IMPACTS OF ANTHROPOGENIC DISTURBANCES AND DROUGHT ON BREEDING BIRD ABUNDANCE AND DIVERSITY IN THE ROLLING PLAINS ECOREGION OF TEXAS

by

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ABSTRACT

IMPACTS OF ANTHROPOGENIC DISTURBANCES AND DROUGHT ON BREEDING BIRD ABUNDANCE AND DIVERSITY IN THE ROLLING PLAINS ECOREGION OF TEXAS

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Bird species, especially grassland species have undergone widespread decline in arid grasslands due to anthropogenic disturbance such as different land usages including cattle grazing and agriculture and addition of physical structures such as buildings, roads and fences which can negatively impact both animals and native vegetation. The objective of this research was to determine the impacts of anthropogenic disturbance (i.e., presence of oil pump jacks and cattle grazing) and drought on breeding bird community dynamics in the Rolling Plains Ecoregion of Texas. The study involved four sites encompassing different land uses including a natural-non-cattle grazed, cattle-grazed, oil field pump jacks and mixed oil field pump jacks/cattle-grazing sites. Bird abundance, species richness and diversity (H') were compared across these sites from 2008 to 2012 excluding the drought year of 2011. Bird counts were undertaken on 10 sub-sites within each of the sites during the bird breeding season and the data used to determine sub-site species richness and diversity. Percent vegetation cover (i.e. woody shrubs or woody shrubs plus grasses) was determined for each sub-site by analysis of Google Earth© images available for the pre-drought year of 2008 and the post-drought year of

2012. Examination of the effects of climate involved analysis of mean June daily high temperature and January-June total precipitation data for the city of Post, Texas, relative to that of North American Breeding Bird Survey (BBS) data for the City of Post, TX Route from 1969 to 2011 in the vicinity of the four study sites. Data on sub-site bird abundance, species richness and species diversity index (H') were used to make three response variables (i.e., response 1: data for all years and sites averaged, response 2: pre-drought mean data across all sites for 2008, 2009, and 2010 minus that for 2012, and response 3: data for all sites in 2009 minus that for all sites in 2012). These three responses were analyzed separately for sub-site bird abundance, species richness and species diversity using a Multiple Factor ANCOVA for the fixed effects of oil pump jacks and cattle grazing with percent sub-site woody vegetation cover as a covariate. Sub-site differences were further examined for the effects of drought, cattle grazing and presence of oil pump jacks using a multiple factor ANOVA with sites and year (2009 versus 2012) as factors. Least Squares Linear Regression Analyses were utilized to determine possible relationships between sub-site percent vegetation cover (woody shrubs and grasses) and the dependent variables of mean sub-site bird abundance, species richness and species diversity index (H') using combined pre- and post-drought data from 2009 and 2012. Climate effects were examined using Least Squares Linear Regression Analyses using the Post, TX, BBS route data for bird abundance, species richness and species diversity index (H') versus June average high daily temperatures and January-June total precipitation values as independent variables. Regression analyses were also conducted on these data using the prior year temperature and precipitation data to examine the latent effects of drought on the avian community. Although post-drought vegetation cover was reduced by 20% across all sub-sites in 2012 after the 2011 drought, it did not impact ($p = 0.187$, 0.792, and 0.384, respectively) 2012 bird abundance, species richness, and species diversity values across the study sites. Nor was BBS data for these three variables significantly correlated ($p = 0.807$, 0.252 and 0.560, respectively) with either mean June daily high temperature or January-June total precipitation.

The only significant correlation ($p = 0.016$) recorded for BBS data was that of species richness versus January-June precipitation in the prior year. Nor did a *post-hoc* Fisher's Least Significant Difference (LSD) procedure with significance set at $p < 0.05$ indicate a difference in the three community response variables between the cattle-grazed and the natural grassland site, but did indicate that the presence of oil pumps jacks, significantly ($p < 0.05$) increased bird abundance and, to a lesser extent, species richness relative to the cattle-grazed and natural grassland sites. The results, in general, appeared to agree with those of previous studies examining similar impacts on grassland avian communities in which the moderate disturbance associated with presence of oil pump jacks increased bird abundance but did not greatly impact species richness or diversity. The lack of impact of drought on grassland bird abundance, species richness and diversity revealed in this study is similar to that reported in other studies. However, negative drought impacts have been recorded for some bird communities especially when examined over a broad geographical scale. Overall, my results indicated that the Rolling Plains Ecoregion grassland bird communities studied appeared relatively resistant to the impacts of both anthropogenic disturbance in the form of oil pump jacks and cattle grazing and to the impacts of drought.

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CHAPTER 1

INTRODUCTION

The human ecological footprint includes not only dwellings, but also recreation and energy needs. As humans disperse deeper into natural habitats threatening plant and animal life, the need for protection and balance has increased. My project examined the relationship of breeding birds with respect to anthropogenic disturbance and drought in Garza County within the 81 county Rolling Plains Ecoregion of Texas (Seyffert,2001) (Fig. 1.1). The project involved assessments of the impacts of oilfields, cattle grazing and drought on bird abundance, species richness and diversity.

Biodiversity is a multifaceted issue that combines natural processes and anthropogenic pressures. Whereas, biodiversity is generally considered as a variety of measures of species diversity, it can also incorporate genetic diversity and community and landscape structure (Whitford, 1997). Factors that decrease ecosystem biodiversity including fragmentation, grazing, forestry and nutrient distribution can also contribute to loss of ecological stability (Tilman, 1996). From an economic standpoint, biodiversity can be discussed in terms of quality, quantity and stability over time. Further, it has been suggested that species richness, composition, biomass productivity and temporal stability interact in a bi-directional, feedback loop, rather than being independent factors (Worm et al., 2003).

Bird species, especially grassland and scrubland dwellers, have experienced population declines for a century, in North America and other parts of the world (Askins et al., 2007). Grassland specialist species in particular have experienced widespread declines in the past 25

years (Askins et al., 2007). As in most ecoregions, the International Union for Conservation of Nature's (IUCN) conservation status of bird species included in my study ranged from those of least concern, such as *Callipepla squamata* (scaled quail), to the near threatened status of *Colinus virginianus* (Northern bobwhite) (IUCN, 2013). Factors involved with bird habitat preference include availability of food, nesting and risk of predation.

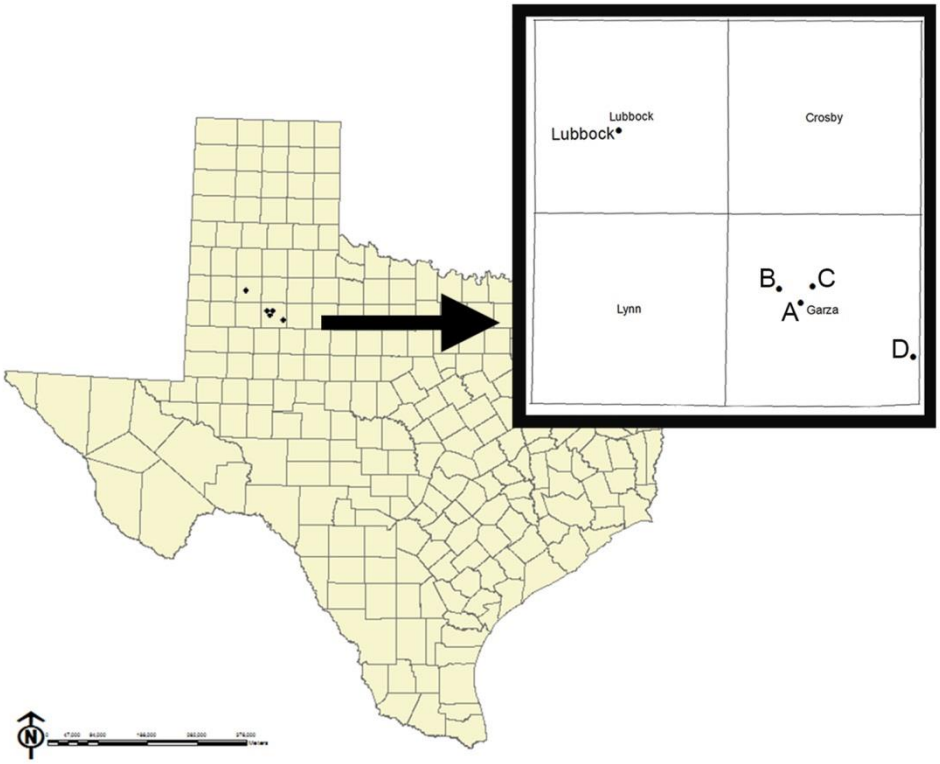

Figure 1.1. State map of Texas and counties with the location of the field research sites indicated. All four sites, A, B, C and D were located in Garza County, Texas, which was located southeast of Lubbock, Texas. The map was generated using ArcGIS (ESRI, 2011).

The North American Breeding Bird Survey (BBS) (Pardieck, 2001) tracks bird species' population densities relative to factors such as habitat fragmentation, nest parasitism, pesticides, pollution and invasive species, all factors required for conservation and restoration of bird habitats. The North American BBS began in 1966, collecting data on over 420 species in the United States and Canada (Sauer et al., 2011). The North American Breeding Bird Surveys

are conducted by the Biological Resources Division of the U.S. Geological Survey (USGS) and involves volunteers, varying in experience and skill. The BBS is conducted between May and June each year along established 39.4 km (listed as 24.5 mi) routes. Data collection consists of point counts for three minutes each at 50 stops (0.5 mile intervals) along each route (Sauer et al., 2012).

The North American BBS includes over 5,100 established routes; with an estimated 2,500 of these route surveys conducted annually on one day by volunteers between May and June (Sauer et al., 2011). These BBS data are important for tracking bird population changes in North America and are used in conjunction with Christmas Bird Counts (CBC) to track migration patterns. The BBS and CBC point counts are sent as raw data sets to the Patuxent Environmental Science Center in Laurel, Maryland, where these data and their analyses for routes and species range maps are made available through the USGS website (Sauer et al., 2012).

In working to determine the effects of roadside habitat bias, Niemuth et al. (2007) designed survey areas called “neighborhoods” to yield a more accurate relationship between birds and the habitat. Their results indicated no significant differences between BBS routes and their designed “neighborhoods” when examining landscape structure that included croplands, grasslands, wetlands, forests and urban areas (Niemuth et al., 2007). Breeding Bird Surveys in Britain and France use parameters that differ from those used in the North American BBS. Britain’s BBS consists of randomly selected 1-km square areas and France’s, 2-km square sites, both of which are surveyed twice within each survey period (Davey et al., 2012, Jiguet et al., 2011).

The primary cause of avian habitat loss has been and remains human land use (St. Louis et al., 2006). Population declines in grassland bird species have occurred due to development of forests in North America on the vast agricultural lands abandoned during the eighteenth and nineteenth centuries. The Homestead Act of 1862 (United States) and the

Canada Dominion Act of 1872 had major negative impacts on grasslands as they allowed them to be turned to agricultural use. In the United States, over 800,000 km of land was distributed to close to 1.5 million people (Samson et al., 2004). However, factors that have led to declines in grassland bird species cannot be attributed simply to changes in available habitat. Indeed, for some grassland bird species, managed agricultural and grazing lands are good habitats for breeding and wintering. However, as agriculture tends toward monoculture, its suitability decreases for some species. Farming increases the use of pesticides and plowing alters drainage patterns. Plowing, harvesting and grazing eliminate the natural edges of fields and/or destroy nesting sites and existing nests. Along with agricultural practices, other factors that cause habitat fragmentation include roads, trails, oil/gas extraction, wind turbine farms and invasive plants. Roads and trails further fragment grasslands by causing bird mortality and acting as vectors for dispersal of invasive non-indigenous grass species (Askins et al., 2007).

Classical landscape fragmentation hypotheses address fragments as islands and metapopulations, which address both habitat and matrix. Naturally occurring patchy landscapes composed of diverse cover types can lead to habitat isolations that are more dramatic than those created by geography alone (Ricketts,2001). Ecological fragmentation effects include roads as well as developed land plots. Habitats that are fragmented are able to maintain gene flow via connecting corridors. In theory, ecological corridors prevent species loss. Thus, it is important to examine the relationship of plants and animals to these habitats. Plants can be tracked by following pollination and seed dispersal by animals (Tewksbury et al., 2002). Whether corridors are effective in conservation efforts can be measured by the levels of use by the species for which they were developed. Unintended results of corridor development can also occur. For instance, one study reported that habitat corridors increased predation rates, thus reducing densities of the species of concern (Weldon, 2006). Considerations for construction of corridors include examining habitation success, species fitness, survival and relations.

Anthropogenic disturbance of habitat not only negatively impacts avian communities, but can risk the biodiversity of entire ecosystems. Species diversity is related to the frequency and intensity of disturbance. High diversity is expected when species composition in under constant change. The Intermediate Disturbance Hypothesis (IDH) states that highest diversity is maintained when disturbances are of intermediate scale. Severe disturbance causes communities to return to an earlier stage of succession as pioneer species move into the newly created space. When the colonization period is too short, species diversity remains low (Connell, 1978). However, IDH is not without detractors because it assumes a species trade-off between competitive ability and capacity to tolerate disruption (Jiang et al., 2008) and because it ignores the "competition-colonization" trade–off and evolutionary pressure based on historical frequencies (Cadotte, 2007).

In desert grasslands, wet-dry cycles drive ecosystem processes whereas periodic fire maintains them (Askins et al., 2007). Periodic fires, in a 7-10 year cycle, eliminate cover in the short term and reduce woody species in the long term. Naturally occurring vegetation prone to burning has been lost due to human activity (Askins et al., 2007). Grazing by native herbivores, bison (*Bison bison*), elk (*Cervus elaphus*), white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), and black-tailed prairie dog (*Cynomys ludovicianus*), and other small vertebrates prevents woody shrubs from establishing in natural grasslands. Replacement of such native grazers with domestic cattle negatively impacts native grass species by promoting establishment of woody species (Askins et al*.,* 2007).

Natural desert grassland plains can become fragmented by agriculture, ranching and energy production (oil fields). Dense stands of invasive plant species can also fragment natural ecosystem vegetation. Throughout the past 200 years, changes in land use have caused landscape fragmentation. Woody vegetation has expanded due to the lack of bison grazing and fires, leading to the loss of tallgrass prairies and a dynamic change in ecosystem structure. (Briggs et al*.,* 2005). The native grasslands of the Rolling Plains region of west Texas were

invaded by woody plants such as *Juniperus pinchotii* (Redberry juniper) and *Prosopis glandulosa* (honey mesquite) as the cattle industry moved across it in the late 1800's (McPherson et al*.,* 1988).

Typically, anthropogenic disturbance is considered as additions of physical structures (roads and buildings), damage or chemical pollution and any form of human presence that change native landscapes. A factor receiving less attention is noise pollution. Industrial and traffic noise have adverse effects on habitat quality for birds, which depend on acoustic communication. Avian sound signals have an essential role in territorial defense and male attraction of mates. Noise levels have been reported to reduce the efficiency of both uses of vocalization, which in turn may have a negative effect on reproduction (Slabbekoorn et al*.,* 2003). Some bird species have changed the amplitude, frequency, timing and song length in response to anthropogenic noise over the past few centuries (Slabbekoorn et al*.,* 2003, Fuller et al*.,* 2007). Considering that a portion of robin song is in low frequencies (i.e., 2 - 9 kHz), increases in urban noise levels within this range could impact the transmittability of male robin mating song leading to a compensatory increase in song pitch to improve the male's ability to attract a mate (Fuller et al*.,* 2007).

While increasing their distance from the source is one way that birds and other animals respond to loud noise, it has also been suggested that birds compensate for anthropogenic noise by singing at night when the volume of human-generated noise is reduced (Fuller et al., 2007). Singing at night increases the metabolic demands of diurnal birds relative to sleeping. Fuller's (2007) research addressed the issue of night singing as a strategy for responding to excessive anthropogenic noise. The singing of urban European robins, *Erithacus rubecula*, was surveyed at 121 locations during night and day in order to assess the impacts of light pollution and determine whether birds were more likely to sing at night when the levels of anthropogenic noise were reduced (48 - 51 dB) compared to daytime (56 - 58 dB) in noise hot spots (Fuller et al*.,* 2007). The night periods used in the Fuller et al. (2007) study were 40 minutes prior to

dawn or past dusk. Based on these defined night periods, it was concluded that the effects of daytime noise were more significant on the periodicity of bird song than artificial night lighting. The results of Fuller et al., (2007) suggested that additional night singing was a response to anthropogenic daytime noise that reduced the metabolic energy reserves that birds devoted to competition and reproduction. They commented that the robins in the noisy urban settings shifted their singing to night. However, although Fuller et al., (2007) demonstrated a measurable increase in robin singing at night defined as 40 min before sunrise and after sunset, the study did not account for the commonly known pre-dawn call that occurs 60 minutes before sunrise that is related to male competition in many species of songbirds as well as the increase of singing at dusk (Erne et al., 2008). A related aspect of my proposed research was an evaluation of the possible negative impacts of sound associated with cycling of pump jacks and operation of oil field related equipment on bird abundance, species richness and diversity during daytime hours.

One study has examined the impact of noise levels from oil production sites on several passerine species. This study focused on bird density relative to distance from noise producing compressor stations and oil pump jacks on well pads (Bayne et al., 2008) and involved sampling 5,129 individuals among 56 species of passerine birds. Passerine bird density was maximal at sites near well pads and declined with increasing distance from them. In contrast, density was minimal at compressor sites and increased with distance from them. Their data indicated that the density of passerines was significantly lower near compressor stations compared to all other tested areas, leading to the conclusion that high noise levels associated with compressors inhibited bird use (Bayne et al., 2008). More recently, Lawson et al., (2011) observed bird abundances at different distances from five active oil wells, four abandoned oil well sites and four roadside sites during two consecutive winters to avoid inclusion of breeding migrant species. Like Bayne et al. (2008), they noted that bird abundances increased near

active oil well sites and to a lesser near abandoned oil well sites. In contrast, bird abundance increased with distance from the roads.

Texas is a major agricultural state, where cattle, cotton and oil are top economic industries. While the effects of cattle grazing on grasslands have been well studied, the impacts of land use variations, such as oilfields, on grassland birds have been less well studied. With current increases in exploration for oil and gas in northwest Texas, the region's economy and ecology will need to coexist. Of published research articles on the impacts of petroleum fields combined with other forms of natural and anthropogenic disturbance on bird diversity, the majority appear to have concentrated on marine species. In contrast, only a few works have examined the impacts of oil fields and disturbance on breeding birds in grassland habitats. Therefore, I studied the impacts of petroleum operations and cattle grazing on grassland bird abundance and diversity relative to that in a more natural area in Northwestern Texas. The project focused on an examination of avian communities in the Rolling Plains Ecological Region of northwest Texas as ecological indicators of ecosystem balance in order to understand their impacts on grassland bird biodiversity with respect to these land uses.

Factors involved in oil well disturbance of avian habitats include: 1) initial exploration and construction of wells (disturbance and Intermediate Disturbance Hypotheses); 2) sound disruption from pumps (pumps are on timers, thus do not represent a continual noise factor); 3) human disruption (a pump reader enters a site twice a daily while truck impingement to load oil from on-site storage tanks is less frequent); and 4) containment of salt water generated by the drilling process in large open storage tanks. My study centered on the influence of oil pump jacks on avian species richness and diversity relative to a natural grassland site. In this study, I examined the impact of human disturbance on vegetation cover, as well as its effect on birds. The goal of the study was to examine the impacts of disturbances caused by oil field production and cattle grazing on bird abundance, species richness and species diversity in comparison to an to an undisturbed site in a wildlife reserve. The effect of temperature and precipitation on

avian communities was also examined in order to assess the impact of the severe drought that occurred in the region during 2011. In addition, the impact of vegetation cover on bird abundance, species richness and species diversity was also investigated.

CHAPTER 2

OBJECTIVES

The objectives of this research are as follows.

1. Determine the impacts of land use on bird numbers, species richness, and species diversity for sites in the Rolling Plains Ecoregion of Texas.
2. Determine the impacts of drought (temperature and precipitation) on bird numbers, species richness, and species diversity in the Rolling Plains Ecoregion of Texas.
3. Determine the impacts of anthropogenic disturbance (i.e., oil pump jack operation and cattle grazing) on bird numbers, species richness, and species diversity in the Rolling Plains Ecoregion of Texas.
4 Determine the impacts of vegetation coverage on bird numbers, species richness, and species diversity in the Rolling Plains Ecoregion of Texas.

CHAPTER 3

METHODS

3.1. Description of Field Sites

The study area was composed of four sites in Garza County, Texas, which is southeast of Lubbock, Texas (Fig. 1.1, Figs. 3.1 A-D, Table 3.1), in a vegetation area known as the Rolling Plains (Seyffert, 2001). These sites were selected to encompass four differing land usages, non-grazed, cattle-grazed, oil field and mixed oil field/cattle-grazed sites. Each site had vegetation differences due to due to variation in use and environmental factors.

Land access permits were obtained for each site. Site A (Figs. 1.1 and 3.1A., Table 3.1), with an area of approximately 1.20 km^2, was subject to mixed land use as an oil field (established in 1982) with sixteen wells and cattle grazing. Site B (Figs.1.1 and 3.1B, Table 3.1), with an area of approximately 3.02 km^2, was used as an oil field from 1984 to present, and included land previously used as a public golf course from 1954 to 2001. After sale of the land in 2003, this site was used as oil field. Site C (Figs. 1.1 and 3.1C, Table 3.1) had an area of approximately 1.44 km^2 and was used exclusively for cattle grazing. Site D (Figs. 1.1 and 3.1D, Table 3.1), the Sam Wahl Wildlife Mitigation Area, included approximately 1457 ha. It was privately owned until 1991 and used for cattle grazing and farming. Upon completion of Lake Alan Henry in 1993, it was designated as a wildlife mitigation area after which it has remained undisturbed except for limited issue of hunting permits on specified dates for deer, mourning dove and quail in season, based on species densities (City of Lubbock Parks and Recreation Department, 2012). Hunting season in Texas for these animals was from September through April. The only hunting that occurred from May through August was for feral hogs, which was

only by permit on weekend (City of Lubbock Parks and Recreation Department, 2012). This site bordered the northeastern shore and John Montford Dam of Lake Alan Henry (Permitted by the City of Lubbock). Ten sub-sites where randomly established within each of these four sites (Fig. 3.1A-D) and their locations recorded with GPS (see Table 3.1 for sub-site coordinates).

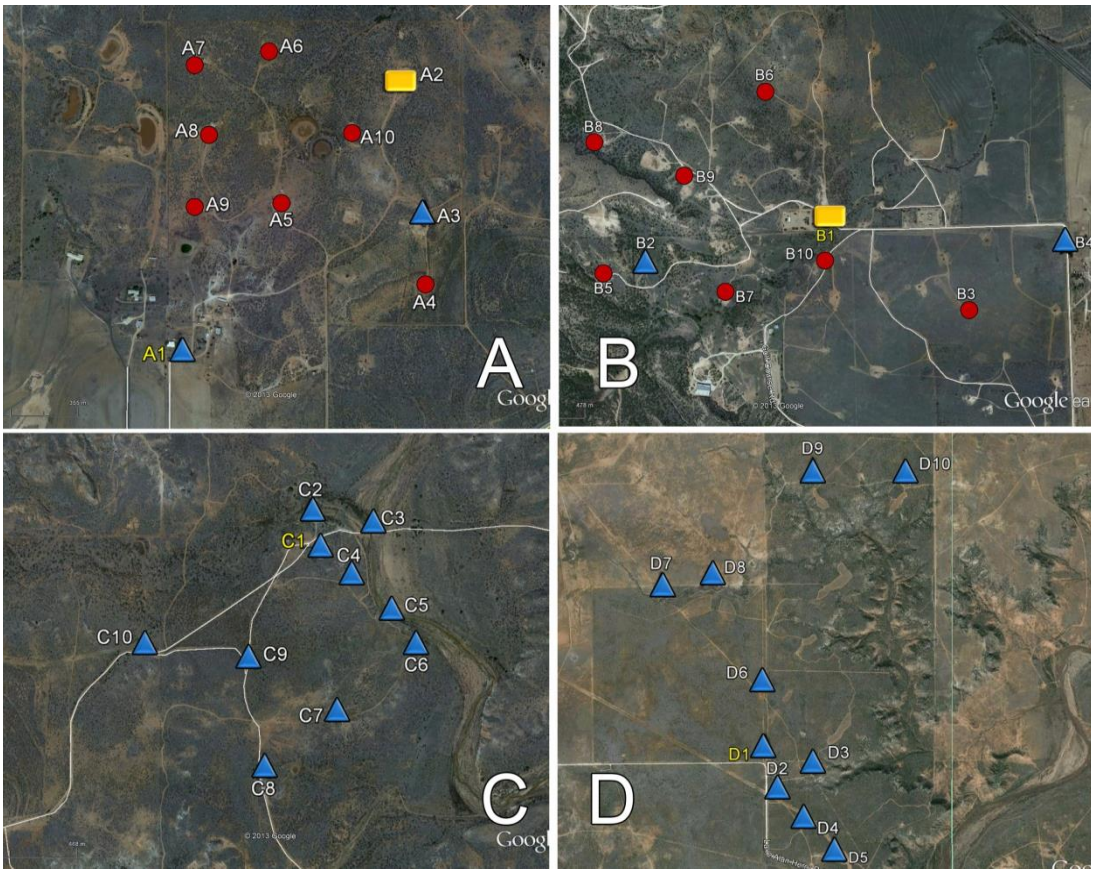

Figure 3.1. Aerial Images of field research sites from Google Earth®. All sites were located in Garza County, Texas. Each site had ten sub-sites for the study. A) Site A was used for oil field and cattle grazing. B) Site B was was used as an oil field. C) Site C was used for cattle grazing. D) Site D was a wildlife mitigation area adjacent to Lake Alan Henry. In all figures, sub-sites with oil well pads are indicated by red dots, oil storage tanks with yellow rectangles, and sites without oil well pads or storage tanks by blue triangles.

Sites A and B included seven sub-sites centered on oil well pads, one centered on an oil storage tank, and two centered on those without oil well pads or storage tanks (Table 3.1, Figs. 3.1A and B). Sites C and D did not have oil well pads or storage tanks such that all sub-sites were centered on areas without these structures. Within each site, the ten sub-sites were

identified by GPS data and notation of land features (Table 3.1, Figs. 3.1A-D). Maps were prepared of these four major sites and their sub-site locations by imputing GPS data into Google Earth to ensure accurate site location identification in the field. The maps were useful, especially in the case of locating sub-sites on sites C and D (Figs. 3.1.C and D), which did not have oil well pads or storage tanks to mark their locations.

Table 3.1. Geographic coordinates of the forty sub-sites. The GPS data was recorded in decimal degrees for use in ArcGIS (ESRI, 2011) and Google Earth programs for further analysis of the areas.

Sub-site	Latitude	Longitude	Sub-site	Latitude	Longitude
A1	33.196312°	-101.360526°	C1	33.233609°	-101.328936°
A2	33.202951°	-101.354021°	C2	33.234806°	-101.329238°
A3	33.199583°	-101.353571°	C3	33.234321°	-101.327134°
A4	33.198048°	-101.353578°	C4	33.232828°	-101.327852°
A5	33.200007°	-101.357640°	C5	33.231678°	-101.326307°
A6	33.203677°	-101.357980°	C6	33.230629°	-101.325424°
A7	33.203344°	-101.360145°	C7	33.228745°	-101.328313°
A8	33.201651°	-101.359706°	C8	33.226907°	-101.330924°
A9	33.199885°	-101.360090°	C9	33.230202°	-101.331389°
A10	33.201703°	-101.355678°	C10	33.230716°	-101.335148°
B1	33.226879°	-101.418151°	D1	33.076081°	-101.057509°
B2	33.225352°	-101.425258°	D2	33.072467°	-101.056333°
B3	33.224021°	-101.412551°	D3	33.074667°	-101.052700°
B4	33.225673°	-101.408574°	D4	33.069867°	-101.053767°
B5	33.225112°	-101.426864°	D5	33.067324°	-101.050111°
B6	33.231283°	-101.420589°	D6	33.081417°	-101.057733°
B7	33.224490°	-101.422065°	D7	33.089750°	-101.067883°
B8	33.229467°	-101.427236°	D8	33.090124°	-101.060764°
B9	33.228310°	-101.423716°	D9	33.099427°	-101.052667°
B10	33.225673°	-101.418300°	D10	33.098967°	-101.043450°

Site A was centered on sub-site A1 at Latitude 33.196312°, Longitude -101.360526° (Table 3.1, Fig.3.1A). Its geographic position relative to the other sites was 8.37 km southeast of Site B, 9.71 km southwest of site C, and 42.97 km northeast of Site D (Fig. 1.1). Structures on this site included two homes, four storage buildings, four barn structures 14 pump jacks and

8 structures related to storage of oil and two of water waste. The entire site was fenced with barbed wire to contain cattle. Three small ponds on this site provided water for cattle and wildlife along with three ponds on adjacent land west of the site (Fig. 3.1A). The majority of the site had a combination of woody vegetation and native grasses. The south portion of this site was flat, with minimal woody coverage, which was better suited for cattle grazing. Although this site had large flat areas, elevation changes included slopes and ravines (Fig. 3.1A).

Site B was centered on sub-site B1 at Latitude 33.226879°, Longitude -101.418151° (Table 3.1, Fig 3.1B). Its geographic position relative to the other sites was: 8.37 km northeast of Site A, 12.87 km west of site C, and 48.1 km northeast of Site D (Fig. 1.1). Elevations in the site ranged from 885.13 m to 807.72m. A series of cliffs were located near sub-sites 8B and 9B. A long ravine was located on the east side of sub-site B5 (Fig. 3.1B). Vegetation on the southeastern portion of this site was predominately grasses, with sparse woody vegetation. Land to the north and west had a higher concentration of woody vegetation.

Site C was centered on sub-site C1 located at Latitude 33.233609°, Longitude -101.328936° (C1) (Table 3.1, Fig. 3.1C). Its geographic position relative to the other sites was: 9.71 km northwest of Site A, 12.87 km east of site B and 48.1 km northwest of site D (Fig.1.1). The eastern side of this site was bordered by the Double Mountain North Fork of the Brazos River. When research was initiated in 2008, the river was permanently flowing and supported fish populations. During July 2012, flow was intermittent with scattered patches of water along an exposed sandy river bottom. A small pond, located near sub-site C8, was the only other natural water body on this site (Fig. 3.1C). Structures on this site were limited to a house (occupied in 2009-2011, but not occupied in 2012) and six unused barns.

Site D was centered on sub-site D1 located at Latitude 33.076081°, Longitude -101.057509° (Table 3.1.1., Fig 3.1D). The distance and directional relation to the other sites was 42.97 km southeast of Site A, 49.86 km southeast of Site B, and 48.1 km southeast of Site

Table 3.2. Plants observed on Garza County Sites

Common name	Scientific name
Silver beardgrass	*Bothriochloa laguroides spp. Torreyana*
Sideoats grama	*Bouteloua curtipendula var. curtipendula*
Hairy grama	*Bouteloua hirsuta*
Texas grama	*Bouteloua regidiseta*
Hooded windmill grass	*Chloris cucullata*
Canada wild rye	*Elymus canadensis*
Plains love grass	*Eragrostis intermedia*
Curly mesquite grass	*Hilaria belangeri*
Tobosa	*Hilaria mutica*
Texas bluegrass	*Poa arachnifera*
White tridens	*Tridens albescens*
Catclaw acacia	*Acacia greggii*
Western ragweed	*Ambrosia cumanensis Kunth in H.B.K.*
Western ragweed	*Ambrosia psilostachya*
Agarito	*Berberis trifoliolata*
Netleaf hackberry	*Celtis laevigata*
Golden Corydalis	*Corydalis aurea*
Cholla	*Cylindropuntia imbricata (Haw.)*
Broom snakeweed	*Gutierrezia sarothrae*
Common sunflower	*Helianthus annuus L.*
Gray goldaster	*Heterotheca canescens (DC.) Shinners*
Rayless goldenrod	*Isocoma wrightii (Gray) Rydb.*
Ashe juniper	*Juniperus ashei Buchholz*
Oneseed juniper	*Juniperus monosperma*
Redberry juniper	*Juniperus pinchotii Sudworth*
Allthorn, Crown of Thorns	*Koeberlinia spinosa*
Bois d'arc, Osage orange	*Maclura pomifera (Raf.) Schn.*
Catclaw mimosa	*Mimosa biuncifera Benth.*
Prickly pear Cactus	*Opuntia engelmannii*
Honey mesquite	*Prosopis glandulosa Torr. var. glandulosa*
Little leaf sumac	*Rhus microphylla*
Lanceleaf salvia	*Salvia reflexa*
Western soapberry	*Sapindus drummondii*
Western horse nettle	*Solanum dimidiatum Raf.*
Silverleaf nightshade	*Solanum elaeagnifolium Cav.*
Saltcedar, Tamarisk	*Tamarix ramosissima Ledeb.*
Goathead , Puncturevine	*Tribulus terrestris*
Narrowleaf yucca	*Yucca angustifolia*
Soapweed yucca	*Yucca glauca*
Lotebush, Gumdrop tree	*Zizyphus obtusifolia*

C (Fig. 1.1). Geological features included cliffs and ravines on the eastern portion of the site. This entire site was fenced, with locked access gate points. Structures on this site included a building used for storage and site management on sub-site D1. Sub-site D2 had a water-pumping windmill and contained a pond. Sub-site D3 had the remains of a cattle chute and fencing. The northern edge of sub-site D8 had barbed wire fencing on the border of the mitigation area (Fig. 3.1D). The southwestern edge of this site was bordered by Lake Alan Henry. Other water on the site included two natural ponds and one man-made pond.

3.2. *Vegetation Analysis*

Within each major site, vegetation coverage and species composition was determined for a one hectare area centered on each of the 10 sub-sites. The plant species occurring on each sub-site were visually identified.

3.2.1. *Vegetation species identification*

A list of common plants of the Rolling Plains was composed from sources including Texas Parks and Wildlife Plants of the Rolling Plains (2009) and the Texas A&M Forest Service (2008). Woody species were identified using the list and field guides specific to Texas or the Rolling Plains vegetation region of Texas (Texas A&M Forest Service, 2008, Lady Bird Johnson Wildflower Center, 2012). Grass samples were either collected, photographed or both to confirm species. Grass species were identified with the appropriate guides including those published by the Gould et al. (1978), Botanical Research Institute of Texas (2012) and the Lady Bird Johnson Wildflower Center (2012). Further verification was provided by Glen Killough (personal communication, 2010), District Conservationist, USDA (United States Department of Agriculture), Post, Texas. For a list of identified plants species across sub-site see Table 3.2.

3.2.2. *Percent Cover*

Percent cover was determined for each sub-site using Google Earth satellite images taken on 03/31/2012 that were quantified with Adobe Photoshop CS6 using image analysis. Prior to creating images from Google Earth, settings in the options features were adjusted for

optimal quality. These settings included “Show Lat/ Long: Decimal Degrees”, “Terrain Quality: Higher”,

Figure 3.2. Percent woody vegetative cover image analysis on sub-site C2.This figure is an example of the procedure used to determine percent vegetative cover on a sub-site using images from Google Earth© and Adobe Photoshop CS6©. The Google Earth image of Sub-site C2 was divided into 318 quadrats by overlaying it with a white grid. Completely vegetation-filled quadrats on Sub-site C2 were delineated in blue, each with an assigned unique number. Contiguous partially filled quadrates equivalent in coverage to a fully filled quadrat were delineated with a red line and marked by a unique number. The top portion of the figure shows the count indicator for fully filled, “Count Group 1” (cyan numbers) and partially filled quadrats “Count Group 2” (yellow numbers). Note the count (“Count :”) of fully and combined partially filled quadrates [“Count: 93(75)”] where 93 was the total number of filled quadrats and 75 the number of combined partially filled quadrats yielding 18 fully filled quadrates (i.e., 93-75 = 18). Thus, there were 93 fully covered quadrats or combined quadrats with enough vegetation to completely fill a single quadrat out of a total of 318 quadrats over the entire sub-site so that percent cover could be computed as (93/318) x 100 or 29.2%.

“Map Size: Large, “Label Size Icons: Small”, “Texture colors: True Color 32 Bit” and “Navigation: Do not automatically tilt while zooming.” These images were used to evaluate percent coverage of woody species for each sub-site. The GPS coordinate of each sub-site, in decimal degree format, was entered onto each sub-site map with the “Add Placemark” Google Earth tool.

GPS data were entered into Google Earth® to image the forty sub-site locations from a standard elevation of 945 m with a scale of 55 m, where North was at the top of the image area. The images were captured as jpeg (Joint Photographic Expert Group) files in 16 bit RGB color with a standard resolution of 300 dpi (dots per inch). East-west and north-south dimensions for each sub-site rectangular image were determined. In order to standardize sub-site images for percent cover analysis, their 16 bit RGB color and 300 dpi features were preset and saved in Adobe Photoshop CS®. The images were then opened in Adobe Photoshop CS®. Each sub-site image was created in three layers. The top layer included a 25.5 X 12.5 grid (8mm x 7mm per grid) which was overlaid on the standard sized sub-site images generating 318 quadrats (Fig. 3.2) with a north-south height of 277 m and an east-west length of 132 m. In order to estimate vegetative ground cover, vegetation was visually examined in each delineated grid quadrat. To be counted as fully covered with woody vegetation an individual grid rectangle had to appear to be approximately greater than 75% covered with dark green vegetation Fig. 3.2), indicating leafed-out evergreens and oaks and/or a textured gray color indicative of still leafless mesquites in March.

The Adobe Photoshop CS® count tool was then used to sequentially number all “completely filled” quadrats (Fig. 3.2). Counts of filled grids were initially made left to right on the top grid line of the image, and then subsequently repeated for the next line below until all filled grid sections had been counted. Subsequently, the pen tool with a blue-colored line was used to outline all completely filled quadrats on the sub-site (Fig. 3.2). The completely filled quadrats were saved in Photoshop as a count in the image file. After counting and color delineating “completely filled” areas, the subsequent evaluation involved estimating vegetation areas in all partially filled quadrats. Where covered regions of a quadrat were difficult to interpret, the image was enlarged to enhance cover area identification. Contiguous partially covered quadrats were outlined with a pen tool, in red, such that enough quadrats were included to contain vegetative

cover equivalent to a single fully filled quadrat. This process was iteratively repeated until all partially covered quadrats were included in the analysis. Each combined set of partially covered quadrats were then counted as a single fully covered quadrat. After all quadrats with partial cover by woody vegetation were so analyzed, grouped quadrats representing a single fully covered quadrat were stored in a separate counting file. Upon completion of all quadrat counts, the total count number was displayed in the top left toolbar of the program. The summed total of covered quadrats were then divided by the total number of sub-site quadrats (318) and multiplied by 100 to give a percent estimate of woody vegetation cover for each of the 40 sub-sites (Figs. 3.2).

Similar methods were used to create estimates of total vegetation. Sub-site images were obtained from Google Earth from the time periods 2008 and 2012. Images were analyzed by making counts of bare ground and anthropogenic structures and then subtracting those values to obtain estimated total vegetation for all sub-sites for in the years 2008 and 2012. The comparison of these two data sets was used to assess the impacts of the extreme drought of 2011 on pre- (i.e., 2008) versus post-drought (i.e., 2012) vegetation cover on the sites.

3.3. Bird Data

3.3.1. Bird Count Methodology

Prior to performing bird counts, a list of the expected birds was developed from that of (Seyffert, 2001) specifically for the Texas Rolling Plains during May to August, corresponding to my field observation period. All field assistants were required to review images and behaviors of the birds on this list from field guide books (Peterson, 2002; Sibley, 2003). In addition, iPad® and iPhone® apps (Audubon Nature Texas, 2011; iBird Pro, 2012; Sibley, 2011) allowed bird visual and call identification in the field. Bird counts were taken during breeding/summer season (June through July) during 2008, 2009, 2010 and 2012. Counts were not made in 2011 at the request of property owners due to exceptional drought and resulting high fire risk. Counts were made only at sites A, B and C in 2008. Site D was added in 2009. In 2010, bird counts

were undertaken only at Sites A, B and D because the land encompassing Site C came under new ownership and the out-of-state owner could not be contacted to gain permission for access the site. Bird counts were conducted by two observers (i.e., the researcher and either of two field assistants) over a 30 min period at each sub-site both in the morning at 6:45 AM to 12:00 PM in the morning and in the evening from 6:30 to 9:15 PM. Data from morning and evening counts were combined for subsequent analyses. The order in which counts were made on sub-sites in any one year was randomized across years of the study such that sub-site counts were not taken at same time during each annual visit to a site. Observers positioned themselves back-to-back within approximately 10 m of the GPS center of the observation area with care being taken to prevent observer overlap. Observers counted all birds within a range permitting clear identification with binoculars (typically 30 to 50 meters with Canon Image Stabilizer® 12 x 36 5°, Nikon Action® 10 x 50 6.5° and Celestron® 10 x42 7° depending on bird size and behavior). In some cases aural localization of bird call was used to find and then visually identify a bird. However, bird calls, even if they could unambiguously identified to species, were not used in counts with visual identification of the calling bird being required because counts based only on bird call identification could result in an over-count of a bird species if a single bird was moving through the sub-site and making calls from different locations. Bird counts were conducted from the periphery of oil well pads and storage tank sites. Presence or absence of well pump jack operation during counting was noted.

All bird counts were entered into field notebooks and Excel® data sheets. Species' common and scientific names and sub-site survey dates and times for listed birds in the study were recorded in the first two columns, followed sequentially by the total counts for all sites, then counts for each of the 10 sub-sites. All data was totaled by site and for each of the sub-sites. Species richness and species diversity were then computed for each sub-site and for combined data from all ten sub-sites within a site. Shannon Weiner Diversity index was the

species diversity index used in my analyses, where *S* is the total number of species given and *p* is the frequency of species in the community *i* (Lande, 1996).

The Shannon Weiner Diversity index formula:

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

3.3.2. Climate Data

Other data recorded during bird observation periods included, time of day and weather conditions. Detailed precipitation and weather data were collected from the West Texas Mesonet site (Schroeder, 2012), for Post and Lake Alan Henry including precipitation, air temperature and wind speed continuously recorded at 5 minute intervals. Historical weather data beyond the range available from the West Texas Mesonet were accessed from the Southern Regional Climate Center (2013) and Weather Warehouse (Gibbas et al., 2013). All temperature and precipitation data were converted to degrees Celsius and centimeters, respectively.

3.4. Breeding Bird Survey

3.4.1. Breeding Bird Survey Data

Individual bird counts and species numbers were accessed from the Breeding Bird Survey (BBS route) # 83092, for Post, Texas, established in 1969. This route began west of the City of Post, TX, in Lynn County. The route along FM 651 was approximately one kilometer west of site A and 2 kilometers west of site C and 8 km east of Site B (Fig. 3.3), but was more distant to D. The landscape adjacent to the route included a broad snapshot of habitat across the county: agricultural, abandoned agricultural, residential, downtown, commercial, cattle-grazed and unimproved lands very similar to that encompassed by the four study sites. Point count data for these routes is combined, therefore eliminating the possibility of discriminating between various land uses. With the exception of a few years, seven different observers have

made annual counts on this route between 1969 and 2011 (Sauer, 2012). Individual counts totaled 37,973 birds. Eighty seven bird species were counted and reported for the Post, TX BBS Route from 1969 to 2011. A breakdown of these species by migration status was 25 permanent residents, 28 short distance migrants and 34 Neotropical migrants (Gough et al., 1998). Based on breeding habitat group designations the species included 19 grassland species, 19 successional-scrub species, 16 open-woodland species, 15 wetland-open/water species and 18 urban species (Gough et al., 1998).

Figure 3.3. Breeding Bird Survey Route 83092, Post, Texas. BBS route began in Lynn County and continued through Garza County for 39.43K. Number of birds observed along this route was reported for 37 years for a period of 1969 to 2011. These data were used by the US Geological Survey to create species abundance maps and other analyses including migration patterns.

Count and species data for the BBS Post Route were acquired from the USGS Patuxent Wildlife Research Center website (http://www.mbr-pwrc.usgs.gov/) under the section "The North American Breeding Bird Survey Summary and Analysis 1966 - 2011." Raw data for point species counts are available at this site; however, the site does not have a search feature or clear description of how to find them. In order to retrieve raw data it was necessary to first go

the webpage that provided the list of all species encountered in any year of the survey. The link to the data for the Post route 83092 (Fig. 3.3) was http://www.mbr-pwrc.usgs.gov/cgi-bin/rtena211.pl?83092. On this page a table of all species was listed, along with other topics that were provided for each species. Accessing the link on the line across from a species," Route Change" opened another page, "Route Population Change Display" on which trend estimate analyses were shown. On this page, the link to the raw data was, "want to see the raw data? Click HERE." Each page of the raw data included the species number, the region and a three columned table with the year, count and observer number. Species data from this page was copied onto a notepad page, which was then opened in an Excel document. This process was repeated for all 87 species included in the survey range for 1969 to 2011. These data were the raw data from the point counts for each year of the Breeding Bird Survey (BBS) (http://www.mbr-pwrc.usgs.gov/cgi-bin/rtena211.pl?83092.) Data for the Post, Texas, route ranged from 1969 – 2011, with several years omitted. In order to create a complete list of all species throughout this time span, data for each species was individually download to a notepad file and then added to an Excel spread sheet. Species abundance, species richness and species diversity (i.e., Shannon Weiner Diversity Index (H'), see formula above) were calculated for each of the 37 years of data**.**

3.4.2. Temperature and Precipitation Data

Historical weather data across the years for which BBS data was available were retrieved from the Weather Warehouse website by accessing several separate pages sorted by city and month (Gibbas, 2013). Weather trend data was used with route data in a regression analysis to detect possible effects of the 2011 extreme drought year. Weather data used in this analysis were, for each year; average high temperature (C°) in June, sum of precipitation from January to June, bird abundance, species richness and species diversity.

3.5. Statistical Analyses

3.5.1. Rank Abundance of Bird Species by Site

Data for bird species counts were combined across all 10 sub-sites at each site for the research years of 2008, 2009, 2010 and 2012. The most numerous bird species in each site for each year of the study were determined by computing the fraction of species individuals relative to the total bird count. Species were listed in order of decreasing fraction of total count for each site on research year until a sum of ≥ 75% of the total bird count was attained or just surpassed.

3.5.2. Analysis of Bird Abundance, Species Richness and Diversity Relative to Environmental Factors

3.5.2.1. Mixed Model Analysis of Covariance

Statistical analysis of the data was developed with the guidance of Dr. James Grover (Department of Biology, The University of Texas at Arlington). Distributions observed by ecologists vary in that abundance data include absence and tend to have non-normal distributions; being commonly distributed with a strong skew to the right (Legendre and Legendre, 1984). The bird count and environmental data for the four sites was analyzed using a mixed model ANCOVA, modified, for the measure of years, fixed effects of presence/absence of oil pump jacks and cattle grazing on sub-sites, with percent sub-site woody vegetation cover as a covariate. In order to account for the lack of balance in survey design, the ANCOVA involved responses averaged over years and additionally constructed time-dependent contrasts as described below that accounted for time-dependent effects without violating the assumption of independent observations (Dr. James Grover, personal communication).

The analyses were based on bird abundance, species richness and species diversity (H') values at the forty sub-sites across the years of the study. Bird counts were not performed on Site C in 2010 or on Site D in 2008, therefore because of these missing data, a standard repeated-measures analysis could not be done. Three response variables were created for

each of these data sets. These three response variables were constructed to make contrasts over time that could be analyzed as independent observations in a standard ANCOVA. Response one in each case was the average of all sub-site data for all years 2008 – 2010 and 2012. Response two was pre-drought years, 2008 - 2010, averaged data minus that for post-drought 2012 data. Response three was based on the years with full data four all four sites (Sites A, B, C and D) computed as pre-drought 2009 data minus the post-drought 2012 data. The two effects utilized in the analysis were presence (1) or absence (0) of oil pump jacks or cattle on each sub-site with a covariate of sub-site percent woody vegetation cover. The intent of constructing these response variables was to make contrasts of pre- and post-drought data that could be treated as independent observations, despite the fact that the survey design was unbalanced. This approach may not completely eliminate potential effects of imbalanced data, but was an expedient way to minimize such effects while conducting as thorough an analysis as possible.

ANCOVA analysis for bird abundance, species richness and species diversity on sub-sites, for the three response variables was performed with the statistical software, Statistica© (2010) using a model allowing interactions between covariates and factors to be tested. Presence/absence of oil or cattle were factors in the analysis, and percent woody vegetation cover, the covariate, along with the interactions of oil pump jacks by cattle, oil pump jacks by percent cover, cattle by percent cover and oil pump jacks by cattle by percent cover.

3.5.2.2. Comparison of Bird Abundance, Species Richness and Diversity among Sites in 2009 and 2012

Differences in sub-site bird abundance, species richness and species diversity (H') as dependent variables were examined by Multifactor ANOVA with site and years (pre-drought 2009 versus post-drought 2012) as the main effects. The years 2009 and 2012 were chosen for the analysis because they represented a complete data set in which data were available for all

four sites. A *post-hoc* Fisher's Least Significant Difference (LSD) procedure was used to determine differences among sub-site means at $p < 0.05$ for the three variables across years and sites.

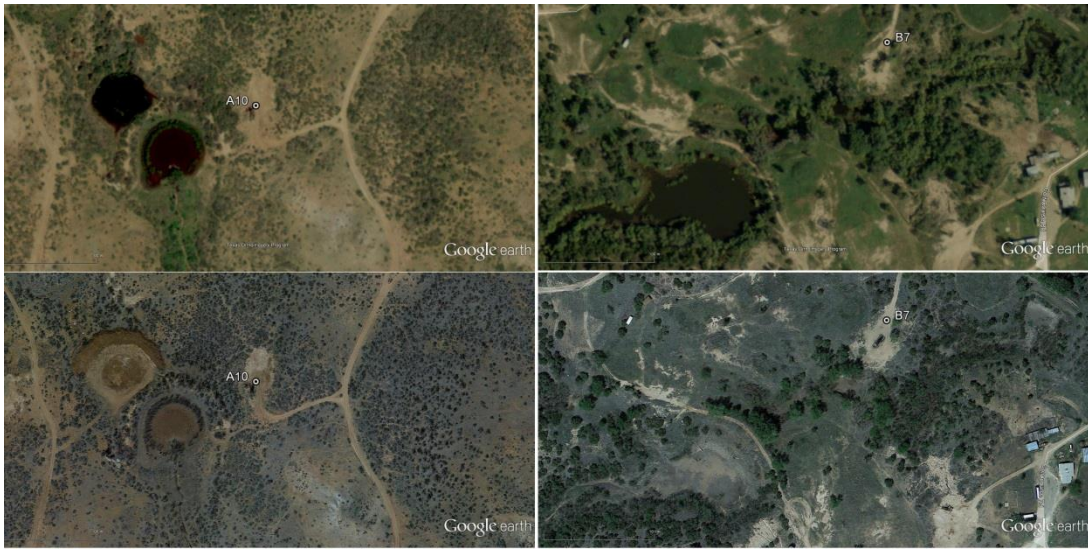

Figure 3.4. Satellite images of sub-sites A10 and B7 during the pre-2011-drought year of 2008 and post-drought year of 2012. A) Sub-site A10 imagery date 3/31/2008. B) Sub-site B7 imagery date 3/31/2008. C) Sub-site A10 imagery date 3/29/2012 D) Sub-site B7 imagery date 3/29/2012. These sub-site images clearly indicate the impacts of the 2011 drought on vegetation and water sources.

3.5.2.3. Pre- and Post-Drought Changes in Sub-site Percent Vegetation Cover

The difference in mean sub-site percent total vegetative cover (including grasses) for the pre-drought year, 2008, versus post-2011 drought year of 2012 were examined with a non – parametric Kolmogorov-Smirnov (K-S) due to data departing from normality. The years 2008 and 2012 were chosen for the analysis because they were periods in which aerial land images could be accessed from Google Earth and were available for all four sites (Fig. 3.4). These time periods (i.e., 2008 and 2012) included the initial and final years of the research period and bracketed the 2011 year of extreme drought in the study area.

3.5.2.4. Correlation between Sub-site Percent Vegetation Cover and Bird Abundance, Species Richness and Species Diversity

Separate Least Squares Linear Regression analyses were utilized to determine if there was any relationship between sub-site percent vegetation cover data (woody shrubs and grasses) combined for both 2009 and 2012 and the dependent variables of sub-site bird abundance, species richness and species diversity (H'). These two years were pre- and post-the 2011 drought which resulted in a major reduction in vegetation cover at all sites in 2012. Percent vegetation cover was estimated for the sub-sites from Google Earth aerial images available in 2008 which were paired with 2009 bird data and 2012 which were paired with 2012 bird data.

3.5.2.5. Analysis of Breeding Bird Survey Data

Separate Least Squares Linear Regression analyses were used to investigate the correlation between the City of Post, TX, Breeding Bird Survey (BBS) Route yearly bird count numbers, species richness and species diversity indices (H') data versus average June high daily temperatures and January-June total precipitation values as independent variables. Initially, bird numbers, species richness and species diversity (H') values were tested against either temperature or precipitation data for the same year on which bird data was recorded. However, because the impacts of temperature during and precipitation prior to bird breeding can have latent effects on bird communities, regression analyses were also conducted for the three dependent variables versus temperature and precipitation data for the year prior to the year on which the dependent variables were recorded.

CHAPTER 4

RESULTS

4.1. Rank Abundance of Bird Species by Site

Qualitative analysis of ranks by site indicated that the proportions of the bird community made up by specific species were variable across years (Table 4.1). However, despite changes in rank at each site across years, the same species of birds tended to make up the top 75% of the total bird population at each site (Table 4.1) for the years 2009 and 2012 when data for all four study sites was available the mean fraction of the site bird count for sites by year was 0.1465 (n = 8, s.d. = 0.0204) with the range across all sites and years being 0.1190 to 0.1701 indicative of a lack dominance by any one species. Thus, the mean number of species making up 75% of the total counted birds at all sites in 2009 and 2012 was 12.6250 (n = 8, s.d. = 2.7223, range = 8 - 16) again suggestive of relatively high levels of species diversity on all study sites.

The relatively high diversity of birds at the study sites was further documented by the mean values for total site number of birds, species richness and species diversity index (H') (n = 14) across all four years of the study (Table 4.2). Across all four years of the study the overall mean for number of individuals was 237.78. s.d. = 95.84, range = 88 - 411, for species richness, 32.57, s.d. = 7.02, range = 16 - 43 and for species diversity index (H'), 2.936, se = ±0.308, range = 2.116-3.280 (Table 4.2).

Table 4.1 Fraction of total bird numbers of species in the total 75% of birds counted on Sites A, B, C and D during 2009 and 2012.

Site A 2009		
Species Common Name	Scientific name	Fraction
Common Grackle	*Quiscalus quiscula*	0.1507
Bullock's Oriole	*Icterus bullockii*	0.0993
Curve-billed Thrasher	*Toxostoma redivivum*	0.0719
Western Kingbird	*Tyrannus verticalis*	0.0616
Barn Swallow	*Hirundo rustica*	0.0548
Cassin's Sparrow	*Aimophila cassinii*	0.0548
Lark Sparrow	*Chondestes grammacus*	0.0548
Greater Roadrunner	*Geococcyx californianus*	0.0445
Mississippi Kite	*Ictinia mississippiensis*	0.0411
Mourning Dove	*Zenaida macroura*	0.0342
Red-winged Blackbird	*Agelaius phoeniceus*	0.0308
Northern Mockingbird	*Mimus polyglottos*	0.0240
Site A 2012		
Species Common Name	Scientific name	Fraction
Mourning Dove	*Zenaida macroura*	0.1699
House Sparrow	*Passer domesticus*	0.1090
Lark Sparrow	*Chondestes grammacus*	0.0929
Cassin's Sparrow	*Aimophila cassinii*	0.0833
Great-tailed Grackle	*Quiscalus mexicanus*	0.0737
Western Kingbird	*Tyrannus verticalis*	0.0737
White-winged Dove	*Zenaida asiatica*	0.0417
Grasshopper Sparrow	*Ammodramus savannarum*	0.0353
Scissor-tailed Flycatcher	*Tyrannus forficatus*	0.0321
Common Grackle	*Quiscalus quiscula*	0.0321
Northern Mockingbird	*Mimus polyglottos*	0.0288
Ash-throated Flycatcher	*Myiarchus cinerascens*	0.0256

Table 4.1 Continued

Site B 2009		
Species Common Name	Scientific name	Fraction
Northern Mockingbird	*Mimus polyglottos*	0.1575
Scissor-tailed Flycatcher	*Tyrannus forficatus*	0.0984
Lark Sparrow	*Chondestes grammacus*	0.0906
Wild Turkey	*Meleagris gallopavo*	0.0827
Western Kingbird	*Tyrannus verticalis*	0.0748
Mourning Dove	*Zenaida macroura*	0.0709
Bullock's Oriole	*Icterus bullockii*	0.0472
Ash-throated Flycatcher	*Myiarchus cinerascens*	0.0433
Brown-headed Cowbird	*Molothrus ater*	0.0394
White-winged Dove	*Zenaida asiatica*	0.0315
House Finch	*Carpodacus mexicanus*	0.0315
Bullock's Oriole	*Icterus bullockii*	0.0472
Ash-throated Flycatcher	*Myiarchus cinerascens*	0.0433
Brown-headed Cowbird	*Molothrus ater*	0.0394
White-winged Dove	*Zenaida asiatica*	0.0315
House Finch	*Carpodacus mexicanus*	0.0315

Site B 2012		
Species Common Name	Scientific name	Fraction
Mourning Dove	Zenaida macroura	0.1701
Lark Sparrow	Chondestes grammacus	0.1443
Wild Turkey	Meleagris gallopavo	0.0670
Scissor-tailed Flycatcher	Tyrannus forficatus	0.0670
Turkey Vulture	Cathartes aura	0.0515
Northern Mockingbird	Mimus polyglottos	0.0515
White-winged Dove	Zenaida asiatica	0.0464
Ash-throated Flycatcher	Myiarchus cinerascens	0.0464
Cassin's Sparrow	Aimophila cassinii	0.0361
Barn Swallow	Hirundo rustica	0.0309
Northern Cardinal	Cardinalis cardinalis	0.0258
Bullock's Oriole	Icterus bullockii	0.0258
House Sparrow	*Passer domesticus*	0.0258

Table 4.1 Continued

Site C 2009		
Species Common Name	Scientific name	Fraction
Red-winged Blackbird	*Agelaius phoeniceus*	0.1531
Scissor-tailed Flycatcher	*Tyrannus forficatus*	0.1327
Common Grackle	*Quiscalus quiscula*	0.0816
Scaled Quail	*Callipepla squamata*	0.0714
Mourning Dove	*Zenaida macroura*	0.0714
Lark Sparrow	*Chondestes grammacus*	0.0612
Mississippi Kite	*Ictinia mississippiensis*	0.0510
Northern Mockingbird	*Mimus polyglottos*	0.0510
Ash-throated Flycatcher	*Myiarchus cinerascens*	0.0408
Killdeer	*Charadrius vociferus*	0.0408
Site C 2012		
Species Common Name	Scientific name	Fraction
Cassin's Sparrow	*Aimophila cassinii*	0.1119
Northern Bobwhite	*Colinus virginianus*	0.0970
Mourning Dove	*Zenaida macroura*	0.0970
Northern Cardinal	*Cardinalis cardinalis*	0.0672
Lark Sparrow	*Chondestes grammacus*	0.0597
Common Nighthawk	*Chordeiles minor*	0.0522
Bank Swallow	*Riparia riparia*	0.0522
Bullock's Oriole	*Icterus bullockii*	0.0522
Scissor-tailed Flycatcher	*Tyrannus forficatus*	0.0448
Northern Mockingbird	*Mimus polyglottos*	0.0448
Golden-fronted Woodpecker	*Melanerpes aurifrons*	0.0373
Mississippi Kite	*Ictinia mississippiensis*	0.0299
Pyrrhuloxia	*Cardinalis sinuatus*	0.0299
Painted Bunting	*Passerina ciris*	0.0299
Brown-headed Cowbird	*Molothrus ater*	0.0299

Table 4.1 Continued

Site D 2009		
Species Common Name	Scientific name	Fraction
Cliff Swallow	*Petrochelidon pyrrhonota*	0.1264
Scissor-tailed Flycatcher	*Tyrannus forficatus*	0.0824
White-winged Dove	*Zenaida asiatica*	0.0714
Lark Sparrow	*Chondestes grammacus*	0.0549
Turkey Vulture	*Cathartes aura*	0.0495
Ash-throated Flycatcher	*Myiarchus cinerascens*	0.0495
Northern Cardinal	*Cardinalis cardinalis*	0.0440
Red-winged Blackbird	*Agelaius phoeniceus*	0.0440
Bullock's Oriole	*Icterus bullockii*	0.0440
Northern Bobwhite	*Colinus virginianus*	0.0385
Western Kingbird	*Tyrannus verticalis*	0.0385
Cassin's Sparrow	*Aimophila cassinii*	0.0385
Northern Mockingbird	*Mimus polyglottos*	0.0330
Curve-billed Thrasher	*Toxostoma redivivum*	0.0330
Mississippi Kite	*Ictinia mississippiensis*	0.0275
Scaled Quail	*Callipepla squamata*	0.0275
Site D 2012		
Species Common Name	Scientific name	Fraction
Northern Cardinal	*Cardinalis cardinalis*	0.1250
Ash-throated Flycatcher	*Myiarchus cinerascens*	0.1023
Cliff Swallow	*Petrochelidon pyrrhonota*	0.0909
Lark Sparrow	*Chondestes grammacus*	0.0795
Bullock's Oriole	*Icterus bullockii*	0.0795
Turkey Vulture	*Cathartes aura*	0.0682
Scissor-tailed Flycatcher	*Tyrannus forficatus*	0.0682
Brown-headed Cowbird	*Molothrus ater*	0.0682
Northern Bobwhite	*Colinus virginianus*	0.0682

Table 4.2. Site values for number of individuals, species richness and diversity index (H') across all study years

Year	Site	Number of Individuals	Species Richness	Diversity Index (H')
2008	A	329	43	3.280
	B	272	33	2.579
	C	182	33	3.141
2009	A	279	38	3.145
	B	258	35	2.972
	C	98	25	2.857
	D	183	31	3.156
2010	A	411	43	3.143
	B	244	37	3.155
	D	345	31	2.116
2012	A	312	32	2.916
	B	194	32	2.968
	C	134	27	3.015
	D	88	16	2.657
Mean		**237.78** **s.d. = 95.84**	**32.57** **s.d. = 7.02**	**2.936** **s.d. = 0.308**

4.2. Mixed Model Analysis of Covariance

4.2.1. Bird Abundance on Sub-sites

For response 1, the numbers of individuals averaged over all years, analyses indicated a significant interaction between oil and percent cover, and therefore the slope for percent cover differed between sites with and without oil development. This analysis indicated an effect of oil (df =1, F = 13.5901, p =.0000838) and oil pump jack by percent cover interaction (df =1, F= 4.581, p = 0.040). One sub-site that included 70 cliff swallows on a bridge caused an outlier. Once this sub-site was removed from the data set, the only significant effect was presence of oil pump jacks (df = 1, F = 6.93623, p= 0.01306) indicative of a higher mean number of birds on sites with oil pump jacks.

Response 2, the difference in pre-drought minus post-drought numbers of birds on sub-sites, with the bridge sub-site removed indicated no significant factor, covariate or interaction effects ($p < 0.05$). Thus, it appeared that these factors had no significant effects on bird numbers on sub-sites between the pre- and post-drought periods.

Response 3, the difference in the number of birds on sub-sites between the 2009 pre-drought and 2012 post-drought years was completely balanced as all sub-sites at the four sites were sampled in both years. The analysis without the bridge sub-site indicated no significant effects of any for the factors and covariates. Nor did a t-test reveal any mean difference in the mean number of birds on sub-sites between the two years ($n = 39$, mean difference = - 0.949, s.d. =12.75467, $p = 0.645$), suggesting that that there was no real response in sub-site bird numbers to drought.

4.2.2. Species Richness on Sub-sites

Analysis of sub-site species richness averaged over all study years (Response 1) indicated that none of the tested factors or covariates (i.e., presence/absence of oil or cattle, percent woody vegetation cover, oil pump jacks by cattle, oil pump jacks by percent cover, cattle by percent cover and oil pump jacks by cattle by percent cover) significantly impacted this variable.

In contrast, sub-site species richness for all pre-drought year minus the post-drought year of 2012 (Response 2) excluding the bridge site was significantly impacted by the presence of oil pump jacks ($n = 39$, $F = 6.930535$, $p=0.013093$) and by the oil by percent cover interaction ($n = 39$, $F = 5.12826$, $p = 0.03068$). These results suggested that well-vegetated sub-sites without oil pump jacks were more likely to show a decrease in species richness. Sites with oil pump jacks had a weaker or opposite relationship with percent cover. On average, sub-sites with oil pump jacks appeared to experience a greater decline in species richness during the drought than those without oil pump jacks. However, highly vegetated, undeveloped sub-sites also appeared to have a tendency to experience a decline in species richness during the

drought. For sub-sites with oil pump jacks, the post-drought decrease in richness was significantly different from zero based on overlap with the 95% confidence interval. Similarly, the change in species richness for the sub-sites without oil pump jacks was not significantly different from zero based on overlap with the 95% confidence interval. There is a caveat that highly vegetated or unvegetated sites experienced greater changes in species richness.

When analyzed without the bridge site, sub-site species richness data for the pre-drought 2009 minus post-drought 2012 (Response 3) was shown to be significantly impacted by the presence of oil pump jacks ($n = 39$, $F = 5.02305$, $p = 0.03231$). The analysis indicated that sub-sites with oil pump jacks displayed a larger decrease in post-drought species richness relative to sites without pump jacks. However, confidence intervals for these decreases at sites with and without oil pump jacks present included zero, suggesting that although these two groups of sites may have responded to droughts in different ways, the responses were relatively weak in both cases.

4.2.3. Species Diversity on Sub-sites

Sub-site species diversity (H') for data averaged over all years for all subsites (Response 1) proved not to be significantly impacted ($p < 0.194$ or lower) by any of the tested factors of presence/absence of oil or cattle and covariant of percent woody vegetation cover and their interaction terms using the data set with the bridge sub-site deleted. This result suggested there were no differences in species diversity between sites across years regardless of the degree of anthropogenic disturbance.

For response 2, sub-site species diversity pre-drought years 2008, 2009 and 2010 minus that recorded during the post-drought year of 2012, scatterplot results and interpretations were similar to those of species richness (see above). With the bridge site excluded, there was a main effect of oil pump jacks ($n = 39$, $F = 7.7386$, $p = 0.00912$) an interaction of oil pump jacks and cattle ($n = 39$, $F = 6.80112$, $p = 0.01389$). There was a suggestion of a somewhat complex pattern of covariate interaction. Thus, for sub-sites with oil pump jacks, those that were less

vegetated sites had a larger decrease in diversity. Conversely, those with more vegetation possibly showed an increase in diversity, while sites without oil pump jacks appeared to display the opposite pattern. After correcting for vegetation cover, there was only a weak difference between sub-site types in their average species diversity response.

Analysis of data (with the bridge sub-site excluded) for sub-site species diversity during the pre-drought year 2009 minus the post-drought year 2012 (Response 3) indicated that the presence of oil pump jacks as the only significant factor (n = 39, F = 5.51048, p = 0.02547). Mean difference in oil pump jack sub-site diversity = -0.19988, se ±0.8480 while the mean difference for non-oil pump jack sub-sites = 0.0546, se ±0.0.4695. There was a tendency for variance of residuals to increase with predicted level. Although the covariate relationships were not significant, the pattern was very similar to Response 2. Similar to species richness, there was a significant difference in the way oil pump jack sub-sites and sub-sites without oil pump jacks responded. Non-oil pump jack sub-sites tended to have a post-drought increase in diversity while the oil pump jack sub-sites displayed a post drought decrease in diversity (see above). These changes between the pre- and post-drought years were weak and zero was included in the 95% confidence intervals for these means.

4.3. *Vegetation Analysis*

4.3.1. Description of Vegetation on Study Sites

The most widely distributed woody vegetation throughout Site A was honey mesquite (*Prosopis glandulosa*). Other woody species included lotebush (*Ziziphus obtusifolia*), catclaw acacia (*Acacia greggii*), agarito (*Berberis trifoliolata*), western soapberry (*Sapindus drummondii*), little leaf sumac (*Rhus microphylla*), prickly pear cactus (*Opuntia engelmannii),* allthorn (*Koeberlinia spinosa)* and netleaf hackberry *(Celtis laevigata).* The average woody vegetation percent cover for the 10 sub-sites was 30.56%. Sub-site A1 had only 5.65 percent cover, and sub-site A4 had the highest percent cover, 46.43%. Percent cover for total

vegetation at sub-sites at Site A (Fig. 3.1A) was 84.31% in 2008 and declined to 69.18% in 2012.

Woody vegetation on the western side of Site B was predominately redberry juniper (*Juniperus pinchotii* Sudw.). Mesquite trees were found throughout the site. Salt cedars (*Tamarix ramosissima*) were found on the perimeter of all observed water bodies on this site which included two ponds and a shallow creek flowing between them (Fig. 3.1B). The average woody vegetation percent cover for the 10 sub-sites was 21.71%. Sub-site B3 had only 3.76 percent cover, and subsite B5 had the highest woody vegetation percent cover, 47.06 %. Total percent vegetation (from 2008 image analysis) estimated at sub-sites was 90.47%. Analysis made from 2012 sub-sites indicated a post-drought overall reduction in percent vegetation cover to 72.64%.

The eastern side of Site C was bordered by the Double Mountain North Fork of the Brazos River. Vegetation along the river was composed primarily of redberry juniper (*Juniperus pinchotii* Sudw.) and salt cedar (*Tamarix ramosissima*). Mesquite trees were found throughout the site. Juniper densities were greatest along the site's northern and eastern borders, but this species occurred throughout the site. Prickly pear cactus (*Opuntia engelmannii*) was very common in six of the 10 sub-sites including sub-sites 4, 5,6,7,8 and 9 (Fig. 3.1C). Analysis of indicated an average of 88.24% total vegetation coverage for all sub-sites, of which 16.82% on average was woody vegetation. General land use at this site was cattle grazing, which was well suited to the lower proportion of woody species. Analysis of post-drought 2012 percent vegetation cover including grasses indicated that it had declined to 74.47% on the 10 sub-sites. Sub-sites C1 and C3 showed the greatest decline in overall percent vegetation coverage from 2008 to 2012. Sub-site C1 received supplemental watering around the house, which was occupied until 2010. Sub-site C3 was located nearest to the banks of the river, which during the 2012 research period was a sandy bed with very little water flow.

Mesquite trees were found throughout Site D. Juniper densities were greatest along the site's northern and eastern on sub-sites 9D and 10D and scattered in the northwestern sub-sites 7D and 8D. Prickly pear cactus was very common throughout the site (Fig. 3.1D). This wildlife mitigation area site had the greatest percent vegetation cover at 96.48% in 2008. During the time that the Google Earth images were created, several factors explain the high vegetation coverage. These include higher precipitation which resulted in higher vegetation productivity and reduced area coverage by dirt roads.

4.3.2. Pre- and Post-Drought Changes in Sub-site Percent Vegetation Cover

The results of a Kolmogorov-Smirnov (K-S) test performed to determine differences in percent total sub-site vegetative cover for the pre-2011 drought year of 2008 versus the post-2011 drought year of 2012 across all sub-sites for both years yielded a DN statistic of 0.65 with a p value of 0.91507×10^{-8}. This result indicated a statistical difference in overall percent vegetation cover between years. The mean sub-site percent vegetation cover including grasses for 2008 was 89.9742% (n = 40, s.d. = 7.5978) which was reduced to a mean of 72.201% (n = 40, s.d. = 12.266) in 2012 after the drought year of 2011. In spite of major variation in sub-site mean percent vegetation cover among sites and across years, when combining all data, Least Squares Linear Regression Analysis revealed no significant correlation between sub-site total vegetation percent cover and bird abundance ($p = 0.8088$), species richness ($p = 0.1616$) or species diversity ($p = 0.1400$).

4.4. Comparison of Bird Abundance, Species Richness and Diversity among Sites in 2009 and 2012

The multivariate ANOVA analyses performed to determine the impacts of the main effects of site (i.e., Sites A, B, C and D) and year (i.e., pre-2011 drought year 2009 versus post-2011 drought year 2012) on sub-site bird abundance, species richness and species diversity index (H') revealed significant differences for mean sub-site bird abundance ($p > 0.00001$) and richness ($p = 0.0145$) among sites (Figs. 4.1 A, B and C). A *post-hoc* Fisher's Least Significant

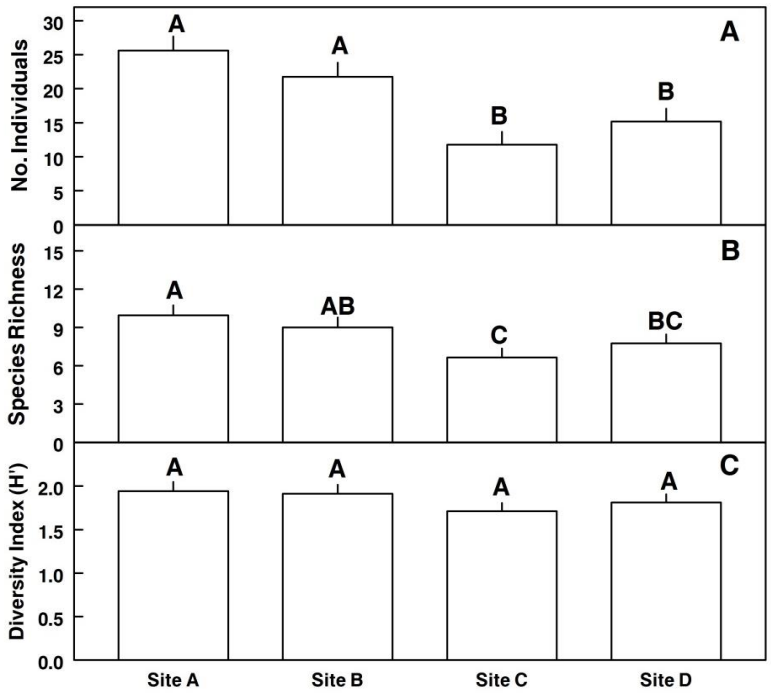

Figure 4.1 Comparisons across sites, by sub-site means for (A) number of individuals, (B) species richness and (C) diversity index (H'). These analyses were made using *post hoc* Fisher's Least Significant Difference (LSD) Procedure. Sites with means that were not signicantly different (p <0.5) are indicated similar letters above histograms.

Difference (LSD) procedure with significance set at p <0.05 indicated that mean sub-site bird abundance was not different between Sites A (oil pump jacks and cattle grazing) and B (oil pump jacks only). Similarly, mean sub-site bird abundance at Sites C (cattle grazing only) and D (undisturbed site) were not different, however, mean sub-site bird abundance at Sites A and B were significantly greater (p <0.05) than those at Sites C and D (Fig.4.1 A). Differences between sub-site mean species richness were more complex. Mean sub-site richness values were not significantly different between Sites A and B, Sites B and D, or Sites C and D (Fig. 4. 1B). However, mean sub-site species richness at Site A was significantly greater (p <0.05) than those of Sites C and D and that of Site C, significantly less (p <0.05) than recorded at Sites A and B. (Fig. 4.1B). In contrast, no differences (p >0.05) occurred in sub-site mean diversity

index (H') across all four sites (Fig. 4.1C). Similarly a *post-hoc* Fisher's Least Significant Difference (LSD) procedure with significance set at $p < 0.05$ indicated that there were no differences in mean sub-site bird abundance, species richness, or species diversity index (H') occurred between 2009 and 2012, pre- and post-drought years respectively (mean sub-site bird abundance: 2009 = 19.899 (se = 1.3914), 2012 = 17.288 (se = 1.3914); mean sub-site species richness: 2009 = 8.432 (se = 0.5226), 2012 = 8.2372 (se = 0.5226); mean sub-site species diversity (H'): 2009 = 1.7992 (se = 0.0708), 2012 = 1.8867 (se =m0.0708).

4.5. Analysis of Breeding Bird Survey Data

4.5.1. Rank Abundance of BBS species over years

Mean species richness across the 42-year span (1969 - 2011) of the Post, Texas BBS was 40.95 (n = 37, s.d. = 9.1559, range = 30 - 60), however, the total species richness for the 37 years included in the study was 87 species as a result of periodic counts of rare species. Qualitative analysis of ranks for all years of the survey indicated that the proportions of the BBS bird species that made up the top 75% of bird numbers included only seven species across all 37 years for which data were available (Table 4.3). The mean for the proportion for the top seven species in the count was 0.1083 (n = 7, s.d. = 0.17845, range = 0.0270 – 0.5090). The most numerous species in the total count was the cliff swallow at 0.509 of the total counted birds.

Table 4.3 Fraction of total bird numbers of species in the total 75% of birds from the Post Route #83092 Bird Breeding Survey (BBS) counts between 1969 and 2011.

Species Common Name	**Scientific name**	**Fraction**
Cliff Swallow	*Petrochelidon pyrrhonota*	0.5090
House Sparrow	*Passer domesticus*	0.0981
Northern Mockingbird	*Mimus polyglottos*	0.0330
Mourning Dove	*Zenaida macroura*	0.0313
Western Kingbird	*Tyrannus verticalis*	0.0311
Scissor-tailed Flycatcher	*Tyrannus forficatus*	0.0287
Northern Bobwhite	*Colinus virginianus*	0.0270

4.5.2. Comparison of Bird Numbers, Species Richness and Diversity across years 1969 to 2011

There was a high level of year-to-year variation in the number of individuals counted, species richness and diversity index (H') as computed from data generated by the annual breeding bird survey (BBS) on the Post, #83092, Texas, route (Fig. 3.3) from 1969-2011 (Figs. 4.2 C, D and E). Some of the variation may have been due to year-to-year differences in the individual performing the counts (i.e., seven different observers from 1969-2011) and climatic conditions occurring during any one specific annual count. This high level of inter-annual variation was reflected in high standard deviations for the overall mean of individual bird counts on the Post Route BBS where the mean equaled 1,026.3 yr^{-1} (n = 37) and the standard

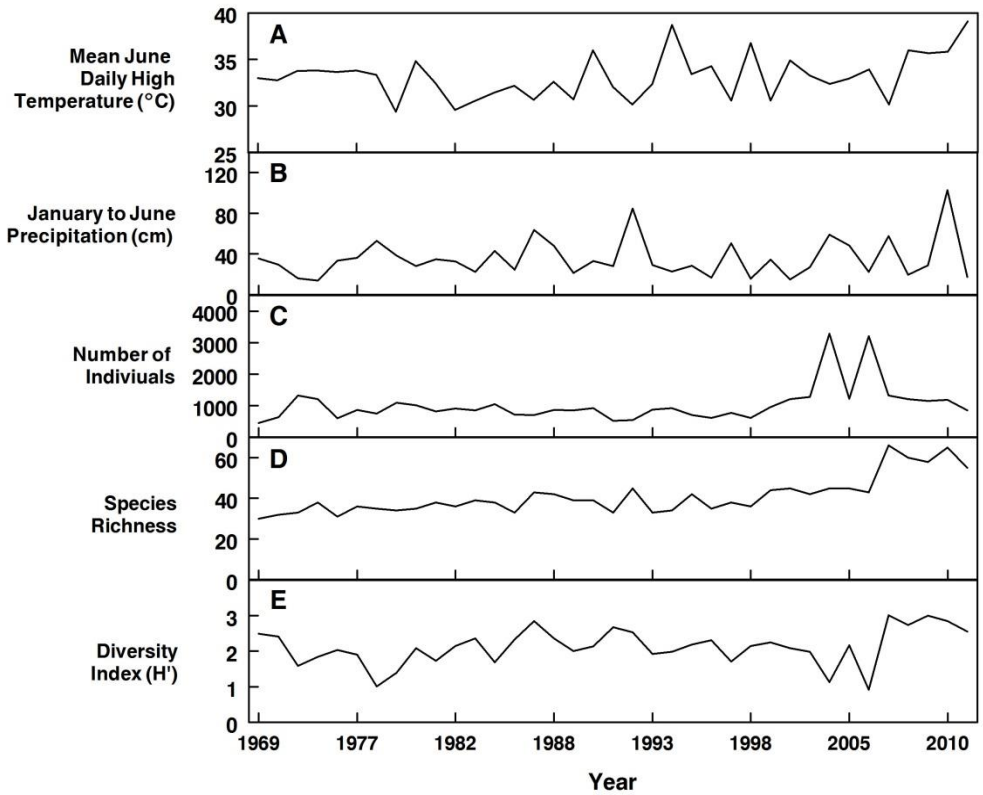

Figure 4.2. Breeding Bird Survey (BBS) data and climate data for the Post, #83092 Texas Route from 1969 to 2011 including (A) mean June daily high temperature (°C), (B) January to June total precipitation (cm) and the BBS parameters, (C) number of individuals, (D) species richness and (E) diversity index (H').

deviation and range, was 588.7 and 477 - 3,279 yr^{-1}, respectively (Fig. 4.2 C). In contrast, the values for overall mean annual species richness and diversity index (H') were far less variable than that for annual bird counts. Overall mean annual species richness was 40.9, s.d. = 1.51, range = 30 - 66 and mean annual species diversity, 2.12, s.d. = 0.509, range = 0.92-3.01 (Figs. 4.2 D and E).

4.5.3. Climate Effects on Bird Abundance, Species Richness and Diversity across years 1969 to 2011

Both annual mean June daily high ambient temperature and January-June total precipitation for the City of Post adjacent to the study sites were relatively variable over the BBS period from 1969 - 2011 (Figs. 4.2 A and B). The annual mean June daily high temperature for Post was 33.19°C (s.d. = 37°C, range 29.39 - 39.11°C) (Fig. 4.2 A) and the mean annual January-June total precipitation, 35.49 cm (s.d. = 19.37 cm, range = 13.64 - 102.77 cm) (Fig. 4.2 B). As indicated by the extensive range of annual mean June daily high temperatures and January-June total precipitation values, the study sites were subjected to major periods of drought and elevated temperatures with the maximum June daily high temperature and minimum January - June precipitation values of 39.11°C and 13.64 cm, respectively, occurring in the exceptional drought year of 2011 encompassed by the 2008 - 2012 study period.

With a single exception, Least Squares Linear Regression Analysis revealed no correlation between annual June average daily high temperature (i.e., during the breeding period) and total January-June precipitation (i.e., precipitation prior to breeding) with the measured bird community parameters (i.e., number of individuals counted, species richness and diversity index (H') on the Post, 383092 Texas, BBS Route for either bird count data collected within a year ($n = 38$, $p > 0.252$ or greater) or in the year subsequent to the climate data year ($n = 37$, $p > 0.184$ or greater). The single exception was species richness, which Least Squares Linear Regression Analysis indicated to be significantly ($p = 0.0164$) positively

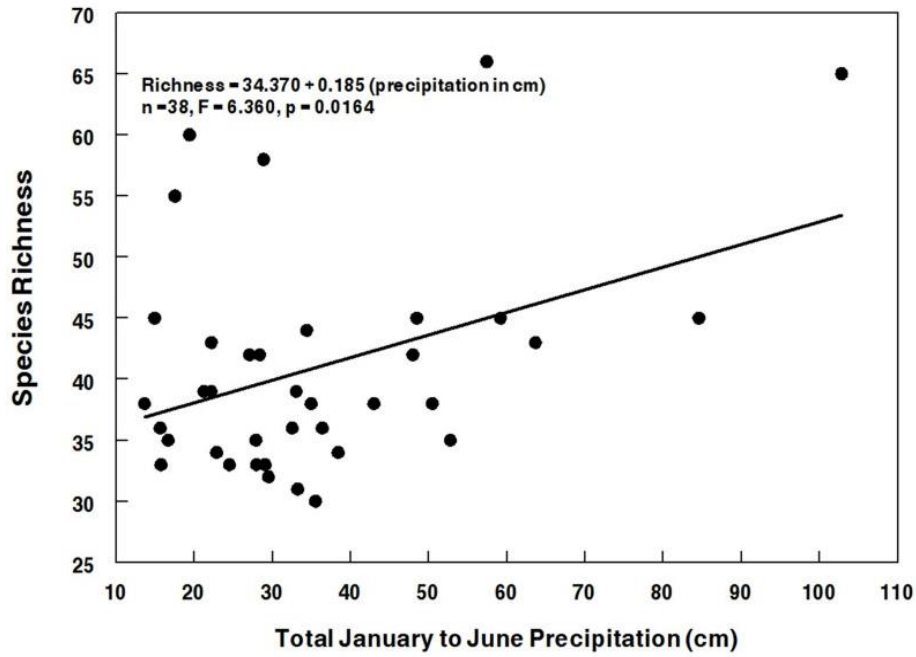

Figure 4.3. The relationship between species richness computed from the Post Route #83092 Breeding Bird Survey data (vertical axis) and January to June total precipitation in Post, Texas from 1969 to 2011 (horizontal axis). Regression analysis indicated a significant positive effect of precipitation on species richness.

correlated with mean January-June precipitation (Figure 4.3) (regression equation: Species richness = 34.70 + 0.185 (cm precipitation), n = 38, F = 6.63, r^2 = 15.373, p = 0.0164). Otherwise lack of correlation between June average daily high temperature and January through June precipitation with the tested parameters may have been due to their high level of year-to-year variability as revealed in Figure 4.2 A and B. It also suggests that bird communities are relatively resistant to the impacts of elevated temperatures during the breeding season and level of precipitation in months prior to the breeding season.

CHAPTER 5

DISCUSSION

5.1. Avian Community Dynamics

Mean data for species richness, and species diversity (H') indicated that the bird communities across the four study sites were relatively diverse (overall mean species richness = 32.57, range = 16 - 43, overall mean species diversity (H') = 2.936, range = 2.116 - 3.280) (Table 4.2). These data were similar to those derived from BBS data recorded during 1969 - 2011 in which mean annual species richness was somewhat greater than recorded in my study at 40.9 (s.d. = 1.51, range = 30 - 66) and mean annual species diversity considerably lower than recorded in my study at 2.12 (s.d. = 0.509, range = 0.92 - 3.01) (see Section 4.1.). Of specific interest was the considerably higher overall mean diversity index (H') derived from the study sites (i.e., 2.936) compared to that of the Post BBS Route survey (i.e., 2.12) suggesting that the bird communities on the study sites were more diverse than that of the broader geographic area in which they were located (Fig. 3.3). A potential factor explaining the apparent lower diversity recorded on the Post BBS route may be that bird counts were dominated by cliff swallows which appeared in great numbers (making up 50.9% of all counted birds, Table 4.3) due to the route crossing bridges used by this species as nesting sites.

Species richness can vary with size of the area over which bird counts are undertaken. Avian community studies can range from those focused on a limited number of species and study sites to studies involving bird communities encompassing a large number of species over an extensive geographical area. The number of species counted will increase with the size of the geographic area investigated, number of individuals counted and the duration of the

counting period (i.e., hours, days, weeks, or years). Thus, in a relatively limited study, Lawson et al. (2011) identified 25 bird species on 13 grassland sites on Padre Island National Sea Shore, Texas. Wilcox et al., (2010) studying the impacts of grazing on bird communities in 16 Florida grassland pastures recorded species richness values of 69 bird species at a monoculture site and 78 bird species at a mixed pasture site. Isacch et al. (2011) reported 28 bird species in short grassland sites and 33 bird species in tall grassland sites in Argentina. My research involving bird counts at 10 sub-sites each at four grassland sites across four different years yielded a total of 57 bird species which fell in the upper portion of that reported in similar studies discussed above (Wilcox et al., 2010, Isacch et al., 2011, Lawson et al., 2011) where species richness in the studied grassland ranged from 25 - 78. This result suggested that species richness in the studied grasslands of the Rolling Plains Ecoregion of Texas was similar to that recorded for grasslands in other geographic areas.

In contrast to these relatively limited studies, BBS-based research can use large data sets with bird species grouped by guilds including synathropes (30 species), permanent residents (82 species), short distance migrants (88 species) and Neotropical migrants (236 species), with a much higher total of 406 bird species counted (Albright et al., 2010). The bird species identified on my sites could also be divided into the migration guilds of permanent residents (19 species), short distance migrants (16 species) and Neotropical migrants (22 species), or into breeding habitat guilds including grassland (10 species), successional-scrub (15 species), wetland-open water (8 species) and urban (13 species). Such classifications of bird species into guilds have been used with BBS data to create a community specialization index (CSI) in order to examine changes in bird species indices across habitat types (Davey et al., 2012).

5.2. Impacts of Vegetation Cover

Even though vegetation cover varied between study sites and was greatly different (K-S test DN = 0.65, p <0.0000001) between the pre-2011 drought year of 2008 (mean 2008 percent cover = 89.87%, s.d. = 7.597, n = 40) and post-2011 drought year of 2012 (mean 2012 percent cover = 72.20%, s.d. = 12.266, n = 40 (Section 4.3.1), Least Squares Linear Regression Analysis revealed no significant correlation between sub-site percent cover (including grasses) combined for the years 2009 and 2012 when bird count data for all four study sites was available and number of individuals (p = 0.8088), species richness (p = 0.1616) or species diversity (p = 0.1400). Further, a mixed model ANCOVA for the three tested variables averaged over all sites (with the confounding bridge sub-site data deleted) and years indicated no significant direct impact of percent cover of woody shrubs on numbers of individuals (p = 0.640), species richness (p = 0.902) or diversity index (H') (p = 0.981) corroborating the lack of correlation between the bird species survey variables and degree of sub-site vegetation cover.

In contrast to my results, other research examining the relationship of a different proportion of vegetative cover type (i.e., woody, grass, bare ground and litter) found mixed, yet overall positive correlation for bird species numbers with vegetation cover (Ranellucci et al., 2012). However, Grant et al. (2004) has reported a significant decline in bird species with increased cover of grasslands by woody vegetation. Similar to my results, Wolf et al. (2012) found that differences in percent grass cover between two sites did not significantly impact the abundance of grassland breeding birds. Thus, the impact of the degree of either woody shrub cover or total vegetation cover on the abundance, richness and diversity of grassland birds appears to be somewhat of a mixed bag with the degree of coverage either increasing species richness or, as occurred in my study, having no effect on species richness.

5.3. Impacts of Anthropogenic Disturbance

5.3.1. Oil Pump Jacks

There was evidence of a significant impact of anthropogenic disturbance on bird communities within the studied sites, primarily associated with the presence of oil pump jacks. Thus, the mixed model ANCOVA for the three tested variables averaged over all sites (with the confounding bridge sub-site data excluded) and years indicated a direct positive impact of oil pump jack presence on mean sub-site bird abundance ($p = 0.013$). However, it had no impact on species richness ($p = 0.621$) or diversity index (H') ($p = 0.601$) suggesting that the presence of oil pump jacks attracted birds to sub-sites, but not specific species and, thus, did not impact species diversity. Similarly, a *post hoc* Fisher's Least Significant Difference (LSD) Procedure for significance at $p < 0.05$ following a multiple factor ANOVA examining the impacts of site (Sites A, B, C and D) and year (2009 versus 2012) on the three tested variables indicated that bird abundance was significantly elevated on sites with oil pump jacks present (Sites A and B) as opposed to sites without oil pump jacks (Sites C and D) (Fig. 4.1A) and that presence of oil pump jacks (Sites A and B) were associated with increased species richness relative to Site C without pump jacks with species richness at the natural Site D not being different from that of Site B (Fig. 4.1B). Like the outcome of the mixed model ANCOVA described above, this analysis appeared to also indicate that presence of oil pump jacks had no significant impact on species diversity (H') which was insignificantly different across all four study sites (Fig. 4.1C). Thus, these results again suggested that the main impact of the presence of oil pump jacks was to attract greater numbers of birds to sub-sites but had little impact on the species attracted. As such, my results did not support the Intermediate Disturbance Hypothesis which suggests that the highest levels of diversity are maintained when habitat disturbances are of intermediate scale (Connell, 1978) as occurred at sites A and B with oil pump jacks. Instead, bird diversity at these sites was insignificantly different from the cattle-grazed Site C and natural grassland Site D with the main effect of the presence of oil pump jacks being to increase bird abundance on

Sites A and B (Fig. 4.1A). Similarly Bayne et al. (2008) in a study involving sampling 5,129 individuals in 56 species of passerine birds found that passerine bird abundance was maximal at sites near well pads and declined with increasing distance from them.

Comparable to my study, research examining birds and distance from active oil well sites recorded a slightly higher number of birds at active wells versus inactive wells and roads potentially as a result of more favorable vegetation in disturbed active well sites (Lawson et al., 2011). Oil field sites had the intermittent sound of pump jacks operating. In research examining noise level disturbance, expected natural sound levels were recorded between 32 and 53 dB for all sites; however, active drilling produced 80 dB while running pumps, 48 dB (Lawson et al., 2011). The number of times that pump jacks were running during observations at my sub-sites was less than three per 16 of the oil wells per year, thus sound measurements were not included in my research. Bayne et al., (2008) reported results similar to those recorded in my study. They also found that the density of passerine birds significantly increased in the vicinity of active oil pump jacks but was significantly lower near continually high noise producing compressor stations compared to all other tested areas. This result lead to them to conclude that the high noise levels associated with compressors inhibited bird presence (Bayne et al., 2008).

Studies of the impact of human activities in avian habitats in both urban and rural settings allow comparisons of such impacts relative to human population density and associated levels anthropogenic environmental disturbance. A well-used method, flight initiation distance (FID) has allowed researchers to quantify avian habituation to anthropogenic disturbance. FID refers to the distance researchers are able to approach birds prior to inducing flight (Evans et al., 2010; Atwell et al., 2012; Jimenez et al., 2013). Clucas and Marzluf (2012) examined the avian responses to humans through a combination of surveys of human reaction toward birds and field data to quantify avian reactions to human presence among six bird species in urban and rural locations in Seattle, Washington and Berlin, Germany. Their results indicated that the

birds tolerated greater proximity to humans in the urban settings (i.e., FID = 9.6 m) than in rural settings (i.e., FID = 12 to 28 m) (Clucas and Marzluf, 2012). Clucas and Marzluf (2012) also showed a positive correlation between flight initiation distance (FID) and the level of positive attitudes toward the birds. The Berlin human survey participants had a greater concern for conservation than did the Seattle participants. (Clucas and Marzluf, 2012).

Evans et al. (2010) also found that song sparrows in the urban habitats permitted a significantly closer human proximity than in rural areas. Evans et al. (2010) reported that the FID range for song sparrows in urban versus rural areas was approximately 7 m and 20 – 22 m, respectively. They also noted that urban males had increased levels of “bold and aggressive” behavior toward humans. Avian habituation to human presence in urban areas has also been noted by Atwell et al. (2012) and Jimenez et al. (2013).

Of the 57 species included in my research, only nine of them were classified as urban in their breeding habitat. These included the common grackle, rock pigeon, Eurasian collared dove, white-winged dove, Inca dove, house finch, northern mockingbird, European starling and house sparrow (Gough et al., 1998). Observation distance from birds in my research ranged between 20 and 50 m, consistent with use of binoculars to collect data as used in avian studies from other rural sites (Evans et al., 2010; Atwell et al., 2012; Jimenez et al., 2013).

A common thread at my sites was the landowners’ interest in the conservation of birds and other animals on their land. A positive effect, as mentioned by Clucas and Marzluf (2102) was noted in the Sub-sites A1 and C1 where supplemental feeding was provided for birds by the people living in the houses. Other factors included influences of human presence on foraging, breeding and nesting. One of the shared aspects at the four research sites was low presence of humans. Although each site did have humans present, generally they were present on the land at regular, but limited daily intervals.

It is possible that habituation to humans contributed to the response of birds at the pump jack sites. Observations of the pump jack reader at Site A throughout the study showed

no aggressive or negative actions directed at birds. Contact with humans at the pump jack sites was generally limited to brief early morning and late afternoon visits to each pump by the pump reader. The pump reader drove to each well pad in a vehicle, took a reading and then drove to the next pump. Nests of curved- bill thrashers were active on pump jacks and in the storage tank walk-way at Site A, each year of the study.

Cattle-grazing sites (Sites A and C) also had a few people present to manage and feed the herds. Better cattlemen check their herds daily, while some only checked their cattle once a week or less, according to the landowner, Wyvonne Kennedy (personal communication). Human interaction at the natural Site D was limited to the site manager, and occasional groups given access via permit, as was my research team and hunters. Though this human disturbance was not quantified, seasonal hunting especially dove hunting, may have had periodic negative effects on birds. However, dove hunting which occurs from September through December in Texas did not take place during my study periods.

Another type of anthropogenic disturbance that was not part of this study, yet poses risks for avian communities is wind turbines and the expanding energy business that creates wind farms across the country and particularly in portions of Texas. Thirty-nine states and Puerto Rico currently have wind turbine energy generating facilities; Texas leads the nation with the largest number of wind turbines and wind farms (American Wind Energy Association, 2013). The two largest of the 54 US farms are also in Texas; the Roscoe Wind Farm, Roscoe, TX, with 627 wind turbines distributed on 100,000 acres, and the Horse Hollow Wind Energy Center, in Taylor and Nolan Counties, TX, with 423 wind turbines on 47,000 acres (Windpower Engineering and Development, 2013).

Mortality of birds and bats colliding with turbine blades has raised ecological concerns. Initially studies of the negative direct impact of wind turbines on wildlife focused on birds; it took over ten years for subsequent studies on bats to be initiated (Grodsky et al., 2012). Three factors that influence bird collisions with wind turbines are bird density, landscape and weather

(Thomas et al., 2011). Density considerations relate to migration versus local residents. In addition, disturbance caused by land clearing during construction of the turbines, roads and other site facilities may produce favorable insect foraging attracting birds and bats to wind turbine farms where they are subject to collisions (Horn et al., 2008). Disturbance studies tend to commonly involve presence/absence data on animals interacting with the landscape. In the case of birds and bats interacting with wind turbines, studies are based on collection and counting of bird or bat carcasses within a determined range of the base of each wind turbine (Korner-Nievergelt, et al., 2011). Scavenger removal of carcasses makes accurate counts of fatalities difficult (Smallwood et al., 2010). Recovery and study of these carcasses takes on a forensic science aspect including mapping location of bodies, photographing bodies and detailed examination of the remains. Baerwald et al. (2008) performed field necropsies on bats, indicating thoracic and abdominal cavity hemorrhaging as well as others with external injuries.

Research conducted at the Oklahoma Wind Energy Center, Oklahoma, in 2004 and 2005 found higher mortality rates in bats (n =111) compared to that for birds (n = 11) (Piorkowski et al., 2010). It is important to consider that not all birds and bats struck by wind turbine rotor blades die on impact. Smallwood et al. (2010) observed three golden eagles with differing degrees of injury that were consistent with wind turbine blade impact injuries. The American Wind Energy Association began work with the US Fish and Wildlife Service (USFWS) in 2007 by creating the Wind Turbine Guidelines Federal Advisory Committee (Anderson, 2013).

5.3.2 Cattle Grazing

In contrast to the presence of oil pump jacks; there was little evidence of a significant impact of cattle grazing on bird communities within the studied sites. Thus, the mixed model ANCOVA for the three tested variables averaged over all sites (with the confounding bridge sub-site data excluded) and years indicated no significant impact of cattle grazing on mean sub-site bird abundance ($p = 0.9323$), species richness ($p = 0.493$) or diversity index (H') ($p = 0.466$)

suggesting that cattle grazing had little impact on bird community dynamics in the study sites. Similarly, a multiple factor ANOVA examining the impacts of site (Sites A, B, C and D) and year (2009 versus 2012) for the three tested variables indicated that bird abundance, species richness and species diversity on the cattle-grazed Site C were not different ($p > 0.05$) from that of the natural site D (Figs. 4.1A-C). Thus, like the outcome of the mixed model ANCOVA described above, this analysis appeared to also indicate that cattle grazing had no significant effect on any of three bird community parameters.

Similar to my results, Rahmig et al. (2008) found that cattle grazing had no significant effect on bird diversity or abundance. Similarly, Powell et al. (2006) found no significant effects of either burning or bison grazing on grassland bird species richness, diversity (H') and evenness (E). Only bird abundance was impacted being significantly increased in unburned idle fields relative to bison-grazed, burned or hayed fields among which bird abundances was not significantly different. In contrast to my results and those of Rahmig et al. (2008) and Powell et al. (2006), Wilcox et al. (2010) reported that, in pastures containing shrubs and trees, total bird species richness, regardless of breeding habitat selection or migratory status, was decreased in in pastures subjected to low or high levels of cattle grazing and that species numbers were greatest in pastures subjected to a low level of grazing (Wilcox et al., 2010). Thus, my and other studies suggest that grazing has little impact on grassland bird community species richness and diversity but that grazing can negatively impact bird abundance (Wilcox et al., 2010) even though this was not the case in my study.

Birds and cattle grazing have a dynamic relationship which is based on its effect on the vegetation that supplies food and habitat for birds and their prey. The use of grasslands for grazing domestic animals is part of the change in land usage that have contributed to the decline in native grasslands, negatively impacting biodiversity, habitat integrity and the ecosystem (Questad et al., 2011). Killdeer lay eggs directly on the ground, placing them at extreme risk of being crushed by grazing cattle. Other species in the research area that nest low

or close to the ground resulting in increased risk of nest and offspring loss included scaled quail, northern bobwhite, common nighthawk and wild turkey. Fourteen of the species that resided on the research sites are listed as ground and low-nesting, many of which were sparrows. Although the great roadrunner is listed as a tree nesting species, I encountered a nest of eggs laid in the base of a bush.

It has been noted that while many studies found significant interactions between cattle grazing and ecosystems, they found the grazing effects to vary among vegetation types with respect to species richness and abundance (Sassi et al., 2009). The work of Ranellucci et al. (2012) was consistent with previous studies, indicating a preference of grassland birds to heterogeneity of the season-long grazing. Research examining grazed versus non- grazed land reported an increase in populations of generalist species birds in sites subjected to periodic/rotating grazing compared to season-long (i.e., continual) grazing (Ranellucci et al., 2012). Site A had been owned by the same family since 1937. Cattle grazing had been in place on the land since the 1940's, but there were no written records of numbers of cattle grazed on it. In general, 75 head of cattle along with horses and mules grazed at this site. The owners stated that the range was as low of 25 head of cattle during severe drought to as high as 125 head for 15 years between 1990 and 2005. Site C changed ownership during the research period. The owner at the beginning of the study leased her land for cattle grazing. Cattle were present across sites A and C throughout the research period. Cattle at the sites in general were not interested the presence of the research team or disruptive during bird counts. Rotational grazing practices were observed at both sites. Supplemental feed was also provided for the herds. Burning was not used at either site as a ranch-land management technique. Controlled burns would have been especially risky at Site A, which included pump jack and oil production equipment.

5.4. Impacts of Climate

My results were mixed for the impact of drought on the tested variables of bird abundance, species richness and species diversity among the study sites. The multiple factor ANCOVA for response 2, the averaged data (with the confounding bridge sub-site data excluded) for pre-drought years 2008, 2009 and 2010 minus that for post-drought0 2012, indicated that there was significant ($p = 0.039$) decrease in mean sub-site bird abundance of 6.029 birds sub-site^{-1} in the post-drought year, 2012, but no significant change in species richness or diversity ($p = 0.188$). For response 3, in which the sub-site means of the three tested variables recorded at all four sites during the pre-drought year 2009 were compared for difference against those recorded in the post-drought year of 2012, neither mean sub-site bird numbers, species richness, or diversity was significantly different ($p < 0.05$) between the pre- and post-drought years. These results suggested that drought had little influence on the bird communities among the studied sites. Similarly, the multiple factor ANOVA examining the impacts of site (Sites A, B, C and D) and year (2009 versus 2012) for the three tested variables indicated that bird abundance ($p = 0.1877$), species richness ($p = 0.7926$) and species diversity ($p = 0.7700$) did not significantly vary between the pre-drought 2009 and post-drought 2012 years. Thus, these results indicate that, in spite of the fact that sub-site vegetation cover was reduced by 20% in the post-drought year of 2012 compared to that in 2008; the drought had no detectable effect on the bird communities at the studied sites. This result correlated with my results that Least Squares Linear Regression Analyses of 2009 and 2012 data combined indicated that neither bird abundance, species richness, nor species diversity (H') were significantly correlated with percent vegetation cover in spite of a major reduction in vegetation cover across sub-sited between 2008 and 2012.

Analysis of the BBS City of Post data over years 1969 to 2012 also revealed little evidence that bird communities were greatly impacted by either drought prior to the breeding season or elevated ambient air temperatures during the breeding season. Thus, Least Squares

Linear Regression Analyses indicated that annual bird abundance (p = 0.807), species richness (p = 0.252) and species diversity (H') (p = 0.560) were not significantly correlated with the mean June daily high temperature for the corresponding year. Similarly, there was no correlation between annual bird abundance (p = 0.578), species richness (p = 0.138) and species diversity (H') (p = 0.571) with mean June daily high temperature for the previous year. Least Squares Linear Regression Analyses also indicated that annual bird abundance (p = 0.759) and species diversity (H') (p = 0.333) were not significantly correlated with the January-June total precipitation for the corresponding year. In contrast, species richness proved to be positively correlated with January-June total precipitation for the corresponding year (p =0.016) (Fig. 4.3). Regression analyses also did not indicate a correlation between annual bird abundance (p = 0.987), species richness (p = 0.0.858) or species diversity (H') (p = 0.764) with January-June total precipitation during the previous year. On the whole, these results like those of the multiple factor ANCOVA and ANOVA described above appear with minor exceptions to indicate that drought has little or no impact on bird abundance, species richness, or species diversity in the Rolling Plains Ecoregion in which this study was conducted.

Previously published research yields conflicting results on the impacts of temperature and precipitation on bird community dynamics. Unlike my BBS results, Albright et al. (2010) using North American BBS data for 1989 - 2005 for dry areas of the central United States encompassing my study area found a positive relationship between precipitation over a prior 32-week period ending in June and both bird abundance and species richness. In contrast, when precipitation in the prior year was used as the independent variable no significant relationship was found with either dry area bird abundance or species richness opposite my research results suggesting a significant relationship between January to June total precipitation and species richness only when precipitation in the prior year was used as the independent variable (Fig. 4.3). Also, in contrast to my BBS results indicating no impacts of mean June daily high temperature on any of the three tested variables, Davey et al. (2012) using BBS data for

randomly selected 3,000 bird count areas throughout Britain described a significant positive correlation between mean breeding season temperature and both species richness and species diversity which was stronger than that of their correlation with precipitation. Taken as a whole, these results suggest that, over long periods, the relationships between bird abundance, species richness and species diversity may be considerably variable among different regions, continents and different climatic areas.

Possible temperature and precipitation effects in bird communities during breeding season could be driven by the habitat choices of migrating birds. Whereas permanent residents may adapt to climate changes, migrating birds may select different locations with appropriate environmental conditions (Albright, 2010). It is suggested that there is a link between a warming climate and community homogenization that will lead to increases in the range of generalist bird species at the expense of specialist bird species (Davey et al., 2012). Specialists are defined by their limitations to resources and habitat, being unable to tolerate broad environmental variation to a lesser degree than can generalist species (Barnagaud et al., 2011). Although bird abundance, species richness and species diversity were not measured in drought year of 2011 at my research sites due to the threat of fire, other researchers have suggested that, during an extreme drought year, avian communities experience reduced reproductive success and survival (Albright et al., 2010) that could have latent impacts on bird community dynamics in the years flowing the extreme drought event. Interestingly, my research showed no latent impacts on bird abundance, species richness and species diversity (H') in 2012 following the extreme drought of 2011 suggesting that such latent negative drought impacts did not affect grassland bird communities in the Rolling Plains Ecoregion. Research has also indicated that entire avian communities can withstand extreme heat waves and that further study is required to better understand the impacts of extreme heat and drought on avian populations and communities (Jiguet et al., 2011).

CHAPTER 6

CONCLUSIONS

The impacts land use on bird numbers, species richness, and species diversity for sites in the Rolling Plains Ecoregion of Texas suggested that oil well land use had higher bird abundance and species richness when compared to cattle grazed land and natural land. Land with cattle grazing had the lowest species richness, with mixed relationships where land used for oil wells was similar to oil/cattle and natural land. Species diversity was similar across all four land use types. Overall, drought did not negatively impact bird numbers, species richness, and species diversity on my study sites or in the BBS study. The factors of anthropogenic disturbance, oil pump jacks, cattle grazing, presence of humans or anthropogenic structures did not appear to adversely impact bird abundance, species richness or diversity (H'). Whereas drought resulted in a significant reduction in vegetation coverage, vegetation coverage was neither a positively or negatively correlated with bird abundance, species richness, nor species diversity in the Rolling Plains Ecoregion of Texas.

Overall, it was clear in this study that oil pump jacks attracted birds and increased species richness, over grazed and natural areas, while, in the area studied, grazing and drought appeared to have little impact on bird abundance, species richness or species diversity. These results suggested that, in rural grasslands, bird communities are relatively resistant to both limited anthropogenic disturbances and the impacts of high ambient temperatures during the breeding season and extreme drought as occurred on the study sites in 2011. In a general

sense, my results appear to agree with other research on the impacts of anthropogenic disturbance, grazing and temperature/drought on grassland avian communities which report limited or no impacts of anthropogenic or climate factors on grassland bird abundance, species richness or species diversity.

Unfortunately, the limited number of short- and long-term studies of the impacts of anthropogenic disturbance, grazing and climate on grassland bird abundance, species richness and/or diversity make drawing of an accurate general assessment of the impacts of these factors on grassland avian communities difficult if not impossible to determine. Clearly, further research is required in this area before any widely accepted, research-supported conclusions can be drawn regarding the impacts of anthropogenic disturbance, grazing, and temperature/drought on grassland bird community dynamics. With the predicted increase of ever more rapid climate change, including higher average temperatures and extended periods of extreme drought, the most important of these understudied areas in grassland bird communities to which to apply future research may be that of the impacts of increased ambient temperature and reduced precipitation.

The state of Texas experienced the worst one year drought period between August 2010 and July 2011 in 116 years of recorded weather (Texas Parks and Wild Life, 2011). Although the studied avian communities of the Rolling Plains Ecoregion displayed a resiliency to the extreme drought of 2011, climate data indicated that more than 80% of the state of Texas has remained in drought conditions, indicating the need for continued studies (U.S. Drought Monitor, 2013). Prior to final publication of this study, the North American BBS data was updated with the 2012 surveys. The Post, Texas BBS route, #83092, indicated an increase in bird abundance in 2012 from 847 in 2011 to 1,027 and species diversity (H') from 2.56 to 2.91. However, results for species richness increased only 3 species from 55 to 58 (Sauer et al., 2013).

Future research could involve use of additional factors not measured in this study in the analysis of the impacts of land use on the bird communities on the study sites that were not measured or included in this dissertation research. These include measurement of sub-site landscape heterogeneity for use as a covariant in mixed-model ANCOVA analysis of bird abundance, species richness and species diversity between sites. To a certain extent this was done by using percent vegetation cover as a covariant, however, in any future studies the degree of sub-site topological variation could be included as an informative covariant as well as sub-site distance from water and/or sub-site vegetation species diversity. Further, the data could be analyzed for differences in bird abundance, species richness and species diversity for different bird guilds (i.e., resident versus migrant bird species, bird feeding guilds and/or different bird taxonomical groups at the family level). It is also important to note that the fact that limited human disturbance can have minor (i.e., grazing) or positive impacts (oil pump jacks) on bird communities should not be overlooked in bird conservation efforts. A multitude of factors impact avian communities. A combination of continued research and application of new information on the impacts of land usage on grassland bird communities could improve the outlook for not only grassland birds, but also the communities and ecosystems in which they exist.

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BIOGRAPHICAL INFORMATION

Natalie is a transplanted Texan and a native Pennsylvanian. She moved Georgetown, Texas, with her family in 1972, and later settled in Arlington, Texas, in 1998 where she pursued higher education. She earned a Bachelor's Science degree in 2003 and a Master of Science degree in Biology in 2005 (Electron microscopic examination of the egg shell structure in ring-necked doves and chickens) both from The University of Texas at Arlington. Natalie has enjoyed teaching in many aspects throughout her life, and in 2006 was selected by the Department of Biology faculty and awards committee as the recipient of the T.E. Kennerly Award. This honor is given to the Graduate Teaching Assistant who "best exemplifies the devotion to teaching and the concern for students exhibited by the late Dr. Kennerly." After receiving her MS degree Natalie took up a position as Life Science Teacher at Nolan Catholic High School in Fort Worth, Texas, in 2007 where she presently continues to teach.

At a young age she was greatly influenced by her mother's passion for nature and ecology and by her father's interest in photography of the diversity of nature and humanity. Earth Day projects, camping and National Parks have always been a part of her life. As an elementary school student, when given the choice of people to write about, she selected Jane Goodall and her research. Natalie is published professional photographer, who enjoys using her talent and skill in all aspects of her career and life. In June of 2007, she spent time in the summer heat of Big Bend National Park with Dr. Joseph Kuban where they laid out the basic plan for her research. Their discussions of ecology and the pressure that humans place on animals and habitats led her to think the ecological impacts of human disturbance. They discussed comparing bird species diversity in landscapes within a grassland ecoregion with different levels anthropogenic disturbance, particularly the disruptions associated with areas that have active oil fields, with and without cattle grazing, and with cattle grazing alone relative to natural undisturbed sites. The following month she did an initial survey of birds and secured research sites in the Rolling Plains Ecoregion near the City of Post, Texas. The first site she

secured had been home to prairie dog towns and burrowing owls. That summer she observed and photographed the site's last burrowing owls to remain after the prairie dogs were long gone. The realization that the two habitat related species were no longer present on the site put a sense of urgency to begin her study which eventually lead to her dissertation research and the granting of a Ph.D. degree in Biology in 2013.

In a speaking engagement that Natalie attended in 2013, Jane Goodall reminded researchers of the importance of involving local people in your work as possible ask questions and to respect them. There is a great factor of trust when you have asked property owners to access their land over great periods of time that a research study requires. Natalie would not have been able to do her research without the aid and involvement of the people who were land owner and other people in the city who were of considerable help accessing land and selection of the appropriate sites for her research.

She plans to continue teaching and explore further aspects of her research interests.