat edges where disturbance causes short vegetation. If perches are available, they select vegetation around 5 cm in height rather than vegetation over 20 cm where perches are absent (Sheffield et al. 2001). They likely prefer short vegetation structure because it facilitates the ability to locate prey (Bildstein and Collopy 1987). Kestrels frequently forage from perches, which is the least energetically costly method compared to flight-

hunting and hover-hunting (Collopy and Koplín 1983). Natural perch sites would include shrubs, trees, and snags. Human development of wild lands has introduced buildings, roads, poles, fence posts, and land-use practices that alter the vegetation structure; wind turbine development areas are no exception: disturbed wind energy construction sites often have short vegetation and available perches.

When wind turbine projects initiate, the habitat can be altered in many ways (National Research Council 2007) as areas of land are cleared of vegetation and utility poles are often constructed. As the vegetation grows back and is maintained at short heights and utility lines are built, kestrels may interpret that as ideal foraging habitat. However the trap exists in that turbines present a mortality risk for adults and especially fledgling kestrels. This could lead to a decrease in adult survival and juvenile recruitment. The potential direct and indirect effects of wind energy development on kestrel demography remain unclear, but it is possible that kestrel populations could be negatively affected. Therefore, research focused in areas of prospective wind energy development is justified.

Little data currently exist on kestrel population and ecology in the Llano Estacado. One way of monitoring kestrel populations is through nest-boxes. According to the American Kestrel Partnership website (2012b), the kestrel nest-box monitoring program nearest to the Llano Estacado is in Dallas, Texas, approximately 480 km away. Consequently I initiated a long-term nest-box monitoring program in Lubbock County, Texas, to study the nesting ecology of kestrels in this region. Kestrel breeding distribution is often limited even in an area of suitable habitat (Smallwood 1987) because

they are obligate secondary cavity nesters; as such, they are likely nest-site limited in the study area. Nest boxes are commonly accepted by kestrels and can raise the local abundance of breeding kestrels (Hamerstrom and Hamerstrom 1973); they are also easy for researchers to access. Because the Lubbock County landscape is dominated by agriculture and grasslands, I can monitor kestrel breeding populations and assess nest success with accuracy. I also explored different nest-box materials to assess their durability in arid climates and use by kestrels.

As an appealing, non-threatening, common bird of prey, kestrels are viewed an ambassador species for wildlife. They are valued because they prey on agricultural pests such as grasshoppers and beetles (Sherrod 1978) and because nest-box programs are a venue for citizen science. I began a nest-box monitoring program and conducted research on kestrel nesting ecology in an area of kestrel increases and burgeoning wind energy development. I had four specific objectives:

1. To establish a long-term nest-box monitoring program in Lubbock County, Texas;
2. To assess nest-box use and reproductive output over time through the nest-box program;
3. To compare hardiness of and kestrel use of alternative nest-box materials; and
4. To compare used nest-box proximity to overall nest-box proximity to wind turbines.

The program is set up so that we can gain insight over long-term monitoring of this charismatic falcon and to create management recommendations for the region and for wind energy interests.

## METHODS

### *Study Area*

The kestrel study occurred in Lubbock County, Texas. This site is located on the Llano Estacado, which is a high flat mesa covering about 83,000 km<sup>2</sup> of northwestern Texas and northeastern New Mexico (Leatherwood 2010). The Llano Estacado is within the Southern High Plains in the southern shortgrass prairie ecosystem. The representative topography is mainly flat tablelands. The study area is semi-arid; average rainfall is 48.6 centimeters annually, and temperature averages range from an average winter low of -2.3 degrees Celsius to an average summer high of 33.1 degrees Celsius (NOAA 2014). Summer temperatures at the study site commonly reach 40 degrees Celsius (K. Linner pers. obs.). Land-use practices are predominately agriculture, urban-developed, and herbaceous grassland (USDA 2009). The landscape was historically treeless except at water sources (Texas State Historical Association 2002) and thus had limited nest sites for kestrels.

I set up nest-boxes in Lubbock County at the Reese Technology Center (hereafter "RTC"), at Lubbock Lake National Historic Landmark (hereafter "Lubbock Lake"), and at the Texas Tech Native Rangeland compound (hereafter "Rangeland"; Fig. 4.1). All sites contain a mix of warm-season grasses (*Bouteloua dactyloides*, *Bouteloua gracilis*), forbs (*Kochia scoparia*, *Solanum elaeagnifolium*), mesquite (*Prosopis glandulosa*), and

yucca (*Yucca* spp.). Within the RTC study area (714 ha) is a decommissioned airfield containing small-scale industrial and wind turbine operations with runways and largely undisturbed shortgrass prairie. Numerous black-tailed prairie dog (*Cynomys ludovicianus*) colonies and some playas, which are ephemeral water sites, are present. The study area houses the Scaled Wind Farm Technology facility in the RTC, where construction of 3 scaled turbines began in December 2012. One separate industrial-sized turbine was constructed in February 2011, and another was constructed in October 2013. The Rangeland is comprised of 55 ha of moderately dense mesquite, native grass species, and a playa. Lubbock Lake is a 136 ha relic of the local native shortgrass prairie, managed for its interpretive trails and archaeological sites.

#### *Field Methods*

In 2012, another researcher attached 13 wooden nest-boxes to utility poles at the RTC. In 2013, I replaced most of those due to weathering and structural compromise. I constructed replacement nest-boxes out of three materials: pine wood, 30.5 cm diameter PVC with a plexiglass lid and bucket lid floor, and a 5-gallon plastic bucket with the lid. The climate of the Llano Estacado is conducive to drying out and warping wooden nest-boxes because of the typically low humidity, low rainfall, and high winds throughout the year. Wood Ducks (*Aix sponsa*) are also secondary cavity nesters and have successfully used 5 gallon plastic buckets as a nesting site (Griffith and Fendley 1981); American Kestrels nested in PVC nest structures in Georgia (Beasley and Parrish 2009) Based on these examples, I constructed kestrel nest-boxes out of PVC pipe and 5 gallon plastic buckets in addition to wooden nest-boxes. I compared how well they withstood weathering, and also evaluated them for kestrel interest or avoidance.

The PVC and bucket boxes were oriented vertically. All box types were comparable in size, though the PVC and bucket nest-boxes were slightly larger than the wooden nest-boxes (Table 4.1). All boxes had a 7.6 cm diameter entrance hole and six 2 cm diameter aeration holes near the box lid and on the floor of the nest-box. The replacement wooden nest-boxes were painted on outer surfaces, screwed together instead of nailed (as done in 2012), and constructed with hook and eye as well as spring-loaded hook and eye latches. I erected the 32 nest-boxes onto utility poles and wooden posts dispersed among three different sites (Table 25). I mounted nest-boxes at a height of 2.0 to 3.8 m above ground. I placed approximately 5 to 8 cm of aspen shavings in each box as a nest material substrate to prevent egg chilling (Varland and Loughin 1993). I checked nest-boxes bi-weekly between April and August, 2012 to 2014 (except for 2013 when nest checks began in early June). When necessary, I checked occupied nest-boxes more frequently to determine clutch initiation dates, hatching dates, and to band nestlings. I visually counted eggs and nestlings, and checked sex of nestlings when the boxes were opened. When found, I removed nests of European Starlings (*Sturnus vulgaris*) and House Sparrows (*Passer domesticus*), which are invasive to North America.

I hand-captured nestlings from nest-boxes when they were between 17-25 days old, and aged nestlings using Klucsarits and Rusbuldt's (2007) photographic guide. On two occasions, I trapped fledglings with limited flight using a large butterfly net. To band adults, I either hand-captured them from the nest-box, caught adults flying out of the nest-box entrance with a butterfly net, or trapped them using a dho-gaza setup with a mist net and a live Great-horned Owl near the nest-box. I banded unmarked adults and

juveniles with aluminum United States Geological Survey (USGS) bands. I recorded band numbers for recaptured adults to determine re-nesting events. All animal handling and banding was done in accordance with USGS bird banding permit 22801 and Institutional Animal Care and Use Committee protocol number 13028-03.

### *Analysis*

I considered a nest-box territory to be occupied when paired birds were present and engaged in territorial defense behavior, demonstrated nest affinity, or other reproductive activities regardless of clutch initiation (Steenhof and Newton 2007). I considered a nest attempt to be underway when clutch initiation was confirmed. I pooled the number of unique nest-boxes used per breeding season each year to determine nest-box use, regardless of multiple nest attempts in the same nest-box. I determined occupancy rate as the proportion of all nest-boxes available that were used per year (counting any box with 2 nest attempts only once).

Kestrels lay one egg approximately every other day and the female typically starts to incubate when she lays the penultimate egg; incubation lasts approximately 30 days in the wild (Bird and Palmer 1988). Thus, I estimated clutch initiation as 30 days prior to the hatching date, plus 3, 5, 7, or 9 days for clutches of 3, 4, 5, or 6, respectively. I averaged the dates of clutch initiation to find the mean date per nesting attempt. I determined hatching date and fledging date based on nestling age, and then averaged the hatching dates or fledging dates for all nest-boxes for their associated nesting attempt. I split clutch initiation, hatching, and fledging dates between first and second nesting attempts and estimated averages within each group.

I defined a re-nesting attempt to be when the same pair initiated a second nest, regardless of success or failure of the first nest attempt. I identified the same pair either by band identification of one or both adults when captured, or by proximity of the new nest to the old nest where the timing of initiation of the second nest corresponded with the timing of the first nest failure (Gault et al. 2004). For diurnal raptors, Steenhof (1987) recommends that when one nestling reaches 80% of fledgling age, the nest can be considered successful because mortality between the 80% age and actual fledging is minimal (Millsap 1981). Fledging age for kestrels in the wild is 28 days (Smallwood and Bird 2002), so the 80% value is 22 days. I defined hatching success as the number of eggs that hatched out of the number laid per nest, while fledging success is the number of hatchlings per nest that reached 22 days or greater. I defined nest success as the initiated nests in which one or more nestlings reached an age of 22 days or greater (Steenhof and Newton 2007), and calculated first and second nest attempts separately. Productivity is defined as the number of nestlings nearing fledging (at least 22 days) per pair (Steenhof and Newton 2007), also separated between first and second nest attempts.

I examined sex ratios of fledglings to see if they differed from a 1:1 male to female ratio across years and across the season in 2014. I also recorded occurrence of sex-biased broods. I classified an early-season nesting attempt as one with clutch initiation between March 5<sup>th</sup> and April 27<sup>th</sup>, and a late-season nesting attempt as clutch initiation between April 28<sup>th</sup> and July 8<sup>th</sup> (Smallwood and Smallwood 1998). The early and late periods corresponded with first breeding attempts and second breeding attempts.

In addition to calculating mean kestrel clutch size, brood size, number of fledglings per brood, and nesting success for all nesting kestrels, I calculated these

parameters per nest-box type. To do this, I pooled 2013 and 2014 data per nest-box type for calculations. I reported means and standard deviations per nest-box type, but could not perform a statistical test to compare among types because of the low sample size of nest attempts.

To assess proximity to turbines of used nest-boxes compared to all nest-boxes, I marked exact locations of nest-boxes and the five wind turbines present at the RTC with a GPS unit. I defined a used nest-box as one which had a nest attempt. One turbine was built between the 2013 breeding season and the 2014 breeding season, and one nest-box was removed in 2014; thus the turbine was not included in the 2013 analysis (Fig. 4.2) and the nest-box was not included in the 2014 analysis (Fig. 4.3). I calculated the distance from each nest-box to the nearest turbine each year using the “Near” tool in ArcMap 10.2. I averaged the distances from nest-box to nearest turbine for all used nest-boxes each year, and averaged the distances of all nest-boxes to their nearest turbine for comparison. Because of the low sample size of nest-boxes used by kestrels in 2013 and 2014, I did not conduct a statistical test to compare the used nest-box distance to nearest turbine to that of the available nest-boxes. Instead, I compared means and whether the standard deviations overlapped to determine differences.

## RESULTS

There were between 13 and 32 nest-boxes available to kestrels depending on year (Table 4.2). In 2013, when the greatest number of nest-boxes was available, 21 nest-boxes were made of pine wood, four were PVC, and seven were 5-gallon plastic buckets (Table 4.2). I assessed kestrel nest-box occupancy from 2012 to 2014 (Table 4.3). Occupancy increased over the three breeding seasons, with a maximum of 10 unique

boxes occupied and 10 unique pairs occupying boxes. The number of nesting attempts increased over the three years as well, with zero attempts in 2012, six in 2013, and 11 in 2014. Due to low sample sizes and a late start to monitoring in 2013, I did not have adequate data for the PVC or bucket boxes used in 2013 except for nest success analyses. Only occupancy in 2012 was included in tables because there were no nest attempts in any nest-boxes in 2012. In both 2013 and 2014, multiple pairs of kestrels fledged two broods. I describe reproductive measures of all nests combined, and of first and second attempts separately for kestrel pairs that attempted multiple nests.

Across the two years (2013-2014), kestrels laid 69 eggs (mean  $4.6 \pm 0.7$ , range 3-6) with an overall hatching success of 84%. Mean brood size was  $3.8 \pm 0.8$  and mean fledglings per nest was  $3.6 \pm 1.0$ , with a fledging success rate of 97%. Fourteen of 17 initiated nests (83%) fledged at least one chick over the 2 years of the study. Even after a successful first nest, one pair re-nested in 2013 and three pairs re-nested in 2014 for an overall rate of 44% re-nesting after a successful first attempt ( $n=9$ ; 2 successful first nest attempts in 2013 and 7 successful first nests in 2014). However, one of two first nests that failed in 2013 laid a replacement clutch and the only failed pair 2014 did not re-nest, yielding a rate of 33% re-nesting after a failed first attempt. Overall, regardless of the success of the first nest attempt, two of four total kestrel pairs re-nested in 2013, and three pairs re-nested out of eight in 2014 (Table 4.4). The overall re-nesting rate over two years equaled 42%. Double-brooding, or successfully raising two broods to fledging, occurred once in 2013 and three times in 2014.

I compared nesting success between first and second nest attempts in 2013. I calculated additional reproductive measures for first and second nesting attempts in 2014. Clutch sizes, brood sizes, and number of fledglings per brood of the first nesting attempt averaged higher than the second in 2014 (Table 4.4). Hatching success, fledging success, and nest success were 68%, 100%, and 88%, respectively, during the first nesting attempt period of 2014. Hatching success (91%) and nest success (100%) were higher in the second nesting attempt (Table 4.4). Productivity was greater in the first nesting attempt (4.5 young per pair occupying a nesting territory) than the second nesting attempt in 2014 (2.5 young per pair occupying a nesting territory). I also did calculations for only the five pairs of kestrels that each had two nests per season across 2 years and found that nest success was higher for second-attempt nests than for first attempts (Table 4.4). Of this subset with two nest attempts, 80% successfully fledged young on their first attempt and 100% were successful on their second attempt. The rate of pairs successfully fledging two nests is 33% in this study.

In 2013, first-attempt nest clutches were initiated at an average date of 25 April, hatched around 1 May, and fledged on approximately 29 May (Table 4.5). The standard deviation for the first-nest clutches was 53 days; however, two nests failed, which reduced the hatching and fledging standard deviation to two days (Table 4.5). Clutch initiation for the first nesting attempts in 2014 was approximately 31 March, with a standard deviation of six days for eight nests. The first nest attempts in 2014 had slightly later hatching and fledging dates as in 2013 (7 May and 3 June, respectively; Table 4.5). The earliest estimated clutch initiation date was 24 March in both 2013 and 2014 for first nest attempts.

The phenology of second nest attempts by kestrel pairs in 2013 and 2014 were similar, with 2013 dates having greater standard deviations (23-25 days) than 2014 (3-5 days). In 2013, second clutches were initiated at an average date of 29 May, hatching occurred at an average date of 4 July, and fledging occurred at an average date of 31 July (Table 4.5). In 2014, kestrels on their second nest attempt initiated their clutch around 8 June, nestlings hatched on approximately 12 July, and fledged on approximately 8 August. The latest estimated clutch initiation date for known second nest attempts was 16 June in 2013 and 11 June in 2014.

The sex ratio of fledgling kestrels was male-biased in the first brood in 2014 (Table 4.4); six of the seven broods in the early season were sex-biased, and four of those were male-biased (Table 4.6). However, the sex ratio of fledgling kestrels was female-biased in second broods in 2013 and 2014 (Table 4.4). The late nesting period revealed a sex ratio of 2:3 males to females. There were two biased broods of the three in the late nesting period; both of these broods were female-biased (Table 4.6).

I compared clutch initiation dates and fledging dates for nest-boxes with two nest attempts. There were two pairs in 2013 that attempted two nests and three pairs in 2014 that attempted two nests (Table 4.7). Both pairs with two nest attempts in 2013 also attempted two nests in 2014 (pairs 1 and 2; Table 4.7). Only one first attempt of the double-nesters failed: the first attempt of pair 2 in 2013. In two instances, kestrel pairs waited approximately two weeks between when their first nest fledged and clutch initiation for the second nest. In the other two re-nesting attempts, the female kestrel began laying a second clutch before any nestlings fledged from her previous nest (Table

4.7). In one of these re-nesting attempts, the kestrel female moved to another nest-box 140 m away whereas the other re-nesting female remained in the same nest-box (Table 4.7).

Kestrels fledged young out of at least one of every type of nest-box. Out of 17 nest attempts in 2013 and 2014, 76% were in wooden nest-boxes, 6% were in PVC nest-boxes, and 18% were in bucket nest-boxes (Table 4.8). Due to low sample sizes and a late start to monitoring in 2013, we did not have data for the PVC box used in 2013 except for nest success. In the bucket boxes, clutch size, brood size, and number of fledglings per brood were lower than in the wooden boxes. Hatching success and fledging success were generally lower in wooden nest-boxes than the other two materials; however there were few nests in the PVC and bucket boxes for comparison. Overall, nest success in wooden boxes was lower than PVC and bucket nest-boxes (77%, 100%, and 100% respectively; Table 4.8). It appears that kestrels are willing to use all of the materials and demonstrated successful nesting in all types of nest-boxes. Bucket nest-boxes were the simplest and least expensive to assemble, PVC nest-boxes more difficult and more costly, and the wooden nest-boxes were the most difficult to assemble and most expensive. All nest-box materials, except the PVC nest-boxes' plexiglass lids, withstood the climate well and did not need to be replaced throughout 2013 and 2014. Two of four lids on the PVC boxes broke in high winds and storms in late summer 2013, and the nest-boxes were not repaired. Spring-loaded hook and eye screws rusted in the second summer and were difficult to unlatch, causing researchers to spend additional unwanted time at the nest-box.

Kestrels nested in three nest-boxes of 18 available at the RTC in 2013 (Fig. 4.2), and in five nest-boxes of 17 available in 2014 (Fig. 4.3). Four nest attempts occurred in the three nest-boxes used in 2013, and seven nest attempts occurred in the five nest-boxes in 2014. There appeared to be no strong selection or avoidance of wind turbines by kestrels at the RTC. In both years, the shortest distance of any nest-box to a turbine was 0.31 km, and the longest was 2.51 km (Table 4.9). In both years, the mean distance of used nest-boxes to nearest turbine was smaller than that of all nest-boxes to turbines; however, the standard deviations were large results overlapped (Table 4.9). All nest-boxes except one at the RTC had a turbine within the greatest recorded home range radius of 2.4 km (Craighead and Craighead 1956; Table 4.9), and in both years, four nest-boxes had a wind turbine within the smallest recorded home range radius of 0.5 km (Smith et al. 1972; Table 4.9). Of use nest-boxes, 67% had a turbine within 600 m in 2013 and 60% did in 2014. I noticed on multiple occasions that up to five post-fledging kestrels would gather on anemometer tower guy wires within 100 m of turbines during the breeding season.

## DISCUSSION

The increase in kestrel nesting attempts from 2012 to 2014 is likely a result of three factors: First, the number of nest-boxes available increased in 2013. Second, after nest-boxes are introduced to an area, a steep increase in occupancy occurs initially (Hamerstrom and Hamerstrom 1973, Smallwood and Bird 2002, Smallwood and Collopy 2009). Third, an extreme drought occurred in the study region and surrounding area beginning in 2011; a gradual increase in rainfall over the study years may have had a positive effect on occupancy and nest success. The overall occupancy of nest-boxes in

this study (22-33%) was at the low end of a range of results from various nest-box monitoring programs across the country (Eschenbauch et al. 2009). They were also lower than studies of nest-boxes in Missouri (53% over three years; Toland and Elder 1987) and another southern latitude study in Georgia of southeastern kestrels (*Falco sparverius paulus*) nesting in tubular cross-arms of transmission towers (74-80% over four years; Beasley and Parrish 2009).

Most kestrel nesting studies report reproductive parameters that are not separated by first or second nest attempt. Mean clutch size in this study is equivalent to mean clutch size determined in a summary by Smallwood and Bird (2002). Brood size in this study is also comparable to the nesting summary by Smallwood and Bird (2002). Overall hatching success, fledging success, and nesting success in this study (84%, 96%, and 82%, respectively) were higher than the review study (Smallwood and Bird 2002).

I found that kestrel clutch initiation, hatching, and fledging dates in Lubbock, Texas, occurred at early end of recorded kestrel nesting phenology across the country according to Smallwood and Bird (2002). Clutch initiation dates for the southeastern kestrel in areas south of this study spanned 7 March to 5 July in north-central Florida (including second clutches; Smallwood and Smallwood 1998) and 22 March to 23 June in north west Florida (including second clutches; Gault et al. 2004). The span of clutch initiation dates for this study in Texas was from 24 March to 25 June. Farther north, in Iowa, they spanned 1 April to 23 June (Varland and Loughin 1993). Because the nest initiation is delayed by days to weeks with increase in latitude, it is logical that nesting

phenology at this Texas study appears to fall between breeding dates recorded at sites to the north and to the south.

The location at the southern breeding latitudes also helps to explain the high occurrence of re-nesting in the Lubbock study site of approximately 42% over two years. Smallwood and Collopy (2009) reported a rate of approximately 11% re-nesting in their study in north-central Florida. Similarly, Gault et al. (2004) found a 10% re-nesting rate in north west Florida and J. Brown (pers. comm.) detected an approximately 20% rate of re-nesting in 2008 and 10% in 2009 at her study site in Florida, attributing the drop in re-nesting attempts to unfavorable weather conditions. The nearest study site in Colorado reported a 9% re-nesting rate (Stahlecker and Griese 1979). The frequency of re-nesting reported from the Texas study is higher than all other published reports, and is worth further investigation. Not only is the re-nesting rate high, but the occurrence of pairs successfully fledging two nests, or double-brooding, is 33% in this study. Gault et al. (2004) did not report any double-brooding in northwest Florida, but Smallwood and Smallwood (1998) reported that 4% fledged two broods in north-central Florida. Kestrels successfully double-brooding is also reported in Missouri (Toland 1985), Colorado (Stahlecker and Griese 1977), and even Idaho (Steenhof and Peterson 1997).

Bowman and Bird (1985) reported 11-12 days as the mean time for kestrels to initiate a replacement clutch. I found that two nests showed similar time delays before laying a new clutch, but also found that two initiated a new clutch up to 2 days before any of her previous nestlings had fledged (Table 4.7). However, because clutch initiation dates are estimated using many assumptions such as a 30-day incubation period,

incubation initiation upon the laying of the penultimate egg, and approximately 2 days between laying of each egg (Bird and Palmer 1988), there is less certainty in these estimations. It may be that kestrels nesting in the southern latitudes of North America incubate fewer than 30 days before the nestlings hatch, explaining some of the discrepancy between new laying dates and previous fledging dates. Both examples of nests where incubation time was assessed were consistent with the expected 30 days for incubation and two days between each consecutive egg, with incubation beginning after the penultimate egg is laid (Bird and Palmer 1988) in the literature. However, because the clutch initiation estimates could be off by up to two days, a small shortening of the incubation time in this warmer, southern latitude could go unnoticed if it exists.

Although sample sizes were small, neither year of this study showed fledgling sex ratios near a 1:1 male to female ratio. In 2013, with lower-than-average precipitation levels, there was a greater ratio of females to males. In 2014, a year where the precipitation was enough to nearly bring west Texas out of a multi-year drought, there was a greater proportion of males than females. This is not consistent with a study by Wiebe and Bortolotti (1992) in Saskatchewan, which showed that parents in poor condition, or that have a decreasing food supply, produced more males than females, opting to produce more of the smaller sex. In contrast, the sex ratios of this study are consistent with the theory that parents in good condition tend to produce more males (Trivers and Willard 1973). This assumes that a slight advantage in fledgling condition would have disproportionately greater benefits for males than females, as males compete for available females. The 2014 breeding season also showed a switch in the sex ratio of fledglings from the beginning to the end of the season. The Early Bird Hypothesis

(Smallwood and Smallwood 1998) is a modification of the Trivers-Willard hypothesis and purports that competition among post-fledgling, dispersing males gives an advantage to males that fledge early in the season. This was illustrated in Florida when sex ratios of resident kestrel broods shifted from being male-biased in the beginning of the season to being female-biased in the end of the season (Smallwood and Smallwood 1998). It is not known if the kestrels in the Lubbock County, Texas, study site are migratory, residential, or a mix of both. Smallwood and Smallwood (1998) found that only residential populations demonstrated the Early Bird Hypothesis, which infers that the Lubbock birds may be year-round residents. Although the sample size of my study is small, the sex ratios across years and across the season appear to support the Trivers and Willard hypothesis and the Early Bird Hypothesis.

Kestrels chose to nest in wooden nest-boxes more often than the other types, but even though the number of nests attempted in these box types was low, all nest attempts in both bucket nest-boxes and PVC nest-boxes were successful. Although kestrels use the buckets less often than the wooden nest-boxes, my study demonstrates that they will use them successfully. Because of their extremely low cost, the lower bucket occupancy rate may be remedied by a greater number of bucket boxes that can be afforded and installed. Before they are installed on a large scale, studies need to be undertaken to assess the temperature and other variables within the buckets compared to the wooden boxes and the outside conditions to ensure they are not harmful to kestrels. The non-durability of the PVC design used in this study and the low nesting attempts in this material establishes that PVC designs may not be worth constructing and installing to augment kestrel populations.

The proximity analysis did not demonstrate that kestrels have an affinity for, or an avoidance of, wind turbines. Kestrels' breeding-season home ranges are 2.4 km in diameter in Michigan and Wyoming (Craighead and Craighead 1956), 0.5 to 1.1 km in diameter in Utah (Smith et al. 1972), and 2.3 km diameter in Illinois (Enderson 1960). If the home range of kestrels at the RTC is the largest estimate, nearly all nest-boxes at the RTC have a wind turbine within their home range. The majority of used kestrel nests in both years had turbines within 600 m of the nest. I suggest that the mean nest-box distance-to-turbine assessment be completed again in future years because conclusions should not be inferred from the small sample sizes of this study. Significant differences in nest-box proximity to turbines, if found, could indicate important influences. If kestrels nest closer to turbines than the mean distance of all nest-boxes to turbines, it could indicate a selection for habitat near turbines. If so, kestrel perception that the turbine area is high-quality habitat may pose an ecological trap (Gates and Gysel 1978, Schlaepfer et al. 2002). This could be due to disturbed, shortened vegetation that evolutionarily indicates good prey availability (Bildstein and Collopy 1987), or an increase in available perch sites on new buildings or utility lines that allow for energy-saving perch-hunting (Collopy and Koplín 1983). If the opposite result was found over time - that kestrels nest farther than the mean distance of all nest-boxes to turbines - it could indicate habitat displacement (as seen in Garvin et al. 2011) due to turbine construction and associated activity.

American Kestrels in the Llano Estacado are faced with the challenge of increasing wind energy development as well as the underlying cause of their population decline. So far in Lubbock County, kestrels have demonstrated multiple strategies to

maintain their population despite adverse conditions. Within three years of monitoring, kestrels in Lubbock County showed an increase in occupancy and nesting attempts. They produced average-sized clutches, broods, and fledglings compared to other studies.

Kestrels in this region are remarkable in their early clutch initiation dates, high occurrence of successfully fledging two nests in one season, and uneven sex ratios.

Future monitoring of the nest-boxes may continue to show interesting patterns of kestrel nesting ecology in the Llano Estacado and could prove useful in creating management recommendations.

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Table 4.1. Exterior dimensions (cm) of three nest-box types installed in this study.

Box type	Wall thickness	Floor length or diameter	Floor width or diameter	Top diameter	Entrance hole diameter	Height: floor to entrance hole	Height: front	Height: back
Pine Wood	2.5	26	23.5	-	7.6	25.4	38.1	43.2
PVC	0.2	30.5	30.5	30.5	7.6	27.9	40.6	45.7
Bucket	0.1	25.4	25.4	30.5	7.6	27.9	45.7	45.7

\*all measurements are in cm

Table 4.2. Distribution of nest-boxes among three sites in 2013.

Box type	Lubbock Lake			All sites
	Reese Technology Center	Texas Tech Native Rangeland	National Historic Landmark	
Pine wood	14	4	3	21
PVC	2	2	0	4
Bucket	2	4	1	7
Total	18	10	4	32
#nest-boxes/ha	0.025	0.182	0.029	0.035

Table 4.3. Occupancy of nest-boxes and number of nest attempts by kestrels at all sites from 2012 - 2014

Year	No. boxes total	No. unique boxes occupied	Occupancy rate	No. pairs occupying	Total no. occupying events*	No. nest attempts
2012	13	1	8%	1	1	0
2013	32	8	25%	7	8	6
2014	30	10	33%	10	12	11

\*including re-nesting events where same box may be used twice in one season

Table 4.4. Comparison of nest attempts, clutch size, brood size, fledglings/brood, nest success, and sex ratio among first and second nesting attempts in 2013 and 2014.

Year	Attempt	Total nest attempts	Proportion that re-nested	Clutch size Mean $\pm$ SD (n)	Brood size Mean $\pm$ SD (n)	Fledglings/ brood Mean $\pm$ SD (n)	Nest success	Sex ratio M:F (n fledglings)
2013	1 <sup>st</sup>	4	50%	4.5 $\pm$ 0.7 (2*)	0 (2), unk (2*)	0 (2), unk (2*)	50%	unk*
	2 <sup>nd</sup>	2		4.5 $\pm$ 0.7 (2)	4.5 $\pm$ 0.7 (2)	3.5 $\pm$ 2.1 (2)	100%	1:1.7 (8*)
	both	6		4.5 $\pm$ 0.6 (4*)	4.5 $\pm$ 0.7 (2*)	3.5 $\pm$ 2.1 (2*)	67% (6)	1:1.7 (8*)
2014	1 <sup>st</sup>	8	38%	5 $\pm$ 0.8 (8)	3.9 $\pm$ 0.9 (7)	3.7 $\pm$ 1.0 (7)	88%	1:0.4 (26)
	2 <sup>nd</sup>	3		3.7 $\pm$ 0.6 (3)	3.3 $\pm$ 0.6 (3)	3.3 $\pm$ 0.6 (3)	100%	1:1.5 (10)
	both	11		4.6 $\pm$ 0.8 (11)	3.7 $\pm$ 0.8 (10)	3.6 $\pm$ 0.8 (10)	91% (11)	1:0.7 (36)

\*only nest success was determined for first 2 nests of 2013 (clutch size, brood size, no. fledglings was not determined)

\*\*Clutch initiation date for one nest in 2013 not known

Table 4.5. Comparison of reproductive measures among first and second nest attempts.

Year	Attempt	Total nest attempts	Proportion that re-nested	Clutch initiation Mean $\pm$ sd (days) (n)	Hatching Mean $\pm$ sd (days) (n)	Fledging Mean $\pm$ sd (days) (n)
2013	1 <sup>st</sup>	4	50%	25-Apr* $\pm$ 53 (3**)	1-May $\pm$ 2 (2)	29-May $\pm$ 2 (2)
	2 <sup>nd</sup>	2		29-May $\pm$ 25 (2)	4-Jul $\pm$ 23 (2)	31-Jul $\pm$ 24 (2)
2014	1 <sup>st</sup>	8	38%	31 Mar $\pm$ 6 (8)	7-May $\pm$ 6 (7)	3-Jun $\pm$ 8 (7)
	2 <sup>nd</sup>	3		8-Jun $\pm$ 3 (3)	12-Jul $\pm$ 3 (3)	8-Aug $\pm$ 5 (3)

\*clutch initiation date was not determined for first 2 nests of 2013

\*\*Incubation initiation date for one nest in 2013 not known

Table 4.6. Fledgling sexes per nest box and sex ratios and biased broods across the "early" and "late" nesting period in 2014.

Nesting period	Nest-box name	Fledglings per nest	#M	#F	Clutch Initiation Date	Sex ratio M:F	No. male-biased broods (n)	No. female-biased broods (n)
early	CR17	3	3	0	24-Mar	9:4	4 (7)	2 (7)
	CRAUSTIN	5	2	3	25-Mar			
	CR19	5	5	0	26-Mar			
	CR26	3	3	0	4-Apr			
	CR09	3	2	1	5-Apr			
	CR11	3	1	2	5-Apr			
	CR18	4	2	2	10-Apr			
late	CR27	3	1	2	4-Jun	2:3	0 (3)	2 (3)
	CRAUSTIN	4	2	2	9-Jun			
	CR09	3	1	2	11-Jun			

Table 4.7. Dates of clutch initiation, estimated first and final nestling fledging dates for only kestrel pairs with two nesting attempts. I also calculated the number of days between first and second nest attempts using the date of first nestling fledging dates as well as the date of the final nestling fledging.

		First attempt			Second attempt		days between 1st nestling fledging and 2nd initiation	days between final nestling fledging and 2nd initiation	Pair moved nest-boxes?
		Clutch initiation	1st Fledgling	Final fledgling	Clutch initiation	1 <sup>st</sup> Fledgling			
2013	Pair 1	27-Mar	31-May	unk	16-Jun	17-Aug	16	unk	Y
	Pair 2	unk	-	-	12-May	14-Jul	-	-	Y
2014	Pair 1	4-Apr	5-Jun	9-Jun	4-Jun	4-Aug	-1	-5	Y
	Pair 2	25-Mar	26-May	27-May	9-Jun	8-Aug	14	13	N
	Pair 3	5-Apr	13-Jun	15-Jun	11-Jun	14-Aug	-2	-4	N

Table 4.8. Comparison of reproductive performance among three nest box types in 2013 and 2014

	Wooden Mean $\pm$ sd (n)	Type PVC Mean $\pm$ sd (n)	Bucket Mean $\pm$ sd (n)	All types Mean $\pm$ sd (N)
No. boxes - 2013, 2014	21, 21	4, 2	7, 7	32, 30
No. nest attempts	13	1	3	17
Proportion of nest types used	76%	6%	18%	-
Avg. Clutch size	4.7 $\pm$ 0.8 (13)	unk*	4 $\pm$ 0 (2*)	4.6 $\pm$ 0.7 (15*)
Avg. Brood size	4.0 $\pm$ 0.8 (10)	unk*	3 $\pm$ 0 (2*)	3.8 $\pm$ 0.8 (12*)
Avg. Fledglings per brood	3.7 $\pm$ 1.1 (10)	unk*	3 $\pm$ 0 (2*)	3.6 $\pm$ 1.0 (12*)
Hatching Success	72%	unk*	75%*	73%
Fledging Success	95%	unk*	100%*	96%
Nest success	77%	100%	100%	82%

\*only nest success was collected for 2013 bucket and PVC boxes (clutch size, brood size, and no. fledglings was not known)

Table 4.9. Distances of nest-boxes to wind turbines at the RTC, summarized by used and all nest boxes for comparison.

Nest-box ID	2013	2014
	Distance to turbine (km)	Distance to turbine (km)
CRAUSTIN	0.6	0.6
CR05	1.7	1.7
CR06	1.2	1.2
CR07	1.9	1.9
CR08	1.2	1.2
CR09	0.5	0.4
CR10	0.3	0.3
CR11	1.3	1.3
CR12	1.8	1.8
CR13	2.2	2.2
CR14	1.1	1.1
CR15	0.5	0.5
CR16	1.6	1.1
CR17	2.5	2.5
CR28	1.4	1.4
CR29	2.1	2.1
CR30	0.8	-
CR31	0.4	0.4
All		
N	18	17
mean	$1.3 \pm 0.7$	$1.3 \pm 0.7$
min	0.3	0.3
max	2.5	2.5
Used		
N	3	5
mean	$0.9 \pm 0.7$	$1.1 \pm 0.9$
min	0.5	0.4
max	1.7	2.5

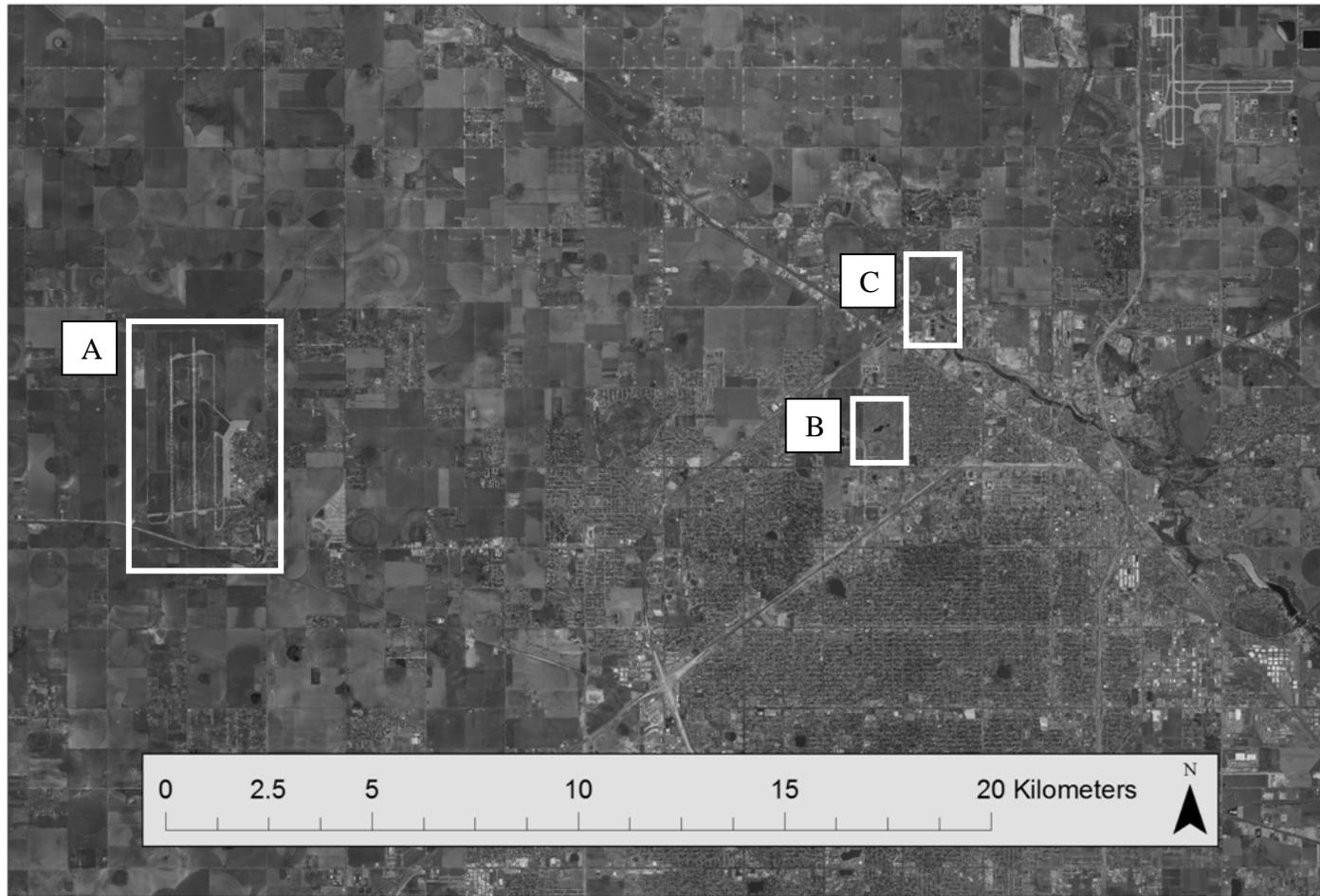


Figure 4.1. Location of three study sites in Lubbock County, Texas. Site "A" is the RTC, site "B" is the Texas Tech University Native Rangeland, and site "C" is Lubbock Lake National Historic Landmark.

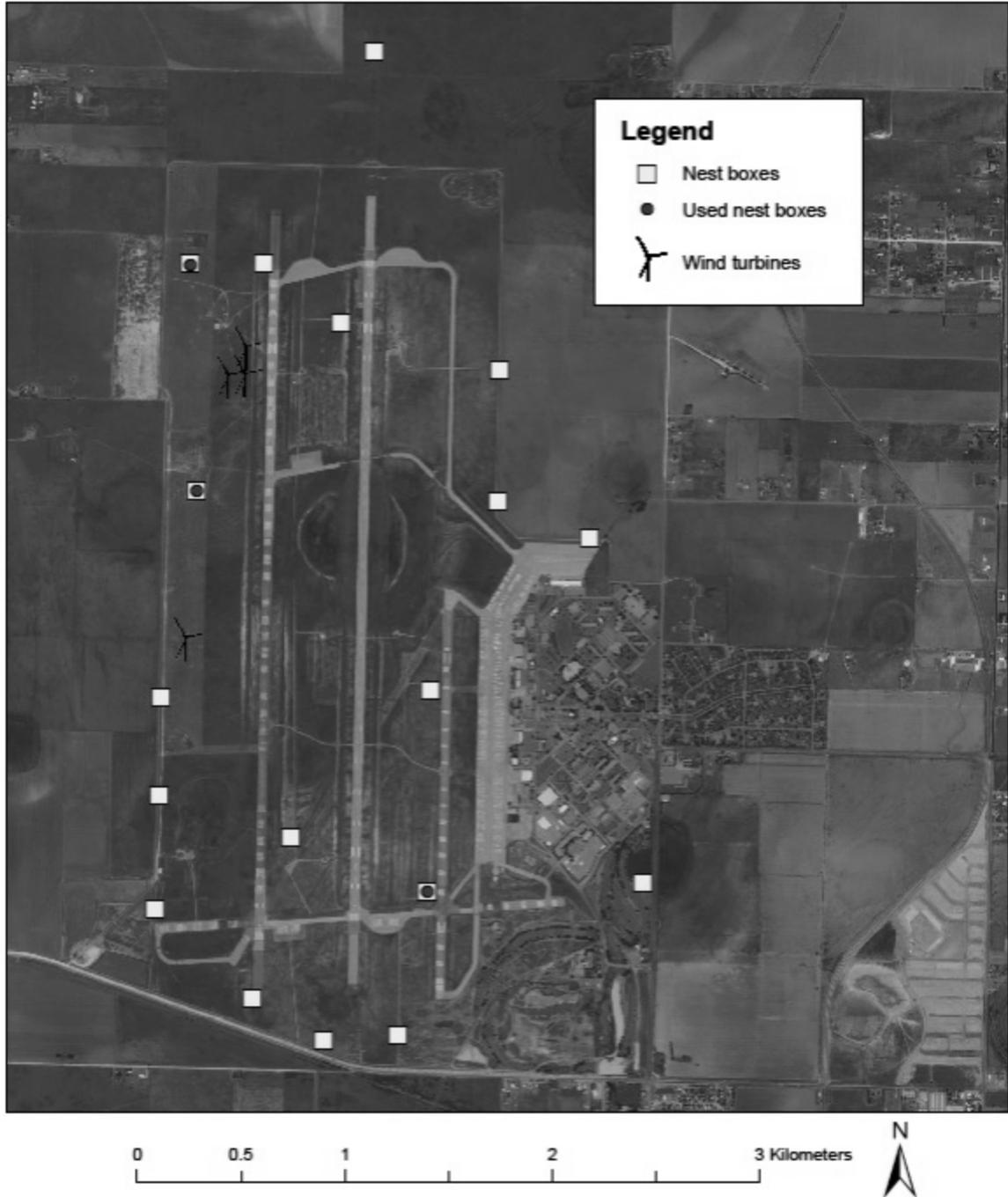


Figure 4.2. Locations of all American Kestrel nest-boxes, with used nest boxes indicated with a dark circle, and wind turbines at the RTC in 2013.

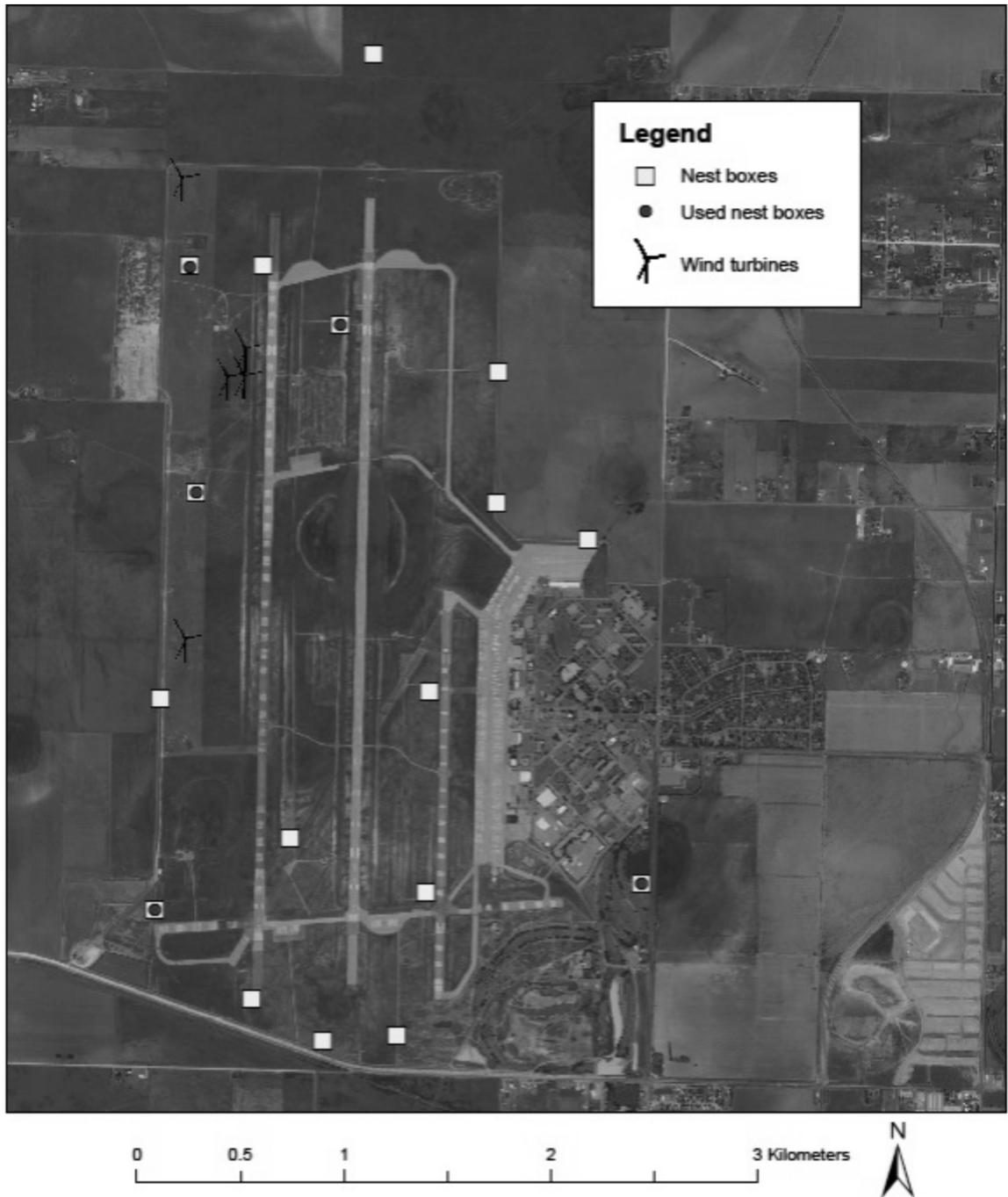


Figure 4.3. Locations of American Kestrel nest-boxes, with used nest boxes indicated with a dark circle, and wind turbines at the Reese Technology Center in 2014.