IMPACTS OF WILDFIRE ON AVIAN COMMUNITIES OF SOUTH TEXAS

A Thesis

by

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ABSTRACT

Impacts of Wildfire on Avian Communities of South Texas

(December 2011)

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On March 14, 2008, an intense wildfire occurred on the Chaparral Wildlife Management Area, located in southern Texas. I sought to examine effects of the fire on avian species abundance and composition by comparing avian species richness, density and presence, and vegetation components on burned and unburned sites. I conducted transect and point-count surveys in winter and summer 2009-2010. Burned sites had higher species richness both winter seasons, but no treatment effect was observed for summer. I observed higher densities and probabilities of presence on burned sites for several granivorous, insectivorous, and/or ground-nesting birds, including several migratory grassland sparrows, during both winter seasons,. Probabilities of presence increased for most winter birds over time, whereas decreases were observed during summer. Most vegetation components showed substantial recovery 2 years post-fire. The effects of wildfire were generally positive for the avian community, and increased habitat for grassland-obligate birds without substantial impact on resident shrubland species.

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CHAPTER I

IMPACTS OF WILDFIRE ON WINTERING AVIAN COMMUNITIES OF SOUTH TEXAS

INTRODUCTION

Fire has played an important role in the maintenance of many ecosystems in North America, particularly grassland ecosystems (Wright & Bailey 1982, Hatch et al. 1991, Bragg 1995). Although researchers are confident that fire played a key role in maintaining grasslands, historical fire intervals are difficult to ascertain due to the absence of scarred tree rings or any long-lived woody vegetation (Higgins 1986). Based on woody plant invasions, fire interval estimates for the semi-arid grasslands of the west and southwest range from 7-10 years (McPherson 1995) to every 25 years (Wright & Bailey 1982).

Historically, warm-season wildfire maintained grasslands by removing most encroaching shrubs, stimulating seed production in grasses and forbs, and providing patches of bare ground by removing excess litter (Brooks 2008). Given that historical fire regimes helped create and maintain grassland ecosystems, the floral and avifaunal species of the region are well-adapted to fire (Reinking 2005). The continent-wide loss of grasslands through land-use change and fire suppression has negatively affected grassland-obligate birds, most of which appear on federal and state sensitive, threatened, or endangered species lists (Abrams 1992, Umbanhowar 1996, Sauer et al. 2001). Out of 19 widespread grassland species in North America, 14 are declining (Knopf 1994). Declines in these avian species have occurred in every grassland ecosystem on the continent, and are considered to be a significant conservation issue (Brennan & Kuvlesky 2005, Askins et al. 2007).

Loss and degradation of the semi-arid shrub-grasslands of the Rio Grande Plains in south Texas are of particular concern, as they provide wintering habitat for a number of migratory grassland species (Pulliam & Dunning 1987). The avian species that inhabit south Texas shrub-grasslands provide both valuable ecological services and economic benefits to the region. Granivorous and frugivorous species act as seed dispersers throughout their range, and passerines provide a food source for larger raptors, mammals and reptiles. Avian species in this ecoregion also help control insect populations, including many agricultural pests. The wintering avifaunal species also contribute significantly to the south Texas economy through birding and other ecotourism activities. Ecotourism is the fastest growing tourism market in Texas, with an annual growth of 10-30 percent (Vincent & Thompson 2002). One example of ecotourism is the Rio Grande Valley Birding Festival (RGVBF) which draws over 2,300 tourists to the region each November, with a total economic impact of approximately 1.5 million dollars (Vincent et al. 2003). Because avian communities are a valuable natural and socioeconomic resource in south Texas, it is important to understand how stochastic events, such as wildfire, impact both resident and migratory wintering bird habitat.

Degradation of this crucial habitat occurs through a decrease in herbaceous species richness and abundance, caused by an increase in the presence or abundance of invasive plant species, and more specifically, woody species. Even minor increases in woody plant height and density in grasslands can translate to decreases in avifaunal species richness (Grant et al. 2004). The detrimental effects of fire on the spread of woody plants has been well-documented, and that suppression of naturally occurring fire historically led to an increase in woody plant density in the shrub-grasslands of south Texas (Bogusch 1952, Hanselka 1980, Archer 1989). The increase in woody vegetation density and subsequent vegetation composition shift in this region has led to an increase in shrub-obligate birds in the avian species composition. The question arises as to how these grass- and shrub-obligate, migratory and resident south Texas birds respond to possible dramatic changes in the vegetation community caused by large ecological disturbances such as wildfire.

Few studies have examined the effects of warm-season wildfire on avian species in Texas. Reynolds and Krausman (1998) studied the effects of a prescribed winter burn on both winter and breeding season birds on the south Texas Plains. Grassland-obligate bird abundance was significantly greater on burned sites, while shrub-obligate species abundance increased on unburned sites. In another study on a south Texas barrier island that examined the effects of winter and summer prescribed burning on the avian community, all observed winter sparrow (*Emberizina*) species were most abundant on burned sites, whereas wren species (*Troglodytida*) were more abundant on control sites (Van't Hul et al. 1997). However, another study conducted on rangelands in the Texas Panhandle following the East Amarillo Complex (EAC) fires indicated that both post-fire wintering and breeding bird populations were similar to those before fire (Roberts 2009).

The objectives for this study were to examine how an intense, warm-season wildfire affected a south Texas shrub-grassland community, and quantify the effects of the fire on both migratory wintering grassland birds and resident avian species. I developed 3 predictions relevant to the research objectives:

I. Reductions in density and presence of shrub-foraging avian species will be observed on burned sites.

II. Increases in density and presence of ground-foraging avian species will be observed on burned sites.

III. Burned sites will have greater overall avian species richness than unburned sites.

METHODS

Study Area

Study areas included the 6,151-ha Chaparral Wildlife Management Area (28° 19' N, 99° 24' W), managed by Texas Parks and Wildlife Department, and the Piloncillo Ranch (28° 17' N, 99° 17' N), located in LaSalle and Dimmit counties in the western Rio Grande Plains of south Texas. I used a 2,460-ha pasture of the Piloncillo Ranch that was adjacent to the Chaparral Wildlife Management Area (WMA) as the unburned control for this study. Soil types were similar for both study areas and included Dilley fine sandy loam, Duval loamy fine sand, Duval very fine sandy loam, and Duval fine sandy loam (NRCS 2011). A possible confounding factor in the study was a six-month cattle lease in the pasture where all unburned sites were located. Grazing occurred between November 2009 – April 2010, and any avian surveys that were disrupted by cattle movements were abandoned and conducted later in the morning.

Study sites were representative of the Rio Grande Plains mixed-brush community (McLendon 1991). Dominant woody species included mesquite (*Proposis glandulosa*), granjeno (*Celtis pallida*), brasil (*Condalia hookeri*) and hogplum (*Colubrina texensis*).

Other woody species included cactus (*Opuntia* spp.), huisache (*Acacia minuta*), leatherstem (*Jatropha diofica*), and Texas persimmon (*Diospyros texana*) (Ruthven et al. 2000). Common herbaceous species included hooded windmill grass (*Chloris cucullata*), Lehman lovegrass (*Eragrostis lehmanniana*), grama grasses (*Bouteloua* spp.), partridge pea (*Chamaescrista fasciculata*), and plains lazy daisy (*Aphanostephus ramosissimus*).

Climate was characterized by temperatures ranging from an average daily minimum of 5°C in January to an average daily maximum of 37°C in July. Ten year (2000-2009) rainfall for this area averaged 58 cm (Chaparral WMA unpublished data). The precipitation pattern is bimodal, with most rain occurring in late spring and early fall (Stevens & Arriaga 1985); however, surveys for winter 2009 (January – February) were conducted during a 16-month drought that occurred in the South Texas region (Figure 1.1, pg. 32). Palmer Drought Severity Index (PDSI) values (Palmer 1965) ranged from -2 (moderate drought) to -4 (extreme drought) throughout most of 2009 (Chaparral WMA unpublished data). By November 2009, above-average rainfall alleviated drought conditions and winter 2010 surveys were conducted that January and February with PDSI values of 2 and 3, respectively.

Vegetation Surveys

Transect lines were established on both study areas to conduct both avian (Burnham et al. 1980) and vegetation surveys from January-February of 2009 and 2010. I established 10, 400-m transects on each study area for a total of 20 transects. Transect locations were selected randomly, but were required to consist of 30-50% brush cover and native grass dominance (less than 30% exotic grass species). Due to the high density of roads on the study areas, several transects crossed a maximum of one secondary dirt road at a perpendicular angle. Transects began at least 30 m from paved roads and power lines. Transects were not located near standing water or man-made infrastructure.

I quantified visual obstruction height (m) of vegetation by placing a Robel pole (Robel et al. 1970) at a random perpendicular distance (0-100 m) and direction (left/right) every 40 m along each transect. Visual obstruction height was estimated 15 m from the pole in each of the four cardinal directions for a total of 40 measurements per transect (400 per study area). I also estimated groundcover composition using a Daubenmire frame (Daubenmire 1959). Frames were placed at a random distance (0-15m) from each pole location in each of the four quadrants of the pole (ex. NE, SW) for a total of 40 frames per transect (400 per study area). I estimated percent abundance for litter, bare ground, grasses, and forbs in each frame.

Avian Line Transect Surveys

To minimize the possibility of repeated observations, transects were placed a minimum of 300 m apart. During the survey, I recorded all visually observed birds, although only observations within 100 m of transects were used for analysis (Appendix I, pg. 66). I obtained exact measurements of the perpendicular distance from each bird to the transect line using a laser rangefinder. A measured, consistent, and continuous pace was maintained during transect surveys. Surveys began 30 minutes after sunrise and continued for three hours. Surveys were not conducted on days with high winds (> 20 km/h), heavy fog or precipitation. To account for hourly morning variation in avian activity, I altered the order in which each line transect was surveyed. All surveys were repeated twice in 2009 and three times in 2010.

STATISTICAL ANALYSIS

I used Program DISTANCE 6.0 release 2 (Thomas et al. 2010) to compute model-averaged density estimates for bird species with > 40 total observations between 2009 and 2010. Observations for each species were truncated at varying maximum distances in order to remove outliers and binned into unique distance categories to achieve greater model fit. I assessed models with 3 possible key functions (Uniform, Half Normal, and Hazard-rate) with the appropriate expansions (Cosine, Simple Polynomial, or Hermite Polynomial). Models were chosen based on Akaike's information criterion values adjusted for small samples (AICc) (Littell et al. 2006). I generated estimates of avian species richness using Program SPECRICH (Burnham & Overton 1979). Probability of presence estimates were obtained for each avian species that had observations on 20-80% of total transects. Avian species richness, individual species densities, and vegetation community characteristics were natural log-transformed to meet assumptions of homogeneity of variances. Vegetation community characteristics included visual obstruction height, percent abundances of litter, bare ground, grass, forbs, and total cover (combined percentages of grass, forbs, and litter). I compared avian and vegetation estimates between study areas and years using a generalized linear mixed model with the appropriate distribution and link function. Explanatory variables used in the analysis included Time (since burn, Year 1 and Year 2), Treatment (Burned and Unburned), and an interactive Time x Treatment term. If the interactive term was not significant, I removed it from the model and ran the analysis again using only Treatment and Time. I assessed 4 covariance structures (compound symmetric, first-order autoregressive, first-order autoregressive moving average, and toeplitz) and chose the appropriate structure based on AICc values (Littell et al. 2006). I used $P = 0.10$ as the cutoff for significance in order to explore all possible trends and causal relationships in the data. I also compared heterogeneity (patchiness) in total cover and visual obstruction height within burned sites and unburned sites for 2009 – 2010 by calculating a coefficient of variation and 95% confidence intervals for each study area.

RESULTS

Vegetation Community

I observed significant differences between study areas for both treatment and time, as well as interactive effects of treatment x time in the vegetation community (Table 1.1, pg. 25). In general, burned sites had lower visual obstruction height, litter, and native grass (Table 1.2, pg. 26). Additionally, overall declines were observed over time for litter, native grass and visual obstruction height on both treatments (Table 1.2, pg. 26).

Significant Treatment x Time interactions were observed for total cover, bare ground, forbs and total grass cover (Table 1.1, pg. 25), which indicates that change in percent cover for these components differed over time between treatments. Total cover was lower on burned sites (68.3, 95% CI = 64.6 – 72.2) than unburned sites (81.1, 95% CI = 76.6 – 85.7) in Year 1 (Y1); however, while total cover on burned sites increased in Year 2 (Y2), total cover on unburned sites remained unchanged. Bare ground was higher for burned sites (33.1, 95% CI = 26.7 – 41.0) than unburned sites (19.4, 95% CI = 15.7 – 24.0) in Y1, and decreased by almost half on burned sites in Y2, while bare ground on unburned sites remained consistent between years (Table 1.2, pg. 26). Forb abundance in

Y1 was initially higher on burned sites compared to unburned sites; however, by Y2, percentages of forbs had increased five-fold for burned sites and thirteen-fold for unburned sites (Table 1.2, pg. 26), with forbs on both study areas comprising between 40-45% of total groundcover. Total grass cover (native and non-native) was similar on burned and unburned sites in Y1, and declined on both study areas from Y1 to Y2, although the decline was greater for unburned sites (Table 1.2, pg. 26).

Burned sites had slightly higher heterogeneity (patchiness) in visual obstruction height (vegetation height and density) between sites (coefficient of variation = 65.91, 95% CI = 57.71 – 74.11), compared to unburned sites (57.47, 95% CI = 47.88 – 67.06) in 2009; but by 2010, study areas were similar in heterogeneity (burned sites: 64.05, 95% CI = 50.31 – 77.80, compared to unburned sites: 63.35, 95% CI = 54.17 – 72.54). Heterogeneity in total cover (percent cover of grasses, forbs and litter) was substantially higher on burned sites (40.19, 95% CI = 33.99 – 46.40) than unburned sites (26.92, 95% CI = 21.74 – 32.09) in 2009; however, by 2010, burned sites were less heterogeneous than unburned sites (burned: 23.71, 95% CI = 18.81 – 28.60; unburned: 31.12, 95% CI = 22.94 – 39.30).

Avian Community

I collected a total of 243 observations (158 on burned sites, 85 on unburned sites) for winter 2009 and 835 observations (550 observations on burned sites, 285 on unburned sites) for winter 2010, for a total of 1,078 observations. I observed a total of 50 bird species in 2009 – 2010 seasons. In winter 2009, I observed 28 species in the burned sites and 26 species in the unburned sites, with 50% of the species observed on both study areas. In winter 2010, I observed 38 species on the burned sites and 28 species on the unburned sites, with a 47% species overlap on both study areas.

Species richness – Species richness was substantially higher for burned sites, with an average of 19 species per site compared to 14 for unburned sites. Overall species richness for both treatments was almost twice as high during Year 2 (22, 95% CI = 17.9 – 26.0) than Year 1 (12, 95% CI = 10.4 – 15.0).

Presence – I analyzed the differences in probability of presence (occurrence) on either study area for 16 species (Table 1.3, pg. 27). Pyrrhuloxia (*Cardinalis sinuatus*) were more likely to occur (probability of presence = 50.0%, 95% CI = 24.0 – 76.0) on burned sites compared to 14.3% on unburned sites (95% CI = 3.0 – 45.0). Cassin's Sparrow (*Aimophila cassinii*) had a 35% probability of presence on burned sites (95% CI = 19.0 – 56.0), compared to 10% for unburned (95% CI = 2.8 – 30.1). Bewick's Wrens were twice as likely to be found on unburned sites than burned sites and also had a significantly higher probability of presence during Year 2 (Table 1.4, pg. 29). Out of 16 species analyzed, eight species exhibited an increase in probability of presence over time on both study areas (Table 1.4, pg. 29).

Density – Five species qualified for density analysis for the winter season (Table 1.5, pg. 30) by having at >40 observations over 2009 – 2010 seasons. While treatment did not have a singular effect on any of the species, time since burn significantly and positively affected the density of both Black-throated Sparrow (*Amphispiza bilineata*) and Western Meadowlark (*Sturnella neglecta*). *A. bilineata* increased from 1.3 (95% CI = 1.12 – 1.39) to 1.6 (95% CI = 1.45-1.80) individuals per hectare from Year 1 (Y1) to Year 2 (Y2). *S.*

neglecta density increased from 1.0 (95% CI = 0.95 – 1.07) to 1.2 (95% CI = 1.11 – 1.26) individuals per hectare from Y1 to Y2.

Patterns of change in density over time differed between treatments for Northern Mockingbird (*Mimus polyglottos*), Vesper Sparrow (*Pooecetes gramineus*), and White-crowned Sparrow (*Zonotrichia leucophrys)*. *M. polyglottos* density remained constant at 1.3 individuals per hectare on burned sites over time, but increased in density on unburned sites from 1.3 (95% CI = 1.11 – 1.45) individuals per hectare in Y1 to 1.5 (95% CI = 1.28 – 1.68) individuals per hectare in Y2. *P. gramineus* density increased on burned sites over time, from 1.2 (95% CI = 1.02 – 1.32) individuals per hectare in Y1 to 1.4 (95% CI = 1.26-1.63) individuals per hectare in Y2, but density remained constant on unburned sites at roughly 1.0 individuals per hectare. While *Z. leucophrys* densities increased on both study areas from Y1 to Y2, the increase on the burned sites was much larger (Table 1.6, pg. 31).

DISCUSSION

Vegetation Community

The majority of significant differences in vegetation component percentages between study areas can be attributed to effects from the wildfire. Fire reduces litter, increases bare ground, and can reduce woody vegetation density, height, and visual obstruction in rangelands (Box et al. 1967, Wright & Bailey 1982, Ruthven et al. 2000, Heisler et al. 2004). The wildfire burned in varying intensities throughout the Chaparral WMA, creating a vegetation mosaic of varying height, density, and structure. However, it is notable that by two years post-fire, total percent cover on burned sites was similar to that of unburned sites, illustrating a post-fire recovery rate for groundcover similar to other Texas studies where prescribed fire was used (Van't Hul et al. 1997, Reynolds & Krausman 1998). Although total cover was similar between study areas in Y2, unburned sites were more heterogeneous than unburned sites, which may have been due to cattle grazing, which would increase the patchiness of the groundcover.

Differences in abundances of non-native grass may have been due to the management histories of the study areas. The Chaparral WMA has a substantial number of paved roads, and is open to public vehicles during portions of the year. Increased vehicular traffic, as well as vehicular access to all pastures inside the WMA, would perpetuate the spread of any exotic grass introduced to the property. Because it is a private property, access to the Piloncillo Ranch is more restricted, and therefore the proliferation of exotic grasses in the pasture would be less likely to occur.

In the two-month period after the wildfire occurred at the Chaparral WMA, the area received roughly 12 cm of rainfall (Chaparral WMA, personal communication). This rainfall, while not enough to sustain soil moisture for a substantial period of time, may have temporarily helped prevent further tree mortality from the wildfire and reduced the impact of the fire on the vegetation and faunal communities. However, rainfall soon subsided and drought conditions prevailed from November 2008 through the growing season and into winter of 2009. Soil moisture is crucial for herbaceous and woody plants, especially during the growing season, in order for plants to store enough energy to survive winter dormancy (Harrington 1991), and a lack thereof may have led to further mortality of trees and shrubs already weakened by fire damage. This increase in mortality may explain the decrease in visual obstruction height from 2009 (Y1) to 2010 (Y2).

Drought has also been shown to negatively affect grasses, and therefore may have also played a role in the decline in percent grass cover.

Avian Community

An increase in heterogeneity in vegetation structure, density and height may help to explain how burned sites had significantly higher species richness and twice as many observations as unburned sites. A mosaic of vegetation seral stages created by intermediate disturbances can provides habitat to the greatest number of species (Rosenstock 1999, Fuhlendorf et al. 2006).

Most wintering migratory sparrows that can be observed in the south Texas region depend primarily on seeds for winter forage (Dunning & Brown 1982). A number of resident south Texas birds also rely on seeds for a portion of their wintering forage, and the increase in forbs provided a substantial seed source for these birds. An increase in forb abundance could have also positively affected arthropod species richness and abundance (Siemann 1998), which, coupled with abundant seed production, could help explain increases in overall density and/or probability of presence in Y2 for White-crowned sparrows (*Zonotrichia leucophrys*), Western meadowlarks (*Sturnella neglecta*), Lincoln's Sparrow (*Melospiza lincolnii*), Blue-gray gnatcatcher (*Polioptila caerulea*), Black-tailed Gnatcatcher (Polioptila melanura), Black-throated Sparrow (Amphispiza bilineata) Bewick's Wren (*Thryomanes bewickii*), Eastern Phoebe (*Sayornis phoebe*), Green-tailed Towhee (*Pipilo chlorurus*). Another study focusing on the effects of patch burning on grassland birds and arthropoda also reported higher species diversity of both avian and arthropod communities on burned sites 1-2 years post-burn (Doxon 2009).

A reduction of litter and an increase in bare ground made the potential for seed exposure greater on burned sites. The return of rainfall in winter 2009 increased soil moisture beyond drought levels and produced an abundance of grasses and forbs on both study areas, which, coupled with increased seed exposure, may have also made burned sites a more conducive wintering forage area to migratory grassland sparrows such as Vesper sparrow (*Pooecetes gramineus*), White-crowned sparrow (*Zonotrichia leucophrys*) and other resident granivorous, ground-foraging species (Best 1979, Woinarski et al. 1999). These results are consistent with another south Texas study by Reynolds and Krausman (1998), where relative abundance of wintering, granivorous birds was higher on burned sites. Higher densities of migratory grassland species such as Vesper Sparrows (*Pooecetes gramineus*) and White-crowned Sparrows (*Zonotrichia leucophrys*) on burned sites further supports the prediction that wildfire may have made these areas more habitable for ground-foraging birds. *P. gramineus* and *Z. leucophrys* are both granivorous, ground-foraging species that prefer open, grassy areas with intermittent bare patches of ground (Grzybowski 1983). The characteristics of the groundcover post-fire were more consistent with grassland obligate, ground-foraging species' habitat requirements, and increases in densities of these birds post-fire have also been observed in other fire studies (Bock et al. 1976, Kirkpatrick et al. 2002, Lee 2006).

An increase in density of mesquite (*Prosopis glandulosa*) and other shrubs in south Texas may play a role in reducing quality habitat for avian species that require a grassland component in their habitat (Lloyd et al. 1998). Reductions in tree and shrub height and density on burned sites may have opened up the area to more closely resemble migratory species preferred grassland habitats. Similar results have been reported where juniper (*Juniperus* spp.) has invaded Arizona grasslands, where grassland-obligate species richness was negatively correlated with juniper encroachment (Rosenstock 1999). Grassland-obligate birds, such as Grasshopper sparrow (*Ammodramus savannarum*), Savannah sparrow (*Passerculus sandwichensis*), and Vesper sparrow (*Pooecetes granimeus*) accounted for the majority of differences between species richness and composition for burned and unburned sites (Appendix I, pg. 66). Coppedge et al. (2008) studied the effects of patch burning on density and abundance of birds, and observed that patch burned sites had higher grassland obligate species richness, as well as total species diversity.

Results for individual species probability of presence also show that the wildfire enhanced the burned sites for granivorous birds that also require at least a nominal shrub component in their wintering habitat. Cassin's Sparrow (*Aimophila cassinii*) generally prefer wintering habitat similar to their breeding grounds (Oberholser 1974), with a mosaic of shrubs within savanna, as shrubs are valuable for song-perching (Sampson & Knopf 1996). Even though the fire initially decreased visual obstruction, *A. cassinii* were still more than three times as likely to be found on burned sites. Pyrrhuloxia (*Cardinalis sinuatus*) are relatively nonmigratory, and inhabit open areas with mesquite trees (*Prosopis. glandulosa*) and other shrubs for nesting during the breeding season (Bent 1968). *C. sinuatus* are primarily ground foragers and seeds are a part of their diet, which would explain their higher probability of presence on burned sites, despite the reduction in visual obstruction. Bewick's wren (*Thryomanes bewickii*) prefers more dense woody vegetation; however, although probability of presence was over 3 times higher on unburned sites, results also indicated an overall increase in probability of presence of *T.*

bewickii on both burned and unburned sites by the second year. While visual obstruction height (or woody plant density) was reduced on burned sites, it may not have been reduced enough to be a limiting factor in forage or shelter availability for shrub-obligate species such as *T. bewickii* for more than 1 year. Kirkpatrick et al. (2002) reported similar results regarding shrub-obligate species on his Arizona study area, where the magnitude of post-fire change in individual species density and presence for shrub-obligate birds was minimal compared to the more obvious impacts that were observed for grassland-obligate species (Kirkpatrick et al. 2002). Other studies have also reported only minimal effects from fire on shrub-obligate species, particularly when fires are patchy in intensity and coverage (Fitzgerald & Tanner 1992). As *T. bewickii* was the only species to have a higher density and/or probability of presence on unburned sites, it is justifiable to say that my prediction that shrub-foraging species would decrease in density and presence on burned is unsupported by the data. Although the fire positively affected wintering migratory sparrows and other ground-foraging species, it did so without significant negative impact on resident shrubland species.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Habitat loss through land use changes has often been implicated as a primary reason for the decline in grassland bird abundance (Knopf 1994). Habitat degradation of south Texas shrub-grasslands through fire suppression and brush encroachment is also a substantial threat to both grassland bird breeding and wintering habitat (Lloyd et al. 1998, Grant et al. 2004). Although the wildfire on the Chaparral WMA was large and intense, it burned patchy in areas, and this mosaic pattern increased structural heterogeneity of woody plants and made habitat more suitable for wintering grassland birds without serious impact on resident shrubland species. Despite the intensity of the wildfire, percent cover of most groundcover components was similar on burned and unburned sites within 2 years post-fire, which is similar to recovery times for prescribed fire in this region. Private and public land managers wanting to increase or sustain avian species richness for the purposes of ecotourism or gamebird production should utilize prescribed fire, when possible, and maintain their land in a variety of post-fire successional stages to maintain habitat heterogeneity for a diversity of game and nongame avian species.

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Table 1.1 Factors (Treatment, Time, or Treatment x Time) affecting visual obstruction height and percent cover of vegetation components during winter seasons on burned and unburned sites (*n*) following 2008 wildfire, based on generalized linear mixed model. *n* = 40 (10 burned sites, 10 unburned sites, over 2 years). I removed interactive terms from models when $P > 0.10$. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA. Estimates for significant effects are provided in Table 1.2.

	Treatment		Time		Treatment x Time	
	F	*P*	*F*	*P*	*F*	*P*
Visual Obstruction Height	13.12	0.0019	8.41	0.0092	–	
Total Cover	4.46	0.0490	24.10	0.0001	28.29	0.0001
Bare Ground	2.38	0.1405	14.05	0.0015	19.90	0.0003
Litter	15.79	0.0009	43.15	0.0001	–	
Forbs	25.09	0.0001	566.35	0.0001	25.84	0.0001
Total Grass	0.12	0.7360	57.56	0.0001	3.57	0.0750
Native Grass	5.36	0.0326	4.95	0.0383	–	
Non-native Grass	12.85	0.0021	2.07	0.1668	–	

Table 1.2. Means for visual obstruction height and percent cover of vegetation components during winter seasons on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear mixed model. B = burned, U = unburned, Y1 = Year 1 post-fire (2009), Y2 = Year 2 post-fire (2010). n = 40 (10 burned sites, 10 unburned sites, over 2 years). Estimates are percentages[a] (above) with 95% confidence intervals (below) for significant effects from either Treatment, Time, or an interacting Treatment x Time. I removed interactive terms from models when $P > 0.10$. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA.

	Treatment		Time		Treatment x Time			
	B	U	Y1	Y2	BY1	BY2	UY1	UY2
Visual Obstruction Height [a]	1.5 (1.4-1.6)	1.7 (1.6-1.8)	1.7 (1.6-1.8)	1.5 (1.5-1.6)	–			
Total Cover	–		–		68.3 (64.6-72.2)	83.3 (78.7-88.1)	81.1 (76.6-85.7)	80.41 (76.0-85.1)
Bare Ground	–		–		33.1 (26.7-41.0)	17.4 (14.1-21.5)	19.4 (15.7-24.0)	20.5 (16.5-25.4)
Litter	31.8 (29.1-34.8)	40.3 (36.9-44.1)	43.3 (39.3-47.2)	29.7 (27.2-32.3)	–			
Forbs	–		–		9.0 (7.3-11.1)	45.4 (36.7-56.1)	3.3 (2.7-4.0)	40.0 (32.4-49.5)
Total Grass	–		–		18.7 (14.0-25.0)	8.8 (6.6-11.7)	22.9 (17.2-30.6)	6.5 (4.9-8.7)
Native Grass	77.2 (70.8-84.1)	88.3 (81.0-96.2)	88.7 (81.0-97.1)	76.8 (70.2-84.1)	–			
Non-native Grass	19.2 (14.1-26.2)	9.1 (6.7-12.4)	–		–			

[a] Visual obstruction height measured in meters (m).

Table 1.3 Factors (Treatment, Time, or Treatment x Time) affecting probability of presence of wintering avian species on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear mixed model. *n* = 40 (10 burned sites, 10 unburned sites, over 2 years). I removed interactive terms from models when *P* > 0.10. I did not detect any Treatment x Time interactions. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA. Estimates for significant effects are provided in Table 1.4.

	Treatment		Time	
	F	P	F	P
Blue-gray Gnatcatcher	1.53	0.2326	6.74	0.0178
Black-tailed Gnatcatcher	2.15	0.1596	5.5	0.0301
Black-throated Sparrow	2.22	0.1534	3.41	0.0804
Cassin’s Sparrow	8.38	0.0096	0.00	0.9985
Bewick’s Wren	4.19	0.0556	5.62	0.0284
Cactus Wren	1.53	0.2326	6.74	0.0178
Eastern Phoebe	0.01	0.9191	3.6	0.0731
Green-tailed Towhee	1.58	0.2243	11.31	0.0033
Ladder-backed Woodpecker	7.22	0.0151	1.79	0.1964
Lincoln’s Sparrow	0	0.9983	16.08	0.0007
Loggerhead Shrike	0.49	0.49	2.03	0.1702

Table 1.3 (Continued)

	Treatment		Time	
	F	P	F	P
Northern Cardinal	1.01	0.3291	0.99	0.3329
Orange-crowned Warbler	0.00	0.9985	0.00	0.9984
White-crowned Sparrow	0.00	0.9986	0.00	0.9986
Western Meadowlark	0.14	0.7122	12.22	0.0024
Vesper Sparrow	0.00	0.9986	0.4	0.5324
Pyrrholuxia	3.77	0.0680	1.47	0.2401
Verdin	0.19	0.6697	0.09	0.7719

Table 1.4 Means for percent probability of presence (above) and 95% confidence intervals (below) for wintering avian species by variables (Treatment or Time) on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear mixed model. B = burned, U = unburned, Y1 = Year 1 post-fire (2009), Y2 = Year 2 post-fire (2010). *n* = 40 (10 burned sites, 10 unburned sites, over 2 years). I removed interactive terms from models when *P* > 0.10. I did not detect any Treatment x Time interactions. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, Dimmit and La Salle counties, Texas, USA.

	Treatment		Time	
	B	U	Y1	Y2
Blue-gray Gnatcatcher	–		18.0 (5.6-45.6)	60.1 (34.2-81.4)
Black-tailed Gnatcatcher	–		22.9 (8.1-50.0)	60.5 (34.7-81.6)
Black-throated Sparrow	–		71.9 (47.3-87.9)	95.8 (72.8-99.5)
Cassin's Sparrow	35.0 (18.7-55.8)	10.0 (2.8-30.1)	–	
Bewick's Wren	31.6 (12.1-60.6)	72.2 (43.0-90.0)	31.5 (13.1-58.4)	72.3 (45.2-89.2)
Eastern Phoebe	–		5.0 (0.6-32.4)	35.0 (16.4-59.6)
Green-tailed Towhee	–		13.2 (3.4-39.6)	65.9 (39.4-85.2)
Ladder-backed Woodpecker	76.3 (45.3-92.6)	19.5 (5.4-50.9)		–
Lincoln's Sparrow	–		0.4 (0.0-100.0)	34.7 (0.0-100.0)
Western Meadowlark	–		9.9 (2.1-36.1)	65.0 (39.8-83.9)
Pyrrholuxia	50.0 (24.4-75.6)		14.3 (3.3-44.6)	–

Table 1.5 Factors (Treatment, Time, or Treatment x Time) affecting species richness and density of wintering birds on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear model. n = 40 (10 burned, 10 unburned, over 2 years). I removed interactive terms from models when $P > 0.10$. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA. Estimates for significant effects are provided in Table 1.6.

	Treatment		Time		Treatment x Time	
	F	*P*	*F*	*P*	*F*	*P*
Species Richness	4.70	0.0437	23.31	0.0001	–	
Black-throated Sparrow	1.76	0.2013	15.33	0.0009	–	
Northern Mockingbird	0.14	0.7125	1.83	0.1927	4.61	0.0457
Vesper Sparrow	14.31	0.0014	3.85	0.0653	3.85	0.0653
White-crowned Sparrow	57.10	0.0001	37.36	0.0001	8.33	0.0098
Western Meadowlark	0.07	0.7894	16.37	0.0007	–	

Table 1.6 Means for avian species richness and wintering avian species densities on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear mixed model. B = burned, U = unburned, Y1 = year 1 post-fire (2009), Y2 = year 2 post-fire (2010). *n* = 40 (10 burned sites, 10 unburned sites, over 2 years). Estimates are # of individuals per hectare[a] (above) with 95% confidence intervals (below). I removed interactive terms from models when *P* > 0.10. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, Dimmit and La Salle counties, Texas, USA.

	Treatment		Time		Treatment x Time			
	B	U	Y1	Y2	BY1	BY2	UY1	UY2
Species Richness[a]	19.0 (15.6-23.3)	14.2 (11.6-17.3)	12.5 (10.4-15.0)	21.6 (17.9-26.0)	–			
Black-throated Sparrow	–		1.3 (1.1-1.4)	1.6 (1.5-1.8)	–			
Northern Mockingbird	–		–		1.4 (1.2-1.5)	1.3 (1.1-1.5)	1.3 (1.1-1.5)	1.5 (1.3-1.7)
Vesper Sparrow	–		–		1.2 (1.0-1.3)	1.4 (1.3-1.6)	1.0 (0.9-1.1)	1.0 (0.9-1.1)
White-crowned Sparrow	–		–		1.6 (1.3-2.1)	4.4 (3.5-5.6)	1.0 (0.8-1.3)	1.4 (1.1-1.8)
Western Meadowlark	–		1.0 (0.96-1.1)	1.2 (1.1-1.3)	–			

Species richness = Estimated number of species

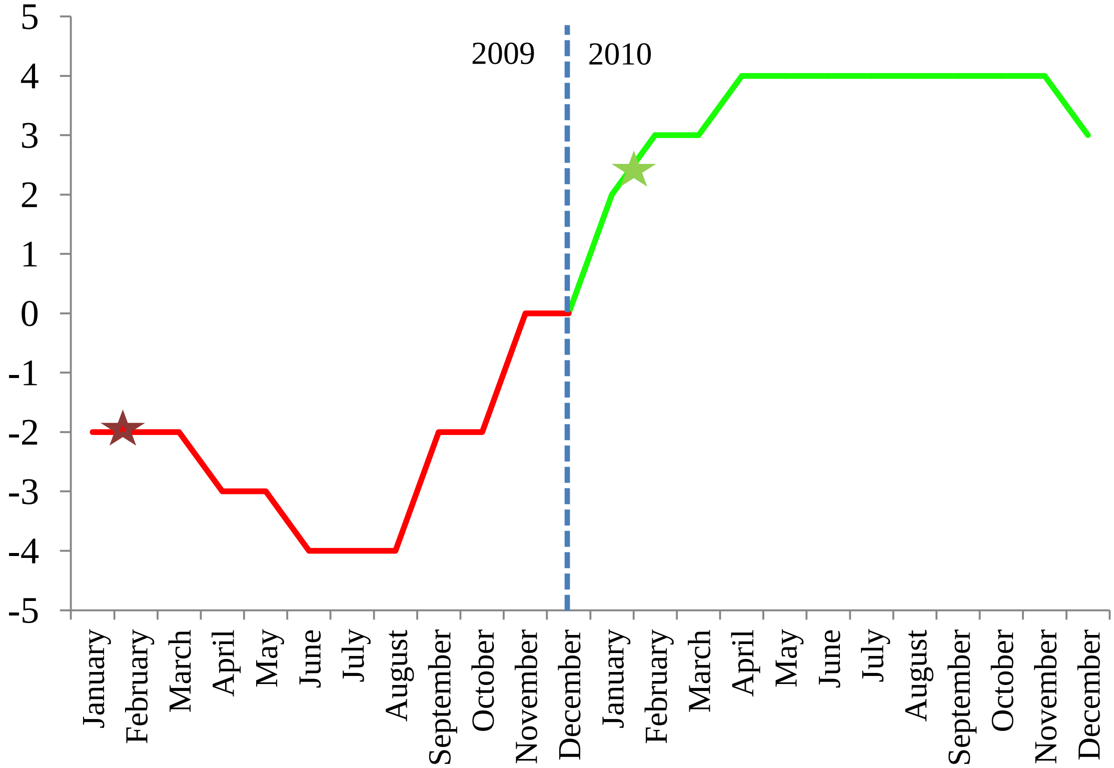

Figure 1.1 Monthly Palmer Drought Severity Index (PDSI) Values (left axis) for study areas. Values below 0 indicate drought. Avian and vegetation surveys occurred during January-February 2009 – 2010. Chaparral WMA and Piloncillo Ranch, Dimmit and La Salle counties, Texas, USA.

CHAPTER II

IMPACTS OF WILDFIRE ON BREEDING AVIAN COMMUNITIES OF SOUTH TEXAS

INTRODUCTION

Periodic fire is responsible for shaping a substantial number of ecosystems throughout North America. The grassland ecosystem is an example of a system that is dependent on fire in maintaining both floral and faunal composition. The suppression of fire in a grassland system can cause woody plant species to proliferate and alter the composition of the vegetation community (Box et al. 1967, Cable 1967). It is difficult to ascertain the historic fire regime for the south Texas Rio Grande Plains, although it appears that fire-return intervals for the western and southwestern semi-arid grasslands ranged from every 7-10 years (McPherson 1995) to once every 25 years (Wright & Bailey 1982). Using a growth rate model for mesquite (*Prosopis glandulosa*) based on varying precipitation patterns, Archer (1989) concluded that woody plant encroachment began in south Texas in the late 1800s, which eventually produced a grassland-shrub community classified as *Prosopis-Acacia-Andropogon-Setaria* savanna (Küchler 1964). The continued use of this region as breeding grounds by migratory and resident avian species, despite the variation in structure and composition of the vegetation community over time, illustrates the plasticity in habitat requirements of these birds (Reinking 2005).

When fire does occur in mesquite grasslands, its effects are similar to those observed in other fire-maintained ecosystems, in that fire temporarily reduces groundcover and woody plant height and/or density (Cable 1967). Although some fire effects are purely beneficial for birds, some may also adversely affect certain avian species. Fire removes litter and debris, which increases exposure of seeds and insects to ground-foraging birds (Bock et al. 1976, Best 1979). Although fire can be beneficial for providing ideal ground-foraging conditions, the removal of litter and debris may limit the available nesting substrate to a level unsuitable for some ground-nesting birds. For instance, Grasshopper Sparrows (*Ammodramus savannarum*) are typically found in tall, dense grass, and may leave an area for 1-2 years post-fire (Bock & Bock 1992*a*, Hands 2007). Fire also has the potential to decrease shrub density and height (Bock & Bock 1992*b*, Ruthven & Synatzske 2002). Although this may be beneficial to grassland-obligate species that prefer more open nesting areas, a decrease in shrub density may be a limiting factor for shrub-nesting species. Any sudden and significant change in availability of nesting substrate can mean a reduction in overall breeding productivity, due to an increase in nest failure or predation as a result of using substandard nest substrates, or simply because fewer individuals choose to nest. For example, in a study focusing on nesting ecology of Scissor-tailed Flycatchers (*Tyrannus forficatus*) in south Texas, in an area altered by brush management activity, 8 *T. forficatus* nests were placed in dead shrubs, and in all cases the nests failed (Nolte and Fullbright 1996).

Understanding the effects of fire on the south Texas avian community is important, not only for the inherent ecological value of these birds, but for their economic value as well. Declines in breeding season productivity can have major implications for the local avian community, and decreases in populations of several species in a community can have even greater impacts on both the ecosystem and the local economy.

According to The National Survey on Recreation and the Environment, 33% of Americans participated in birding one or more times between 2000-2001 and this represents a 12% increase from 1982-1983 (U.S. Forest Service 2011). Because Texas is largely considered one of the top birding destinations in the country (Mathis & Matisoff 2004) it is important to understand how stochastic natural events such as wildfire impact this valuable environmental and economic commodity.

Information on the effects of warm-season wildfires in Texas is limited, because most research has focused on cooler-season prescribed fire. My objectives for this study were to investigate how an intense, warm-season wildfire impacted both the avian and vegetation communities of the western Rio Grande Plains of south Texas and examine how changes in vegetation may have affected breeding season avian species composition, abundance and richness. I developed three predictions relevant to the research objectives:

I. Reductions in abundance and presence of shrub-nesting birds will be observed on burned sites.

II. Increases in abundance and presence of ground foraging avian species will be observed on burned sites.

III. Burned sites will have greater bird species richness than unburned sites.

METHODS

Study Area

Study areas included the 6,151-ha Chaparral Wildlife Management Area (28° 19' N, 99° 24' W), managed by Texas Parks and Wildlife Department, and the Piloncillo Ranch (28° 17' N, 99° 17' N), located in LaSalle and Dimmit counties in the western Rio

Grande Plains of south Texas. I used a 2,460-ha pasture of the Piloncillo Ranch that was adjacent to the Chaparral Wildlife Management Area (WMA) as the unburned control for this study. Soil types were similar for both study areas and included Dilley fine sandy loam, Duval loamy fine sand, Duval very fine sandy loam, and Duval fine sandy loam (NRCS 2011). A possible confounding factor in the study was a six-month cattle lease in the pasture where all unburned sites were located. Grazing occurred between November 2009 – April 2010, and cattle were removed just before breeding season data collection occurred (May – June).

Study sites were representative of south Texas Rio Grande Plains mixed-brush community (McLendon 1991). Dominant woody species included mesquite (*Proposis glandulosa*), granjeno (*Celtis pallida*), brasil (*Condalia hookeri*) and hogplum (*Colubrina texensis*). Other woody species included cactus (*Opuntia* spp.), huisache (*Acacia minuta*), leatherstem (*Jatropha dioica*), and Texas persimmon (*Diospyros texana*) (Ruthven et al. 2003). Common herbaceous species included hooded windmill grass (*Chloris cucullata*), Lehman lovegrass (*Eragrostis lehmanniana*), grama grasses (*Bouteloua* spp.), partridge pea (*Chamaescrista fasciculata*), and plains lazy daisy (*Aphanostephus ramosissimus*).

Climate was characterized by temperatures ranging from an average daily minimum of 5°C in January to an average daily maximum of 37°C in July. Ten year (2000-2009) rainfall for this area averaged 58 cm (Chaparral WMA unpublished data). Precipitation pattern is bimodal, with most rain occurring in late spring and early fall (Stevens & Arriaga 1985); however, surveys for summer 2009 (May – June) were conducted during a 16-month drought in the South Texas region (Figure 2.1, pg. 65). Palmer Drought Severity Index (PDSI) values (Palmer 1965) ranged from -2 (moderate drought) to -4 (extreme drought) throughout most of 2009 (Chaparral WMA unpublished data). Although the region was still in drought levels, the study sites did receive sufficient rainfall in April and May 2009 to produce nominal forb cover and sustain vegetation through the summer. By November 2009, above-average rainfall alleviated drought conditions and summer 2010 vegetation and avian surveys were conducted during May and June with PDSI values of 3 and 4, respectively.

Vegetation Surveys

Point count stations were established to collect both avifaunal (Ralph et al. 1995) and vegetation data. I established 20 point count stations on each study area. Point count stations were selected randomly, while still meeting chosen criteria of 30-50% brush cover and native grass dominance (less than 30% exotic grass species) within a 50-m radius of the point count center. Due to ongoing brush management activities at the Chaparral WMA, I established two point count stations near each line transect established during the winter portion of this study, which would reduce the likelihood of stations being affected by mowing or aeration activity.

To examine vegetation height and density, I quantified visual obstruction height by placing a Robel pole (Robel et al. 1970) at 10 random locations within the 50-m radius of each point count center. Visual obstruction height (m) was estimated 15 m from the pole in each of the four cardinal directions for a total of 40 measurements per point count station (800 per treatment). I also estimated groundcover composition at each of the pole locations using a Daubenmire frame (Daubenmire 1959) for a total of 10 measurements per point count station (200 per treatment). Percentages were estimated inside each frame for litter, bare ground, grasses, and forbs. I also estimated live woody canopy cover and woody species composition using the line intercept method (Bonham 1989). Two 20-m lines were randomly placed within the 50-m radius of the point count center. In order to avoid trampled vegetation and maintain a thorough survey effort, lines did not cross each other or over the point count center. All live woody vegetation that intercepted the line was identified and measured to obtain percent horizontal coverage along the line.

Avian Point Count Surveys

To decrease the possibility of repeated observations, all point count stations were separated by at least 300 m. No point count stations were placed near standing water or man-made infrastructure. Surveys were conducted at each station for 7 minutes, with a 2-minute waiting period to allow birds to settle after the initial arrival disturbance. I recorded all aural and visual detections of identified avian species within 50 m of the point count center (Appendix II, pg. 68). Aural detections were later utilized only for presence analysis. Exact visual distance measurements were obtained using a laser range-finder. Surveys began 30 minutes after sunrise and continued for 3 hours. Avian surveys were not conducted on days with high winds (> 20 km/h), heavy fog or precipitation. To account for hourly variation in avian activity, I altered the order in which each point count was surveyed. I conducted surveys at each station 4 times in 2009, and 5 times in 2010.

STATISTICAL ANALYSIS

I used Program DISTANCE 6.0 release 2 (Thomas et al. 2010) to compute model-averaged density estimates for bird species with > 40 total observations between 2009 and 2010. Observations for each species were binned into unique distance categories to achieve greater model fit. I assessed models with 3 possible key functions (Uniform, Half Normal, and Hazard-rate) with the appropriate expansions (Cosine, Simple Polynomial, or Hermite Polynomial). Models were chosen based on Akaike's information criterion values adjusted for small sample sizes (AICc) (Littell et al. 2006). I generated estimates of species richness using Program SPECRICH (Burnham & Overton 1979). Probability of presence estimates were obtained for each avian species that had visual or aural observations on 20-80% of total point counts. Avian species richness, individual species densities, and vegetation community characteristics were natural log-transformed to meet assumptions of homogeneity of variances. Vegetation community characteristics included visual obstruction height, percent abundances of litter, bare ground, grass, forbs, and total cover (combined percentages of grass, forbs, and litter). I compared avian and vegetation estimates between treatments and years using a generalized linear mixed model with the appropriate distribution and link function. Explanatory variables used in the analysis included Time (since burn, Year 1 and Year 2), Treatment (Burned and Unburned), and an interactive Time x Treatment term. If the interactive term was not significant, I removed it from the model and ran the analysis again using only Treatment and Time. I assessed 4 covariance structures (compound symmetric, first-order autoregressive, first-order autoregressive moving average, and toeplitz) and chose the appropriate structure based on AICc values (Littell et al. 2006). I used $P = 0.10$ as the cutoff for significance in order to explore all possible trends and causal relationships in the data. I also compared heterogeneity (patchiness) in total cover and visual obstruction height within burned sites and unburned sites for 2009 – 2010 by calculating a coefficient of variation and 95% confidence intervals for each study area.

RESULTS

Vegetation Community

Patterns of change over time differed between treatments for visual obstruction height, live woody cover, total cover, forbs, litter, bare ground, total grass cover, and native grass cover (Table 2.1, pg. 56). Visual obstruction height (m) was significantly lower on burned sites than unburned sites in Year 1 (Y1) and Year 2 (Y2); additionally, visual obstruction height increased over time from 1.8 m (95% CI = 1.7 – 1.9) to 2.0 m (95% CI = 1.9 – 2.1), whereas visual obstruction height on unburned sites stayed relatively the same (Table 2.2, pg. 57). Live woody percent cover was significantly lower on burned sites in 2009; however, coverage increased on burned sites from Y1 to Y2, while cover on unburned sites decreased slightly over time, leading to relatively similar woody percent coverage values (Table 2.2, pg. 57) between treatments by Y2. Total cover (grasses, forbs, and litter) increased from 53.2% (95% CI = 49.0-57.8) to 82.4% (95% CI = 75.9-89.5) on burned sites from Y1 to Y2, while total percent cover on unburned sites stayed relatively the same over time, leading to similar values between treatments by Y2. Forb cover increased from roughly 7% for both study areas in Y1, to 18.9% for burned sites (95% CI = 16.1-22.1) and 25% for unburned sites (95% CI = 21.6-29.8) in Y2. Litter was initially lower on burned sites in Y1, but by Y2, percentages were similar between burned and unburned sites (Table 2.2, pg. 57). Although bare ground was almost twice as abundant on burned sites (45.4, 95% CI = 35.7-57.7) compared to unburned sites (23.6, 95% CI = 18.5-30.0) in Y1, percentages were also similar between treatments by Y2. Total grass cover was slightly lower on burned sites in Y1; however, while grass cover increased slightly on the burned sites over time (Table

2.2, pg. 57), I observed a sharp decrease in cover for unburned sites between Y1 (16.5, 95% CI = 12.3-21.3) and Y2 (3.5, 95% CI = 2.7-4.5). I observed the same negative response over time for percent cover of native grass on unburned sites, while coverage on burned sites remained the same from Y1 to Y2. Percent cover of non-native grass was 12.3% on burned sites (95% CI = 7.3-20.7), compared to 6.5% for unburned (95% CI = 3.9-10.8), although the relationship was relatively weak (P = 0.0818).

Visual obstruction height (vegetation height and density) on burned sites (coefficient of variation: 48.00, 95% CI = 42.72 – 53.28) was significantly more heterogeneous than unburned sites (34.39, 95% CI = 28.09 – 40.68) in 2009, but study areas became more similar in Y2 with increased time since fire (burned: 31.44, 95% CI = 26.84 – 36.03; unburned: 30.79, 95% CI = 27.53 – 34.06). Heterogeneity (patchiness) in total cover was also higher on burned sites (43.57, 95% CI = 38.77 – 48.37) than unburned sites (30.23, 95% CI = 24.66 – 35.81) in 2009; but by Y2, heterogeneity was similar between burned (21.36, 95% CI = 17.84 – 28.88) and unburned sites (22.37, 95% CI = 17.07 – 31.44).

Avian Community

I collected a total of 1,053 observations for summer 2009 and 955 observations for summer 2010 for a total of 2,008 observations. Observations for burned and unburned sites were similar in 2009 (535 and 515, respectively), although in 2010 observations on burned sites were almost twice as high as unburned sites (609 compared to 346). I observed a total of 45 bird species during the summer seasons 2009 and 2010. In summer 2009, I observed 39 species in the burned sites and 37 species in the unburned sites, with 77% of the species observed on both study areas. In summer 2010, I observed 32 species on the burned sites and 29 species on the unburned sites, with a 79% species overlap on both study areas.

Species richness – I did not detect a difference in species richness between treatments; however, overall species richness for both treatments decreased substantially from an average of 23 species in Y1 (95% CI = 20.7 – 25.8), to 17 species in Y2 (95% CI = 14.8 – 18.5).

Presence – I analyzed differences in probability of presence for 18 species (Table 2.3, pg. 59). I observed 2 species, Bewick's Wrens (*Thryomanes bewickii*) and Cassin's Sparrows (*Aimophila cassinii*), that had differing patterns of change over time between treatments. While *T. bewickii* had a substantially higher probability of presence on burned sites in Y1 (90.0%, 95% CI = 65.7-97.7) compared to unburned sites (70.0%, 95% CI = 45.9-86.5), presence in both study sites declined in Y2 to the point where probabilities were similar between treatments. Conversely, presence of *A. cassinii* was similar between treatments in Y1 (Table 2.4, pg. 61); however, while probabilities almost doubled in burned sites for Y2, they remained relatively unchanged for unburned sites.

Four bird species had significantly higher probabilities of presence on either burned or unburned sites. Presence of Ladder-backed Woodpecker (*Picoides scalaris*) was twice as high on burned sites than unburned sites (Table 2.4, pg. 61). Scissor-tailed Flycatchers (*Tyrannus forficatus*) were almost 3 times as likely to be found on burned sites compared to unburned sites (Table 2.4, pg. 61). Probabilities of presence were also higher for Pyrrhuloxia (*Cardinalis sinuatus*) on burned sites (90.0%, 95% CI = 73.6 – 96.7) compared to unburned sites (67.6%, 95% CI = 49.5 – 81.5). Conversely, presence of Bell's Vireo (*Vireo bellii*) was higher on unburned sites (52.4%, 95% CI = 31.0 – 73.1) compared to burned sites (13.6%, 95% CI = 4.2 – 36.3).

Two species, Common ground-doves (*Columbina passerine*) and Verdins (*Auriparus flaviceps*) differed significantly in presence between treatments in addition to between years. *C. passerine* were more than twice as likely to be found on burned sites than unburned sites (Table 2.4, pg. 61), although probabilities of presence decreased overall on both treatments from an average of 55.4% in Y1 (95% CI = 38.4 – 71.2) to 13.7% in Y2 (95% CI = 5.7 – 29.3). Presence of *A. flaviceps* were 3 times higher on unburned sites than burned sites, although probabilities also declined on both treatments from 38.4% in Y1 (95% CI = 23.5 – 55.9) to 17.1% in Y2 (95% CI = 7.9 – 33.1). I also observed significant declines in presence between Y1 and Y2 for Black-tailed gnatcatcher (*Polioptila melanura*), Brown-headed cowbird (*Molothrus ater*), Cactus wren (*Campylorhynchus brunneicapillus*), Greater roadrunner (*Geococcyx californianus*), Northern bobwhite (*Colinus virginianus*), Northern cardinal (*Cardinalis cardinalis*), Olive sparrow (*Arremonops rufivirgatus*), and Painted bunting (*Passerina ciris*) (Table 2.4, pg. 61). I did not observe any effect on presence for Brown-crested flycatcher (*Myiarchus tyrannulus*) or Mourning dove (*Zenaida macroura*) (Table 2.3, pg. 59).

Density – Out of 5 species that met criteria for density analysis, 3 species exhibited a difference in density trends over time between treatments (Table 2.5, pg. 63). Black-tailed Gnatcatcher (*Polioptila melanura*) density was slightly lower on burned sites in Y1, with 1.1 birds/ha (95% CI = 0.9 – 1.3) compared to 1.4 birds/ha on unburned sites (95% CI = 1.2 – 1.7), but by Y2, density of *P. melanura* on burned sites increased and was similar to that of unburned sites (Table 2.6, pg. 65). I observed slightly higher densities of Northern Mockingbirds (*Mimus polyglottos*) on burned sites (1.5 birds/ha, 95% CI = 1.3 – 1.8) compared to unburned sites (1.3 birds/ha, 95% CI = 1.1 – 1.5) in Y1; however, by Y2, *M. polyglottos* densities on burned sites increased substantially to 3.0 birds/ha (95% CI = 2.6 – 3.5), while densities on unburned sites only rose to 1.5 birds/ha (95% CI = 1.2 – 1.7).

Densities of Pyrrhuloxia (*Cardinalis sinuatus*) were also slightly higher on burned sites (1.5 birds/ha, 95% CI = 1.3 – 1.7) than on unburned sites (1.2 birds/ha, 95% CI = 1.0 – 1.4) in Y1, and by Y2, I observed a slight but significant increase in density on burned sites, while densities remained unchanged on unburned sites (Table 2.6, pg. 65). I did not observe any significant effects for Painted Bunting (*Passerina ciris*) or Black-throated Sparrow (*Amphispiza bilineata*) density (Table 2.5, pg. 63).

DISCUSSION

Vegetation Community

A number of significant differences in vegetation component percentages between treatments illustrates the effects of the wildfire on the vegetation community. Fire plays a key role in reducing litter, increasing bare ground, and decreasing woody plant density in rangelands (Box et al. 1967, Wright & Bailey 1982, Ruthven et al. 2003, Heisler et al. 2004). Although most studies focus on cool-season prescribed fire, I observed a similar 3 year recovery time from the warm-season wildfire for live woody cover, total cover, litter, and bare ground. Reynolds & Krausman (1998) also observed a rapid recovery of herbaceous vegetation in a south Texas mesquite-grassland approximately 6 months post-burn. Other south Texas fire studies have observed recovery of groundcover components by 1 year post-burn (Ruthven & Synatzske 2002, Mix 2004).

The substantial decrease in Y2 for total grass cover on unburned sites can be attributed to a six-month cattle lease in the pasture where all unburned sites were located. Grazing occurred between November 2009 – April 2010, and cattle were removed just before breeding season data collection occurred (May – June). The presence of cattle, even at light stocking rates, caused a rapid decrease of grass cover on all unburned sites. Although this confounding factor made it impossible to compare total grass cover between treatments in Y2, it is notable that grass cover for burned sites in Y2 (16.3%, 95% CI = 12.6 – 24.1) mirrored grass cover percentages on the undisturbed, unburned sites in Y1 (16.5%, 85% CI = 12.7-21.3). The possibility exists that, had grazing not occurred on control sites, grass cover would be similar between treatments in Y2, and would therefore reflect the same recovery rate as other groundcover components in this study.

Differences in abundances of non-native grass may have been due to the management histories of the study areas. The Chaparral WMA has a substantial number of paved roads, and is open to public vehicles during portions of the year. Increased vehicular traffic, as well as vehicular access to all pastures inside the WMA, would perpetuate the spread of any exotic grass introduced to the property. Because it is a private property, access to the Piloncillo Ranch is more restricted, and therefore the proliferation of exotic grasses in the pasture would be less likely to occur.

A significant reduction in visual obstruction for burned areas can be explained by the top-kill and subsequent defoliation of woody plants by the wildfire. The fire, while patchy in some areas, was sufficiently intense to reduce both vegetation height and density. Similar results for woody vegetation have been reported in other Texas fire studies (Scifres & Hamilton 1993, Reynolds & Krausman 1998, Mix 2004). Although the decrease in visual obstruction was an expected result of fire, further decreases in visual obstruction on both treatments from Y1 to Y2 pointed to another factor. One possible explanation is the drought in 2009, which may have suppressed new growth in woody species the subsequent growing season. In a study modeling possible variables influencing woody cover change, Fensham et al. (2005) reported that rainfall patterns and initial woody canopy cover were the sole factors in determining changes in woody under- and overstory cover, while fire and grazing practices provided less or no explanatory power. The authors concluded that any increase in cover corresponded with low initial cover when rainfall was above-average, and that decreases in cover typically occurred when initial cover was high, regardless of rainfall levels (Fensham et al. 2005). Because fire had not occurred on either study area for at least 5 years, woody vegetation density could be considered initially high, and therefore the possibility of drought negatively impacting woody vegetation characteristics in both study areas is entirely plausible.

Avian Community

Reductions in litter, increases in bare ground, and decreases in woody cover made the burned sites more conducive to a number of granivorous species. The removal of litter provides more access to patches bare ground, which in turn enhances foraging conditions for granivorous species due to increased seed exposure (Best 1979). This may explain the higher probabilities of presence on burned sites for species such as Common Ground-dove (*Columbina passerine*), Cassin’s Sparrow (*Aimophila cassinii*) and

Pyrrhuloxia (*Cardinalis sinuatus*). Increases in abundances of birds in this foraging guild following fire have also been reported in other studies (Reynolds and Krausman 1998, Davis et al. 2000, Mix 2004, Lee 2006).

Higher probabilities of presence for insectivorous birds such as Scissor-tailed Flycatcher (*Tyrannus forficatus*) and Ladder-backed Woodpecker (*Picoides scalaris*), and higher densities of Northern Mockingbirds (*Mimus polyglottos*) on burned units could be due to a possible increase in arthropod abundance post-fire. The stimulation of growth in herbaceous plants by fire, coupled with increased tree and shrub death following the fire may have enabled the burned units to support higher abundances of insects favored by these birds. In a study focusing on fire effects on arthropod and avian species in south Texas, Mix (2004) reported a significant decrease in arthropod abundance from pre-fire levels in the 5 weeks post-fire; however, overall abundance was either similar to or much higher than pre-fire values for all orders after the 5-week post-fire period. Potts et al. (2003) studied bees and their response to fire, and reported a similar pattern of initial catastrophic loss, followed by recolonization and a subsequent peak in bee diversity and abundance 2 years post-fire. Davis et al. (2000) also found higher abundances of 3 species of insectivorous bark-gleaning avian species on burned sites during both study years, which supports the findings by this study as well as the possibility that fire may have increased arthropod abundance on burned sites.

Although fire possibly stimulated arthropod populations on the burned sites, the increase in food availability may not have been enough to override the need for adequate nesting shrub cover for birds that require thick nesting foliage. Bell's Vireo (*Vireo bellii*) and Verdin (*Auriparus flaviceps*) are species that require dense shrub cover for foraging as well as nesting (Austin 1977, Budnik et al. 2000) and both species had higher probabilities of presence on unburned sites,. Decreases in woody cover on burned sites may have limited nest site availability, and therefore would have limited the presence of *V. bellii* and *A. flaviceps* on burned sites during the breeding season. Renwald (1978) studied the effects of fire on woody plant selection by nesting birds, and noted that avian species that are shrub-nesters, particularly those that prefer certain woody species, are particularly susceptible to shortages in available nesting sites following fire (Renwald 1978).

Although some studies have reported some negative effects of fire on a number of ground-nesting species (Reynolds and Krausman 1998, Kirkpatrick 2002), I did not detect any negative impacts from the fire on this nesting guild, although not all ground-nesting birds had sufficient observations for presence or density analysis and surveys were not conducted until one year post-fire. I observed similar probabilities of presence of Cassin's sparrow (*Aimophila cassinii*) on both treatments in Y1 and a substantial increase in presence on burned sites in Y2, indicating a positive impact from fire on *A. cassinii* from 1 – 2 years post-fire. Conversely, *A. cassinii* has been observed entirely avoiding burned areas for 2 years post-fire (Bock and Bock 1992) and declining in relative abundance by 217% on burned sites from 1 – 2 years post-fire (Kirkpatrick 2002). I failed to detect either positive or negative effects of fire for Mourning doves (*Zenaida macroura*), another ground-nesting bird, although other studies have reported positive effects of fire on this species (Bock and Bock 1976 and 1992*a;* Kirkpatrick 2002). Higher presence of Common ground-doves (*Columbina passerine*) on burned sites during breeding season also support the conclusion that after one year post-burn, the effects of the fire were not detrimental to ground-nesting species.

Probabilities of presence decreased over time for Brown-headed cowbird (*Molothrus ater*), Black-tailed gnatcatcher (*Polioptila melanura*), Cactus wren (*Campylorhynchus brunneicapillus*), Common ground-dove (*Columbina passerine*), Greater roadrunner (*Geococcyx californianus*), Northern bobwhite (*Colinus virginianus*), Northern cardinal (*Cardinalis cardinalis*), Olive sparrow (*Arremonops rufivirgatus*), Painted bunting (*Passerina ciris*), and Verdin (*Auriparus flaviceps*), regardless of treatment. A possible explanation for this phenomenon could be the drought conditions affecting south Texas in 2009, which led to a substantial reduction in forbs as a food source (only 7% of total groundcover) and minimal live or residual grass cover for nesting substrate and cover. A reduction in overwintering ground forage from the wildfire, in addition to an above-average number of days with temperatures exceeding 38° C in summer 2009 (Chaparral WMA, personal comm.) through the 2009 nesting season may have led to local population declines from 2009-2010 for these avian species. In a study examining the effects of drought and extreme temperatures on grassland birds, George et al. (1992) observed declines in species richness, as well as declines in density and nesting success for several bird species during the drought, and suggested that lower individual species densities were from lack of recruitment to the area due to poor quality of the vegetation. Lowered rates of nesting success could be due to energy constraints on the incubating female or heat stress (George et al. 1992). Nesting effort and success in Northern bobwhites are especially affected by extreme heat and drought conditions (Guthery et al. 1988, Hernandez et al. 2005). Because live woody plant coverage declined on both treatments from 2009-2010, it is also possible limited availability of nesting sites may have also led to local population declines for shrub-obligate species (Renwald 1978).

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Although the wildfire at the Chaparral WMA was large and intense, it was a patchy fire and produced a mosaic of groundcover in varying seral stages, as well as increased heterogeneity in woody vegetation height. Despite the fire's scale, intensity, and occurrence during extreme fire conditions (low humidity, high winds, etc.), recovery time for most vegetation components were similar to observed recovery times for prescribed fire in this region (Reynolds and Krausman 1998, Ruthven and Synatzke 2002, Mix 2004).Effects from the fire on birds were generally positive, and foraging quality on the burned sites increased for a number of grassland granivorous and insectivorous species. Shrub-obligates were relatively unaffected, with the exception of Verdin and Bell's Vireo. Drought was a confounding factor in this study, and led to reductions in likelihood of occurrence for a number of species, regardless of study area. Land managers looking to maintain or enhance avian species diversity should utilize prescribed fire to provide a mosaic of post-fire seral stages. Efforts should also be made to utilize prescribed fire opportunities in the early growing season (Ansley and Jacoby 1998) in order to more effectively control woody plant encroachment and density.

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Table 2.1 Factors (Treatment, Time, or Treatment x Time) affecting visual obstruction height and percent cover for vegetation components during summer seasons on burned and unburned sites (*n*) following 2008 wildfire, based on generalized linear mixed model. *n* = 80 (20 burned, 20, unburned, over 2 years). I removed interactive terms from models when *P* > 0.10. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA. Estimates for significant effects are provided in Table 2.2.

	Treatment		Time		Treatment x Time	
	F	*P*	*F*	*P*	*F*	*P*
Visual Obstruction Height	4.47	0.0410	50.76	0.0001	5.93	0.0197
Live Woody Cover	8.34	0.0064	0.07	0.7870	3.28	0.0779
Total Cover	10.37	0.0026	56.36	0.0001	29.47	0.0001
Bare Ground	4.04	0.0515	48.46	0.0001	16.55	0.0002
Litter	11.94	0.0014	14.02	0.0006	11.39	0.0017
Forbs	3.49	0.0694	212.32	0.0001	3.53	0.0680
Total Grass	24.94	0.0001	28.19	0.0001	52.20	0.0001
Native Grass	4.17	0.0480	6.25	0.0168	6.12	0.0180
Non-native Grass	3.19	0.0818	0.10	0.7566	–	

Table 2.2 Means for visual obstruction height and percent cover of vegetation components during summer seasons on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear mixed model. B = burned, U = unburned, Y1 = Year 1 post-fire (2009), Y2 = Year 2 post-fire (2010). *n* = 80 (20 burned sites, 20 unburned sites, over 2 years) Estimates are percentages[a] (above) with 95% confidence intervals (below) for significant effects from either Treatment, Time, or an interacting Treatment x Time. I removed interactive terms from models when $P > 0.10$. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA.

	Treatment		Time		Treatment x Time			
	B	U	Y1	Y2	BY1	BY2	UY1	UY2
Visual Obstruction Height[a]	–		–		1.8 (1.7-1.9)	2.0 (1.9-2.1)	2.3 (2.2-2.4)	2.2 (2.1-2.3)
Live Woody Cover	–		–		19.4 (15.6-24.0)	23.0 (18.6-28.6)	31.7 (25.5-39.2)	25.1 (20.2-31.1)
Total Cover	–		–		53.2 (49.0-57.8)	82.4 (75.9-89.5)	74.3 (68.4-80.7)	79.7 (73.4-86.6)
Bare Ground	–		–		45.4 (35.7-57.7)	16.5 (13.0-20.9)	23.6 (18.5-30.0)	18.1 (14.2-23.0)
Litter	–		–		30.0 (26.5-34.1)	43.3 (38.1-49.2)	45.9 (40.4-52.1)	46.8 (41.2-53.1)
Forbs	–		–		7.4 (6.3-8.7)	18.9 (16.1-22.1)	7.6 (6.5-8.9)	25.4 (21.6-29.8)
Total Grass	–		–		12.9 (10.0-16.7)	16.3 (12.6-21.1)	16.5 (12.7-21.3)	3.5 (2.7-4.5)

[a] Visual obstruction height measured in meters (m).

Table 2.2 (Continued)

	Treatment		Time		Treatment x Time			
	B	U	Y1	Y2	BY1	BY2	UY1	UY2
Native Grass	–		–		82.7 (57.2-119.5)	82.3 (56.9-118.9)	89.0 (61.6-128.6)	36.2 (25.0-52.3)
Non-native Grass	12.3 (7.3-20.7)	6.5 (3.9-10.8)	–		–			

Table 2.3 Factors (Treatment, Time, or Treatment x Time) affecting probability of presence of summer birds on burned and unburned sites over time (*n*) following 2008 wildfire, based on generalized linear mixed model. *n* = 80 (20 burned sites, 20 unburned sites, over 2 years). I removed interactive terms from models when *P* > 0.10. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA. Estimates for significant effects are provided in Table 2.4

Species	Treatment		Time		Treatment x Time	
	F	*P*	*F*	*P*	*F*	*P*
Bewick's Wren	0.38	0.5403	5.09	0.0299	3.36	0.0748
Brown-crested Flycatcher	0.94	0.3383	0.40	0.5333	–	
Bell's Vireo	6.28	0.0166	2.34	0.1339	–	
Brown-headed Cowbird	1.33	0.2558	11.59	0.0015	–	
Black-tailed Gnatcatcher	0.07	0.7966	6.82	0.0127	–	
Cassin's Sparrow	2.61	0.1142	3.55	0.0673	6.29	0.0165
Cactus Wren	0.20	0.6584	3.21	0.0809	–	
Common Ground-dove	3.66	0.0633	13.07	0.0008	–	
Greater Roadrunner	0.18	0.6748	9.32	0.0041	–	
Ladder-backed Woodpecker	5.72	0.0219	0.07	0.7999	–	
Mourning Dove	1.63	0.2094	2.49	0.1224	–	
Northern Bobwhite	0.07	0.7971	19.69	0.0001	–	
Northern Cardinal	0.24	0.6257	5.12	0.0292	–	
Olive Sparrow	2.41	0.1291	11.20	0.0018	–	

Table 2.3 (Continued)

Species	Treatment		Time		Treatment x Time	
	F	*P*	*F*	*P*	*F*	*P*
Painted Bunting	1.15	0.2905	2.89	0.0970	–	
Pyrrhuloxia	4.52	0.0401	0.10	0.7591	–	
Scissor-tailed Flycatcher	7.47	0.0096	1.60	0.2133	–	
Verdin	7.68	0.0086	4.48	0.0407	–	

Table 2.4 Means for percent probability of presence (above) and 95% confidence intervals (below) for summer birds by variables (Treatment, Time, Treatment x Time) on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear mixed model. B = burned, U = unburned, Y1 = Year 1 post-fire (2009), Y2 = Year 2 post-fire (2010). *n* = 80 (20 burned sites, 20 unburned sites, over 2 years). I removed interactive terms from models when *P* > 0.10. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, Dimmit and La Salle counties, Texas, USA.

Species	Treatment		Time		Treatment x Time			
	B	U	Y1	Y2	BY1	BY2	UY1	UY2
Bewick's Wren	–		–		90.0 (65.7-97.7)	50.0 (28.3-71.7)	70.0 (45.9-86.5)	65.0 (41.2-83.1)
Brown-crested Flycatcher	–		–		–			
Bell's Vireo	13.6 (4.2-36.3)	52.4 (31.0-73.1)	–		–			
Brown-headed Cowbird	–		77.9 (61.7-88.5)	39.7 (25.3-56.2)	–			
Black-tailed Gnatcatcher	–		30.0 (17.4-46.6)	62.5 (46.0-76.6)	–			
Cassin's Sparrow	–		–		50.0 (28.3-71.7)	95.0 (69.3-99.4)	65.0 (41.2-83.1)	55.0 (32.5-75.7)
Cactus Wren	–		67.5 (50.9-80.7)	47.5 (32.0-63.5)	–			
Common Ground-dove	43.6 (26.1-63.0)	20.2 (9.3-38.7)	55.3 (38.4-71.2)	13.7 (5.7-29.3)	–			

Table 2.4 (Continued)

Species	Treatment		Time		Treatment x Time			
	B	U	Y1	Y2	BY1	BY2	UY1	UY2
Greater Roadrunner	–		45.0 (29.9-61.2)	15.0 (6.6-30.4)	–			
Ladder-backed Woodpecker	35.0 (22.1-50.6)	12.4 (5.3-26.4)	–		–			
Mourning Dove	–		–		–			
Northern Bobwhite	–		72.5 (56.0-84.6)	20.0 (9.9-36.1)	–			
Northern Cardinal	–		29.9 (17.4-46.4)	9.9 (3.6-24.5)	–			
Olive Sparrow	–		44.8 (29.4-61.2)	9.3 (3.3-23.6)	–			
Painted Bunting	–		87.9 (73.1-95.1)	73.0 (56.5-84.9)	–			
Pyrrhuloxia	90.0 (73.6-96.7)	67.6 (49.5-81.5)	–		–			
Scissor-tailed Flycatcher	52.6 (35.4-69.2)	19.4 (9.1-36.8)	–		–			
Verdin	13.8 (5.8-29.6)	44.6 (28.7-61.7)	38.4 (23.5-55.9)	17.1 (7.9-33.1)	–			

Table 2.5. Factors (Treatment, Time, or Treatment x Time) affecting avian species richness and density of summer birds on burned and unburned sites (*n*) over time following 2008 wildfire, based on generalized linear mixed model. *n* = 80 (20 burned, 20 unburned, over 2 years). I removed interactive terms from models when *P* > 0.10. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA. Estimates for significant effects are provided in Table 2.6.

	Treatment		Time		Treatment x Time	
	F	*P*	*F*	*P*	*F*	*P*
Species Richness	1.72	0.1971	18.29	0.0001	–	
Black-Tailed Gnatcatcher	1.58	0.2160	6.22	0.0171	3.09	0.0868
Black-throated Sparrow	1.07	0.3075	0.13	0.7180	–	
Northern Mockingbird	26.89	0.0001	34.97	0.0001	13.88	0.0006
Painted Bunting	0.03	0.8578	0.82	0.3721	–	
Pyrrholuxia	27.65	0.0001	0.85	0.3615	4.71	0.0362

Table 2.6. Least squares means for avian species richness and summer bird density on burned and unburned sites (n) over time following 2008 wildfire, based on generalized linear mixed model. B = burned, U = unburned, Y1 = Year 1 post-fire (2009), Y2 = Year 2 post-fire (2010). n = 80 (20 burned sites, 20 unburned sites, over 2 years). Estimates are # of individuals per hectare[a] (above) with 95% confidence intervals (below). I removed interactive terms from models when $P > 0.10$. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, Dimmit and La Salle counties, Texas, USA.

	Treatment		Time		Treatment x Time			
	B	U	Y1	Y2	BY1	BY2	UY1	UY2
Species Richness[a]	–		23.1 (20.7-25.8)	16.6 (14.8-18.5)	–			
Black-Tailed Gnatcatcher	–		–		1.1 (0.9-1.3)	1.5 (1.3-1.8)	1.4 (1.2-1.7)	1.5 (1.2-1.8)
Black-throated Sparrow	–		–		–			
Northern Mockingbird	–		–		1.5 (1.3-1.8)	3.0 (2.6-3.5)	1.3 (1.1-1.5)	1.5 (1.2-1.7)
Painted Bunting	–		–		–			
Pyrrhuloxia	–		–		1.5 (1.3-1.7)	1.9 (1.6-2.1)	1.2 (1.0-1.4)	1.1 (1.0-1.3)

[a] Species richness = Estimated number of species

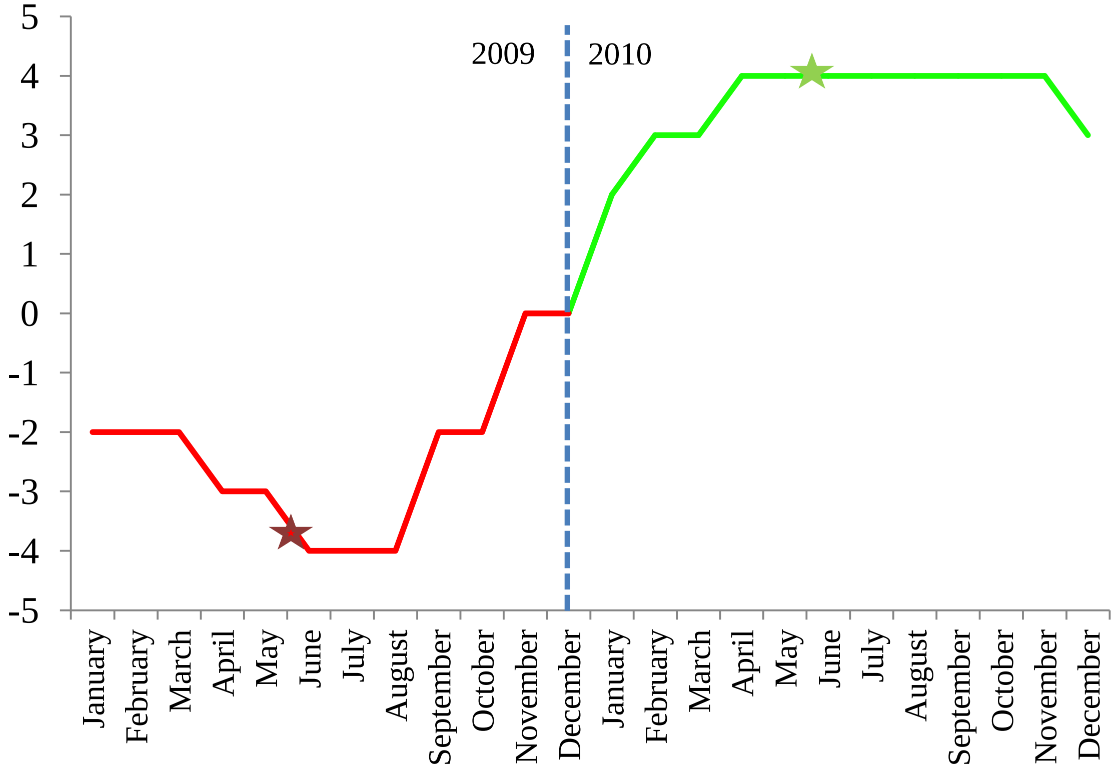

Figure 2.1 Monthly Palmer Drought Severity Index (PDSI) Values (left axis) for study areas. Values below 0 indicate drought. Avian and vegetation surveys occurred during May – June, 2009 – 2010. Chaparral WMA and Piloncillo Ranch, Dimmit and La Salle counties, Texas, USA.

Appendix I. List of observed winter species on each study area and year. BY1 = burned year 1, BY2 = burned year 2, UY1 = unburned year 1, UY2 = unburned year 2. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, Texas, USA

Species	Scientific Name	BY1	BY2	UY1	UY2
Turkey Vulture	*Cathartes aura*			X	X
Northern Harrier	*Circuscyaneus*	X		X	
Red-tailed Hawk	*Buteo jamaicensis*			X	
Crested Caracara	*Caracara cheriway*	X	X	X	X
American Kestrel	*Falco sparverius*			X	X
Northern Bobwhite	*Colinus virginianus*	X			
Mourning Dove	*Zenaida acroura*		X	X	
Common Ground-Dove	*Columbina passerina*	X			
Burrowing Owl	*Athene cunicularia*				X
Golden-fronted Woodpecker	*Melanerps aurifrons*	X		X	
Ladder-backed Woodpecker	*Picoidesscalaris*	X	X	X	X
Eastern Phoebe	*Sayornis phoebe*		X	X	X
Vermillion Flycatcher	*Pyrocephalus rubinus*	X	X		
Loggerhead Shrike	*Lanius ludovicianus*	X	X	X	X
White-eyed Vireo	*Vireo griseus*			X	X
Green Jay	*Cyanocorax yncas*		X		
Black-crested Titmouse	*Baeolophus atricristatus*				X
Verdin	*Auriparus flaviceps*	X	X	X	X
Bewick's Wren	*Thryomanes bewickii*	X	X	X	X
House Wren	*Troglodytes aedon*		X		
Cactus Wren	*Campylorhynchus brunneicapillus*	X	X	X	X
Ruby-crowned Kinglet	*Regulus calendula*		X		X
Black-tailed Gnatcatcher	*Polioptila melanura*	X		X	X
Blue-gray Gnatcatcher	*Polioptila caerulea*	X	X	X	X
Sage Thrasher	*Oreoscoptes montanus*	X	X	X	X

Appendix I. (Continued)

Species	Scientific Name	BY1	BY2	UY1	UY2
Curve-billed Thrasher	*Toxostoma curvirostre*	X			X
Long-billed Thrasher	*Toxostoma longirostre*		X		
Orange-crowned Warbler	*Vermivora celata*		X	X	X
Yellow-rumped Warbler	*Dendroica coronata*				X
Pyrrhuloxia	*Cardinalis sinuatus*	X	X	X	X
Northern Cardinal	*Cardinalis cardinalis*	X	X	X	X
Green-tailed Towhee	*Pipilo chlorurus*	X	X	X	X
Olive Sparrow	*Arremonops rufivirgatus*		X	X	
Cassin's Sparrow	*Aimophila cassinii*		X		X
Black-throated Sparrow	*Amphispiza bilineata*	X	X	X	X
Field Sparrow	*Spizella pusilla*	X	X		
Clay-colored Sparrow	*Spizella pallid*		X		X
Chipping Sparrow	*Spizella passerina*		X		
Grasshopper Sparrow	*Ammodramus savannarum*		X		
Savannah Sparrow	*Passerculus sandwichensis*	X	X		
Vesper Sparrow	*Pooecetes gramineus*	X	X		
Lark Bunting	*Calamospiza melanocorys*		X		
Lark Sparrow	*Chondestes grammacus*		X		
White-crowned Sparrow	*Zonotrichia leucophrys*	X	X		X
Song Sparrow	*Melospiza melodia*		X		
Lincoln's Sparrow	*Melospiza lincolnii*	X	X		
Western Meadowlark	*Sturnella neglecta*	X	X	X	X
Brown-headed Cowbird	*Molothrus ater*		X		
Audubon's Oriole	*Icterus graduacauda*	X	X		

Appendix II. List of observed summer species on each study area and year. BY1= burned, year 1 (2009), BY2 = burned, Year 2 (2010), UY1 = unburned, Year 1 (2009), UY2 = unburned, Year 2 (2010). 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA

Species	Scientific Name	BY1	BY2	UY1	UY2
Black-bellied Whistling Duck	*Dendrocygna autumnalis*	X		X	
Turkey Vulture	*Cathartes aura*	X		X	
Northern Bobwhite	*Colinus virginianus*	X	X	X	X
Mourning Dove	*Zenaida macroura*	X	X	X	X
Ground Dove	*Columbina passerina*	X	X	X	X
White-winged Dove	*Zenaida asiatica*				X
Yellow-billed Cuckoo	*Coccyzus americanus*		X	X	X
Greater Roadrunner	*Geococcyx californianus*	X	X	X	X
Groove-billed Ani	*Crotophaga sulcirostris*			X	
Lesser Nighthawk	*Chordeiles acutipennis*		X		
Buff-bellied Hummingbird	*Amazilia yucatanensis*			X	
Golden-fronted Woodpecker	*Melanerpes aurifrons*	X	X	X	X
Ladder-backed Woodpecker	*Picoides scalaris*	X	X	X	X
Vermillion Flycatcher	*Pyrocephalus rubinus*	X	X	X	
Scissor-tailed Flycatcher	*Tyrannus forficatus*	X	X	X	X
Ash-throated Flycatcher	*Myiarchus cinerascens*	X	X	X	X
Brown-crested Flycatcher	*Myiarchus tyrannulus*	X	X	X	X
Bell's Vireo	*Vireo bellii*	X	X	X	X
White-eyed Vireo	*Vireo griseus*		X	X	X
Green Jay	*Cyanocorax yncas*	X		X	
Black-crested Titmouse	*Baeolophus atricristatus*	X			
Verdin	*Auriparus flaviceps*	X	X	X	X
Bewick's Wren	*Thryomanes bewickii*	X	X	X	X
Cactus Wren	*Campylorhynchus brunneicapillus*	X	X	X	X
Black-tailed Gnatcatcher	*Polioptila melanura*	X	X	X	X

Appendix II. (Continued)

Species	Scientific Name	BY1	BY2	UY1	UY2
Blue-gray Gnatcatcher	*Polioptila caerulea*	X		X	
Northern Mockingbird	*Mimus polyglottos*	X	X	X	X
Curve-billed Thrasher	*Toxostoma curvirostre*	X	X	X	X
Long-billed Thrasher	*Toxostoma longirostre*	X			
Blue Grosbeak	*Passerina caerulea*	X		X	
Painted Bunting	*Passerina ciris*	X	X	X	X
Dicksissel	*Spiza americana*		X		
Northern Cardinal	*Cardinalis cardinalis*	X	X	X	X
Pyrrhuloxia	*Cardinalis sinuatus*	X	X	X	X
Black-throated Sparrow	*Amphispiza bilineata*	X	X	X	X
Lincoln's Sparrow	*Melospiza lincolnii*	X			
Olive Sparrow	*Arremonops rufivirgatus*	X	X	X	X
Lark Sparrow	*Chondestes grammacus*	X	X		
Cassin's Sparrow	*Aimophila cassinii*	X	X	X	X
Audubon's Oriole	*Icterus graduacauda*	X	X	X	X
Bullock's Oriole	*Icterus bullockii*	X	X		
Hooded Oriole	*Icterus cucullatus*	X			
Red-winged Blackbird	*Agelaius phoeniceus*	X		X	
Brown-headed Cowbird	*Molothrus ater*	X	X	X	X
Bronzed Cowbird	*Molothrus aeneus*	X		X	X

Appendix III. Means (above) and 95% confidence intervals (below) for percent (%) cover of woody vegetation species observed on burned and unburned sites during summer 2009. Species are listed in order of descending percent cover. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA

Burned		Unburned	
Species	Percent Cover (%)	Species	Percent Cover (%)
Hogplum (*Colubrina texensis*)	23.49 (18.77-28.96)	Leatherstem *(Jatropha dioica)*	13.68 (10.17-18.16)
Honey Mesquite (*Prosopis glandulosa)*	13.26 (9.69-17.88)	Hogplum (*Colubrina texensis*)	11.58 (8.36-15.82)
Prickly Pear (*Opuntia engelmannii)*	12.50 (9.04-17.04)	Prickly Pear (*Opuntia engelmannii)*	10.18 (7.18-14.23)
Huisache (*Acacia minuta)*	10.99 (7.76-15.33)	Honey Mesquite (*Prosopis glandulosa)*	8.77 (6.01-12.63)
Lantana (*Lantana urticoides)*	8.33 (5.57-12.29)	Lantana (*Lantana urticoides)*	8.07 (5.44-11.82)
Leatherstem *(Jatropha dioica)*	6.06 (3.76-9.62)	Texas Persimmon (*Diospyros texana)*	6.67 (4.31-10.18)
Palo Verde (*Parkinsonia texana)*	4.92 (2.90-8.24)	Brasil (*Condalia hookeri)*	5.61 (3.48-8.92)
Whitebrush (*Aloysia gratissima)*	3.79 (2.07-6.83)	Coyotillo (*Karwinskia humboldtiana*)	5.26 (3.22-8.50)
Brasil (*Condalia hookeri)*	3.41 (1.80-6.35)	Twisted Acacia (*Acacia schaffneri)*	5.26 (3.22-8.50)
Granjeno (*Celtis pallida)*	3.41 (1.80-6.35)	Blackbrush (*Acacia rigidula)*	3.51 (1.92-6.34)
Blackbrush (*Acacia rigidula)*	2.27 (1.04-4.87)	Tasajillo (*Opuntia leptocaulis*)	3.51 (1.92-6.34)
Narrowleaf Forestiera (*Forestiera angustifolia)*	2.27 (1.04-4.86)	Narrowleaf Forestiera (*Forestiera angustifolia)*	2.46 (0.19-4.98)
Twisted Acacia (*Acacia schaffneri)*	1.89 (0.81-4.36)	Texas Kidneywood (*Eysenhardtia texana*)	2.11 (0.97-4.52)
Texas Persimmon (*Diospyros texana)*	1.14 (0.39-.329)	Whitebrush (*Aloysia gratissima)*	2.11 (0.97-4.52)

Appendix III. (Continued)

Burned		Unburned	
Species	Percent Cover (%)	Species	Percent Cover (%)
Lime Prickly Ash *(Zanthoxylum fagara)*	0.76 (0.21-2.72)	Huisache (*Acacia minuta)*	1.75 (0.75-4.04)
Catclaw Acacia (*Acacia greggii*)	0.38 (0.07-2.11)	Palo Verde (*Parkinsonia texana)*	1.75 (0.75-4.04)
Knifeleaf Condalia (*Condalia spathulata*)	0.38 (0.07-2.11)	Granjeno (*Celtis pallida)*	1.40 (0.55-3.55)
Lotebush (*Ziziphus obtusifolia*)	0.38 (0.07-2.11)	Amargosa (*Castela erecta*)	1.05 (0.36-3.05)
Strawberry Cactus (*Mammillaria dioica*)	0.38 (0.07-2.11)	Lotebush (*Ziziphus obtusifolia*)	1.05 (0.36-3.05)
		Shrubby Blue Sage (*Salvia ballotiflora*)	1.05 (0.36-3.05)
		Wolfberry (*Lycium berlandieri*)	1.05 (0.36-3.05)
		Desert Yaupon (*Schaefferia cuneifolia*)	0.70 (0.19-2.52)
		Coma (*Sideroxylon celastrinum*)	0.35 (0.06-1.96)
		Guajillo (*Acacia berlandieri*)	0.35 (0.06-1.96)
		Guayacan (*Guajacum angustifolium*)	0.35 (0.06-1.96)

Appendix IV. Means (above) and 95% confidence intervals (below) for percent (%) cover of woody vegetation species observed on burned and unburned sites during summer 2010. Species are listed in order of descending percent cover. 2009 – 2010. Chaparral WMA and Piloncillo Ranch, La Salle and Dimmit counties, Texas, USA

Burned		Unburned	
Species	Percent Cover (%)	Species	Percent Cover (%)
Hogplum (*Colubrina texensis*)	23.49 (18.77-28.96)	Leatherstem *(Jatropha dioica)*	15.28 (11.58-19.89)
Honey Mesquite (*Prosopis glandulosa)*	13.26 (9.69-17.88)	Prickly Pear (*Opuntia engelmannii)*	12.85 (9.47-17.21)
Prickly Pear (*Opuntia engelmannii)*	12.50 (9.04-17.03)	Hogplum (*Colubrina texensis*)	1.11 (7.98-15.26)
Huisache (*Acacia minuta)*	10.99 (7.78-15.33)	Honey Mesquite (*Prosopis glandulosa)*	10.42 (7.39-14.48)
Lantana (*Lantana urticoides)*	8.33 (5.57-12.29)	Lantana (*Lantana urticoides)*	7.29 (4.82-10.89)
Leatherstem *(Jatropha dioica)*	6.06 (3.76-9.62)	Brasil (*Condalia hookeri)*	6.94 (4.54-10.48)
Palo Verde (*Parkinsonia texana)*	4.92 (2.90-8.24)	Texas Persimmon (*Diospyros texana)*	5.56 (3.45-8.83)
Whitebrush (*Aloysia gratissima)*	3.79 (2.07-6.83)	Palo Verde (*Parkinsonia texana)*	4.86 (2.92-7.99)
Brasil (*Condalia hookeri)*	3.41 (1.80-6.35)	Tasajillo (*Opuntia leptocaulis*)	3.82 (2.14-6.71)
Granjeno (*Celtis pallida)*	3.41 (1.80-6.35)	Blackbrush *(Acacia rigidula)*	3.13 (1.65-5.83)
Blackbrush *(Acacia rigidula)*	2.27 (1.05-4.87)	Coyotillo (*Karwinskia humboldtiana*)	3.13 (1.65-5.83)
Narrowleaf Forestiera (*Forestiera angustifolia)*	2.27 (1.05-4.87)	Twisted Acacia (*Acacia schaffneri)*	3.13 (1.65-5.83)
Twisted Acacia (*Acacia schaffneri)*	1.89 (0.81-4.36)	Narrowleaf Forestiera (*Forestiera angustifolia)*	2.43 (1.18-4.93)
Texas Persimmon (*Diospyros texana)*	1.14 (0.39-3.29)	Granjeno (*Celtis pallida)*	2.08 (0.96-4.47)

Appendix IV. (Continued)

Burned		Unburned	
Species	Percent Cover (%)	Species	Percent Cover (%)
Lime Prickly Ash *(Zanthoxylum fagara)*	0.76 (0.21-2.72)	Whitebrush (*Aloysia gratissima)*	1.39 (0.54-3.52)
Catclaw Acacia (*Acacia greggii*)	0.38 (0.07-2.11)	Coma (*Sideroxylon celastrinum*)	1.04 (0.35-3.02)
Knifeleaf Condalia (*Condalia spathulata*)	0.38 (0.07-2.11)	Shrubby Blue Sage (*Salvia ballotiflora*)	1.04 (0.35-3.02)
Lotebush (*Ziziphus obtusifolia*)	0.38 (0.07-2.11)	Texas Kidneywood (*Eysenhardtia texana*)	1.04 (0.35-3.02)
Strawberry Cactus (*Mammillaria dioica*)	0.38 (0.07-2.11)	Huisache (*Acacia minuta)*	0.69 (0.19-2.50)
		Lime Prickly Ash *(Zanthoxylum fagara)*	0.69 (0.19-2.50)
		Wolfberry (*Lycium berlandieri*)	0.69 (0.19-2.50)
		Desert Yaupon (*Schaefferia cuneifolia*)	0.35 (0.06-1.94)
		Amargosa (*Castela erecta*)	0.35 (0.06-1.94)
		Knifeleaf Condalia (*Condalia spathulata*)	0.35 (0.06-1.94)
		Lotebush (*Ziziphus obtusifolia*)	0.35 (0.06-1.94)

VITA

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PARENTS:	Pamela & Allistair de Verteuil
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EDUCATION:	Master of Science degree, December 2011 Texas A&M University – Kingsville Kingsville, Texas Major: Range & Wildlife Management Bachelor of Science degree, December 2004 Florida State University Tallahassee, Florida Major: Environmental Studies

PROFESSIONAL EXPERIENCE:

GRADUATE RESEARCH: January 2009 – December 2011
Thesis Title: Impacts of Wildfire on Avian Communities of South Texas

WILDLIFE BIOLOGIST: January 2006 – December 2008
Employer: Florida Fish & Wildlife Conservation Commission

Responsible for habitat management, population monitoring of Red-cockaded Woodpeckers in Citrus Wildlife Management Area.

BIOLOGICAL TECHNICIAN: March 2005 – August 2005
Employer: United States Department of Agriculture; Department of Biological Control

Monitor and control spread of invasive cactus moth (*Cactoblastis cactorum*) throughout Gulf coast of Florida & Alabama.

GIS TECHNICIAN: August 2004 – November 2004
Employer: Florida Fish & Wildlife Conservation Commission

Process GIS data requests from public and produce maps for endangered species location and monitoring

INTERNSHIPS:
The Nature Conservancy – Apalachicola Bluffs & Ravines Preserve: April – July 2004
Department of Environmental Protection – Tallahassee, FL: February – May 2003