THE LONG-TERM EFFECTS OF FIRE SEASONALITY ON A ROLLING PLAINS SMALL VERTEBRATE AND VEGETATION COMMUNITY

by

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ABSTRACT

Fire is a natural process in grasslands, which makes prescribed burning an extremely important management tool that is widely used to replicate that natural process that is often missing from modern grasslands. However, most research addressing the impacts of prescribed fire have focused on short-term responses and studies examining the seasonality of fire are largely lacking. Prescribed burn studies on nongame wildlife are also generally lacking. Therefore filling in this data gap with longer-term studies will help us better-manage these species that are potentially in trouble and being ignored. In 2004, a project was initiated to evaluate the impacts of summer and winter burning on vegetation and small vertebrates in a sand sage prairie ecosystem within the southeastern Texas Panhandle. Although the initial project was designed to look at the short-term effects, I initiated a project to continue data collection to begin evaluating the longer-term effects on vegetation and small vertebrates. A randomized complete block design with 5 blocks and 3 treatments was established at the Matador Wildlife Management Area (WMA) in Cottle County, Texas. Each plot within a block was randomly assigned 1 of 3 treatments: winter burned, summer burned or unburned. Herpetofauna and small mammals were sampled using drift fence arrays with pitfall traps during the spring and summer of all years of sampling. Herbaceous vegetation cover and frequency was

measured twice annually using quadrats and woody vegetation was measured during late summer using the line-intercept method. I regressed sample year against community metrics to evaluate longer-term responses to burning. In the early season, summerburning increased species richness, evenness, and diversity of vertebrates over time, but individual species responses were variable. In contrast, treatment did not generally alter community metrics for vegetation. However, comparison of early season indices of similarity suggested that burning shifted the species composition of the herbaceous plant community over time, whereas during the late season, the small vertebrate community shifted in the unburned treatment, but not in either burn treatment. Thus, burning is shifting the plant community regardless of weather patterns, but it may be ameliorating the effects of long-term drought on the small vertebrate community. These trends were weak and interpretation has been made difficult because of the long-term severe drought that has persisted in the region for the duration of this project. However, my results generally suggest summer burning seems to have a greater positive benefit on small vertebrate communities in this system than winter-burning, and both burning treatments appear to improve conditions relative to lack of burning. Longer-term research on the effects of growing and dormant season burning on small vertebrate and vegetation communities, especially during wet and drought years, is needed to evaluate the effects of seasonal prescribed burns in the Rolling Plains of Texas.

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CHAPTER I

LONG- TERM EFFECTS OF FIRE SEASONALITY ON VEGETATION IN A ROLLING PLAINS ECOSYSTEM

INTRODUCTION

Fire plays a key role in developing and maintaining many ecosystems. Therefore, fire is a natural component within prairie ecosystems. In general, fire has largely been recognized as a significant constituent of the original disturbance regimes (Wright and Bailey 1982) and it serves as a crucial role in grassland ecology. Following climate, fire is one of the single most influential factors that determine the vigor and sustainability of grasslands (Axelrod 1985). Throughout the range of North American grasslands, summer fire was a common phenomenon caused by both anthropogenic ignitions and lightning and vast savannahs and prairies covered millions of hectares and were burned on a regular basis (Bragg 1982, Higgins 1984, Pyne 1994). Fire naturally occurred in the Rolling Plains of Texas prior to European settlement with a highly variable frequency of 5 to 10 years (Wright and Bailey 1982).

However, the role that fire plays along with other influences which contributed to maintaining these grasslands, such as grazing, droughts, topography, and soil condition, were not constant or consistent in time and space (Anderson 1990). Decades of widespread fire suppression coupled with overgrazing has led to an unnatural fuel accumulation, woody species encroachment, and altered natural fire regimes (Chang

1996, Keane et al. 2002). The lack of burning in much of today's landscapes removes one of the key natural processes from fire-dependent ecosystems. Therefore, there is a need to re-introduce fire back into grasslands as a management tool to restore native ecosystem processes and functioning. Today, prescribed burning is a common tool used to rehabilitate prairie habitat, reduce unwanted vegetation, increase nutrient cycling, and accomplish ecosystem restoration (McNaughton 1979). Rangeland managers are beginning to employ the use of prescribed fire in order to maintain woody plant diversity and increase herbaceous vegetation preferred by both wildlife and livestock, reduce brush cover, and enhance wildlife habitat (Box and White 1969, Hansmire et al. 1988, Ruthven et al. 2000, Ruthven et al. 2002).

Grasslands occur where dry seasons and or occasional droughts are prevalent and in areas with smooth to steeply rolling topography, which allows fires to occur and carry (Sauer 1950). Therefore, grasses are better adapted to drought than trees. The adaptation that allowed grasses the ability to survive through drought periods was the ability to thrive in the underground organs while being top-killed. This mechanism not only ensured surviving throughout the drought periods, but also provided protection from periodic fires (Gleason 1922). Grassland productivity is a function that can be measured by time since burning. Over time, productivity declines as dead matter and litter begins to accumulate (Knapp and Seastedt 1986). Although these vegetative components contribute to and are important for vertical structure, which has capacity to enhance habitat quality, there is a reduction in resource availability, which can in turn reduce habitat quality if there is an absence of fire (Vinton and Collins 1989).

An inherent characteristic of fires in grassland ecosystems is the season in which they may occur (Anderson et al. 1970, Henderson 1981). However, studies on the seasonality of fire are often short-term, less evident and difficult to interpret, and primarily focus on spring burning. Spring prescribed burns are more common and preferred for livestock management with an intention to enhance forage quality for cattle (Engle and Bidwell 2001). Many studies have demonstrated the versatility of fire to stimulate forage production in grasslands that have remained stagnant over time (Weaver and Tomanek 1951, Weaver and Albertson 1956, Ehrenreich 1959, Hadley and Kieckhefer 1963, Vogl 1965, Wright 1972a, Anderson et al. 1970). Therefore, burning conducted during the freeze-free period of spring has been recommended for optimal cattle production (Launchbaugh and Owensby 1978). On the other hand, burning conducted in the early spring or winter has been preferred to enhance wildlife habitat (Riggs et al. 1996). Approximately two-thirds of The Rolling Plains Ecoregion of Texas is primarily managed for cattle production, therefore, spring prescribed burns are the more prominent management approach for this region (Engle et al. 2000).

Golley and Golley (1972) indicated that the productivity of grassland ecosystems decline if accumulated herbage is not removed by a method mimicking the original disturbance regimes (McNaughton 1979, Risser et al. 1981). Grasslands subjected to fire suppression can be effectively converted to forests or shrublands and fire has been an adequate management tool in suppressing or eradicating woody vegetation and favoring more open grasslands. Fuel accumulation is largely because of fire suppression, which in turn degrades ecosystem integrity and increases the risk of wildfires. In the Great Plains, excess litter accumulation will suppress grass and seed yields during productive years (Weaver and Rowland 1952, Old 1969). Build up of accumulated herbaceous litter in excess of 2,241 kg/ha ties up nutrients, slows the process of the nitrogen cycle, and lowers soil temperatures which then retards bacterial activity, particularly during wet years (Wright 1974). Fire also has potential to contribute economical means of maintaining traditional methods of chemical and mechanical brush treatments to control woody species, which are often costly (Scifres and Hamilton 1993). Increased soil temperatures after a fire will also increase forage yields and enhance nitrification of organic matter within grasslands as compared to unburned areas (Sharrow and Wright 1977).

The Rolling Plains are covered by mesquite-grasslands of a savannah nature and encompass the southwestern portion of Oklahoma and the northcentral range of Texas (Hamilton 1962). The Rolling Plains meet the High Plains along the edge of the Caprock Escarpment and occur within the transition zone of the mixed grass and short grass prairies of the Great Plains. Although it is considered to be primarily composed of honey mesquite (*Prosopis glandulosa*) and grama (*Bouteloua* spp.) grasses, the region displays a mosaic of plant communities, comprised of mixed-grass savannahs and riparian corridors. This region is described as having rolling prairies with secluded mesas with small canyons and is characterized by clay soils. Following European settlement, fires were suppressed out of public concern and philosophy and overgrazing occurred. Overgrazing by cattle weakened native plant communities, removed fuel loads needed for a fire to occur, and led to encroachment of invasive woody species (Wright 1974). These factors, coupled with variable and persisting droughts within the region accelerated the process.

Although voluminous literature exists on the effects of fire, studies evaluating the effects of fire in this region are largely lacking. And such studies are often short-term. Therefore, this project was initiated to explore the longer-term effects of growing and dormant season burning on herbaceous and woody vegetation in the Rolling Plains of Texas. As such, my objective was to determine the effects of summer and winter prescribed fire on herbaceous and woody vegetation communities within the Rolling Plains of Texas.

STUDY AREA

Matador Wildlife Management Area

My study site lies within the 11,405-ha Matador Wildlife Management Area (WMA) about 10 km north of the town of Paducah in Cottle County, Texas (Figure I.1). The property was purchased by the state of Texas in 1959 with Pittman-Robertson funds and is managed as a research and demonstration area by the Wildlife Division of Texas Parks and Wildlife Department. The area is comprised of grass savannahs and riparian corridors along the Pease River drainage. The area's climate is identified as a subtropical climate with having dry winters, and relatively hot, humid summers with an average annual precipitation of 54 cm. The rainy season primarily occurs from May to June and the growing season averages 219 days (Richardson et al. 1974). The area is susceptible to periods of drought with one year out of every 10 receiving less than 33 cm of rain.

Conversely, annual rainfall exceeds 76 cm one year out of every 10. Little benefit is acquired during these extremely wet years because heavy downpours result in an excessive runoff (Richardson et al. 1974). The terrain is rough in the west and levels off in the east with elevations ranging from 488 and 640 m above sea level. Topography is characterized by rolling plains to steep slopes and canyons (Richardson et al. 1974).

The dominant woody vegetation on Matador WMA includes sand sagebrush (*Artemisia filifolia*), honey mesquite, eastern cottonwood (*Populus deltoides*), salt-cedar (*Tamarix* sp.), western soapberry (*Sapindus saponaria*), netleaf hackberry (*Celtis reticulata*), and redberry juniper (*Juniperus pinchotii*). The primary grasses found on the area include sideoats grama (*Bouteloua curtipendula*), sand dropseed (*Sporobolus cryptandrus*), purple three-awn (*Aristida purpurea*), little bluestem (*Schizachyrium scoparium*), and blue grama (*Bouteloua gracilis*). Common forbs found include western ragweed (*Ambrosia psilostachya*), lamb's quarters (*Chenopodium album*), marestail (*Erigeron canadensis*), and plaintain (*Plantago* spp; Hodge 2000, Hall 2005)

METHODS

Burning and Experimental Design

My study site consisted of 273-ha of sand sagebrush grasslands in Headquarters Pasture. In 2004, Headquarters pasture was divided into a 3 x 5 randomized complete block design; each block containing 3 18-ha plots, resulting in 15 total plots (Figure I.2). Each plot within a block was randomly assigned 1 of 3 treatments: winter-burned, summer-burned, or unburned. The burning regime for the pasture has been maintained by the Texas Parks and Wildlife Department and each burned plot has been treated 3-4 times since the initial project. Summer burns are conducted between July and August and winter burns are conducted between December and February largely depending on manpower, weather conditions, and fuel loads available.

The decision was made beginning in 2007/2008 that continuous cattle grazing was to be initiated on the study site. Grazing was not restricted to a plot-by-plot basis. Therefore, the availability of grazing was equal across all study plots. The pasture was stocked at 16 animal units (cow/calf).

The project was started in 2004, and data for herbaceous and woody vegetation was collected over the years of 2005 - 2009. Assigned burning regimes continued even after data collection ended. In 2018 and 2019, I replicated their methods and the methods of data collection are consistent with the ones I describe below.

Herbaceous Vegetation Sampling

Community – Post treatment herbaceous vegetation community metrics were measured in each study plot using Daubenmire frames during each year of sampling: 2005, 2007 – 2009 and 2018-2019. I estimated percent cover for bare ground, litter, grasses, and forbs using 100 randomly selected 0.5 x 0.2 m quadrat frames (Daubenmire and Daubenmire 1968) during the early season (May-June) and late season (July-August) of each year. The randomly selected locations for each frame were determined by using a dual analog/digital time watch and starting at the center of each plot. The seconds displayed on the digital time was used to determine the number of paces to be taken in the direction of the analog seconds hand. The seconds hand was used to determine the direction to pace from each data point collected. After reaching the location determined by the direction and number of paces, I blindly tossed the quadrat frame over my shoulder and recorded the cover and frequency data. Herbaceous vegetation frequency was determined by the presence of species rooted within the quadrat frame summed over the 100 frames for each plot. Canopy cover was estimated by the percent forbs, grasses, litter, and bare ground in the quadrat frame averaged across the 100 frames per plot.

Metrics – Herbaceous vegetation species diversity was calculated using Shannonweiner (H') and modified Simpsons's (Mod D) diversity indices. Shannon-weiner diversity was calculated as follows:

$$H' = -\Sigma pi(lnpi)$$

Modified Simpson's diversity was calculated as follows:

Mod
$$D = 1 - \Sigma p i^2$$

Where pi = proportion of the ith species in the sample (Pielou 1975, Magurran 1988). Herbaceous vegetation evenness was calculated as follows:

$$Evenness = \frac{H'}{H'_{max}}$$

Where H' = Shannon-Weiner diversity and H' $_{max} = \ln(S)$, where S = species richness (Magurran 1988).

Metrics for the various herbaceous vegetation covers, frequency, diversity, and evenness were then regressed using a linear regression against sample year to evaluate the longer-term responses to the different burning regimes over this time period. Analyses were conducted for early and late season data separately to avoid the potentially confounding influence of sampling season on treatment effects. For herbaceous vegetation, I also calculated a Jaccard's indices of similarity to compare plant species documented in each year to the species composition I detected in 2019. Jaccard's index was calculated as follows:

$$J = j/(a+b-j)$$

Where a = number of species captured in year 1, b = number of species captured in year 2, and j = number of species captured in both years. I regressed Jaccard's index of similarity to 2019 against each year to look for changes in species composition over time (Magurran 1988). Because analyses were exploratory, I chose $\alpha = 0.1$ to explore trends.

Frequencies for the 6 (4 grasses, 2 forbs) most abundant species present were also regressed over time: blue grama, fringed signal grass (*Brachiaria ciliatissima*), sand dropseed, sideoats grama, western ragweed, and erect dayflower (*Commelina erecta*). Analyses were conducted for early and late season data separately to avoid the potentially confounding influence of season on treatment effect. Because analyses are exploratory, I set $\alpha = 0.1$ for all comparisons.

Woody Vegetation Sampling

Pre-treatment estimates of percent woody canopy cover were measured in 2005. Post treatment estimates of percent woody canopy cover were measured in each study plot using the line-intercept method during each year of sampling: 2005-2009, 2011, and 2018-2019 (Chambers and Brown 1983). In each plot, 3 evenly distributed 200 m north – south running transects were established and parallel to each other. Each of these main north – south running transects had 12 30-m perpendicularly placed transects originating off the main transect. Each of these 30-m transects were evenly spaced every 20-m apart (Figure I.3). Woody plant cover was recorded as the cm of woody stems intersecting the 30-m transects. Canopy cover was determined by taking the cover of each species on each of the 12 perpendicular transects across the 3 main transects in each plot and averaging the totals across the 3 north and south running transects in each plot to estimate individual plot canopy cover.

Average woody canopy cover per plot was regressed against sampling year to evaluate the longer-term responses to the different burning regimes over this time period. Because analyses were exploratory, I set α =0.1 for all comparisons.

RESULTS

Herbaceous Vegetation Sampling

Community – In the early season of sampling, percent bare ground increased in the summer-burned plots ($F_7 = 24.130$, P = 0.007, $r^2 = 0.858$) and the unburned plots ($F_7 = 5.869$, P = 0.072, $r^2 = 0.595$) as compared to the winter-burned plots ($F_7 = 1.337$, P = 0.311, $r^2 = 0.250$, Table I.1, Figure I.4). Grass cover decreased in the summer-burned plots ($F_7 = 8.911$, P = 0.041, $r^2 = 0.690$) as compared to the winter-burned plots ($F_7 = 2.329$, P = 0.202, $r^2 = 0.368$) and unburned plots ($F_7 = 4.487$, P = 0.101, $r^2 = 0.529$), which was approaching a significant decrease in cover over time (Table I.2, Figure I.5). Litter cover did not vary among the summer-burned ($F_7 = 0.518$, P = 0.512, $r^2 = 0.115$), winter-burned ($F_7 = 0.723$, P = 0.443, $r^2 = 0.153$), and unburned ($F_7 = 0.605$, P = 0.480, $r^2 = 0.131$) treatments (Table I.3). Forb cover also did not vary among treatments (Table I.4). However, forbs were approaching a significant decrease in cover over time in the summer-burned (F₇ = 4.261, P = 0.108, $r^2 = 0.516$) and winter-burned plots (F₇ = 4.490, P = 0.101, $r^2 = 0.529$) as compared to the unburned plots (F₇ = 3.468, P = 0.136, $r^2 = 0.464$, Figure I.6).

In the late season of sampling, percent bare ground did not vary among summerburned ($F_7 = 0.129$, P = 0.738, $r^2 = 0.031$), winter-burned ($F_7 = 0.266$, P = 0.633, $r^2 = 0.062$), and unburned ($F_7 = 2.569$, P = 0.184, $r^2 = 0.391$) treatments (Table I.1). Litter cover also did not vary among summer-burned ($F_7 = 0.665$, P = 0.460, $r^2 = 0.143$), winter-burned ($F_7 = 0.294$, P = 0.617, $r^2 = 0.068$), and unburned ($F_7 = 0.817$, P = 0.417, $r^2 = 0.170$) treatments (Table I.3). Grass cover did not vary among summer-burned ($F_7 = 0.686$, P = 0.454, $r^2 = 0.146$), winter-burned ($F_7 = 3.199$, P = 0.148, $r^2 = 0.444$), and unburned ($F_7 = 3.932$, P = 0.118, $r^2 = 0.496$) treatments (Table I.2). Forb cover did not vary among summer-burned ($F_7 = 0.977$, P = 0.379, $r^2 = 0.196$), winter-burned ($F_7 = 1.647$ P = 0.269, $r^2 = 0.292$), and unburned ($F_7 = 0.593$, P = 0.484, $r^2 = 0.129$) treatments (Table I.4).

In the early season of sampling, species richness for herbaceous vegetation did not vary among summer-burned ($F_7 = 0.642$, P = 0.459, $r^2 = 0.114$), winter-burned ($F_7 = 0.293$, P = 0.611, $r^2 = 0.055$), and unburned ($F_7 = 1.480$, P = 0.278, $r^2 = 0.228$) treatments. Both modified Simpson's (Mod D, summer: $F_7 = 0.855$, P = 0.398, $r^2 = 0.146$; winter: $F_7 = 0.007$, P = 0.934, $r^2 = 0.002$; unburned: $F_7 = 0.503$, P = 0.510, $r^2 = 0.091$) and Shannon-Weiner (H', summer: $F_7 = 0.541$, P = 0.495, $r^2 = 0.098$; winter: $F_7 = 0.073$, P = 0.798, $r^2 = 0.014$; unburned: $F_7 = 0.541$, P = 0.495, $r^2 = 0.098$) diversity indices did not vary among treatments. Evenness did not vary among summer-burned ($F_7 = 0.472$, P = 0.523, r^2 = 0.086), winter-burned (F₇ = 0.003, P = 0.957, r^2 = 0.001), and unburned (F₇ = 0.017, P = 0.901, r^2 = 0.003) treatments from herbaceous vegetation results (Table I.5). However, Jaccard's index of similarity demonstrated that in the early season, species composition of the plant community did shift overtime, regardless of treatment (summer: F₇ = 9.796, P = 0.035, r^2 = 0.710; Winter: F₇ = 4.511, P = 0.101, r^2 = 0.530) as compared to the unburned treatment, which showed no trend in shifting the species composition of the plant community over time (F₇ = 0.417, P = 0.554, r^2 = 0.094, Table I.6, Figure I.7).

In the late season of sampling, species richness for herbaceous vegetation did not vary among summer-burned ($F_7 = 0.001$, P = 0.977, $r^2 < 0.001$), winter-burned ($F_7 = 0.052$, P = 0.828, $r^2 = 0.010$), and unburned ($F_7 = 0.004$, P = 0.953, $r^2 = 0.001$) treatments. Modified Simpson's (Mod D, summer: $F_7 = 0.376$, P = 0.567, $r^2 = 0.070$; winter: $F_7 = 0.297$, P = 0.609, $r^2 = 0.056$; unburned: $F_7 = 0.355$, P = 0.577, $r^2 = 0.066$) and Shannon-Weiner (H', summer: $F_7 = 0.017$, P = 0.901, $r^2 = 0.003$; winter: $F_7 = 0.341$, P = 0.585, $r^2 = 0.064$; unburned: $F_7 = 0.237$, P = 0.647, $r^2 = 0.045$) diversity indices did not vary among treatments. Evenness did not vary among summer-burned ($F_7 = 0.112$, P = 0.752, $r^2 = 0.022$), winter-burned ($F_7 = 1.517$, P = 0.273, $r^2 = 0.233$), and unburned ($F_7 = 2.187$, P = 0.199, $r^2 = 0.304$) treatments from herbaceous vegetation results (Table I.7). Jaacard's index of similarity to 2019 indicated no significant change in species composition over time in the late season of sampling for summer-burned ($F_7 = 2.104$, P = 0.221, $r^2 = 0.345$), winter-burned ($F_7 = 3.296$, P = 0.144, $r^2 = 0.452$), and unburned ($F_7 = 0.087$, P = 0.783, $r^2 = 0.021$) treatments (Table I.6).

Frequency—In the early season, there were no differences among summer-burned ($F_7 = 1.209$, P = 0.322, $r^2 = 0.195$), winter-burned ($F_7 = 0.098$, P = 0.767, $r^2 = 0.019$), and unburned ($F_7 = 0.817$, P = 0.407, $r^2 = 0.141$) treatments for frequencies of blue grama (Table I.9). There were no differences among summer-burned ($F_7 = 0.305$, P = 0.605, $r^2 = 0.057$), winter-burned ($F_7 < 0.001$, P = 0.986, $r^2 < 0.001$), and unburned ($F_7 = 0.409$, P = 0.550, $r^2 = 0.076$) treatments for frequencies of fringed signal grass (Table I.10), and there were no differences among summer-burned ($F_7 = 0.799$, P = 0.413, $r^2 = 0.138$), winter-burned ($F_7 = 0.001$, P = 0.973, $r^2 < 0.001$), and unburned ($F_7 = 0.241$, P = 0.644, $r^2 = 0.046$) treatments for frequencies of sand dropseed (Table I.11). Frequency of sideoats grama was significantly higher in the summer-burned treatment ($F_7 = 12.44$, P = 0.017, $r^2 = 0.713$) and the winter-burned treatment ($F_7 = 12.19$, P = 0.017, $r^2 = 0.709$) as compared to the unburned treatment ($F_7 = 1.63$, P = 0.257, $r^2 = 0.246$, Table I.8, Figure I.8).

In the late season, there were no differences among summer-burned ($F_7 = 1.111$, P = 0.340, $r^2 = 0.182$), winter-burned ($F_7 = 2.746$, P = 0.158, $r^2 = 0.354$), and unburned ($F_7 = 0.983$, P = 0.367, $r^2 = 0.164$) treatments for frequency of sideoats grama (Table I.8). Frequency of blue grama significantly increased in the summer-burned treatment ($F_7 = 11.41$, P = 0.019, $r^2 = 0.695$) and the unburned treatment ($F_7 = 7.21$, P = 0.043, $r^2 = 0.591$) as compared to the winter-burned treatment ($F_7 = 1.06$, P = 0.350, $r^2 = 0.175$, Table I.9, Figure I.9). Frequency of fringed signal grass significantly increased in the summer-burned to the winter-burned treatment ($F_7 = 0.574$) as compared to the winter-burned treatment ($F_7 = 0.574$) as compared to the winter-burned treatment ($F_7 = 0.234$, P = 0.649, $r^2 = 0.045$) and the unburned treatment ($F_7 = 2.98$, P = 0.145, $r^2 = 0.373$, Table I.10, Figure I.10). Frequency of sand dropseed significantly
increased in the unburned treatment ($F_7 = 6.13$, P = 0.056, $r^2 = 0.551$) as compared to the summer-burned ($F_7 = 0.247$, P = 0.640, $r^2 = 0.047$) and winter-burned ($F_7 = 2.01$, P = 0.216, $r^2 = 0.287$) treatments (Table I.11, Figure I.11).

In the early season, there were no differences in frequencies of western ragweed among summer-burned ($F_7 = 0.240$, P = 0.645, $r^2 = 0.046$), winter-burned ($F_7 = 0.482$, P = 0.519, $r^2 = 0.088$), and unburned ($F_7 = 0.007$, P = 0.939, $r^2 = 0.001$) treatments (Table I.12). There were also no differences in frequencies of erect dayflower among summerburned ($F_7 = 2.158$, P = 0.202, $r^2 = 0.301$), winter-burned ($F_7 = 2.233$, P = 0.195, $r^2 = 0.309$), and unburned ($F_7 = 0.010$, P = 0.925, $r^2 = 0.002$) treatments (Table I.13). However, frequency of erect dayflower was approaching significance in the winterburned treatment ($F_7 = 2.23$, P = 0.195, $r^2 = 0.309$) as compared to the summer-burned ($F_7 = 2.16$, P = 0.202, $r^2 = 0.301$) and unburned ($F_7 = 0.01$, P = 0.925, $r^2 = 0.002$) treatments (Figure I.12).

In the late season, there were no differences in frequencies of western ragweed among summer-burned ($F_7 = 0.473$, P = 0.522, $r^2 = 0.086$), winter-burned ($F_7 = 0.058$, P = 0.819, $r^2 = 0.012$), and unburned ($F_7 = 0.104$, P = 0.760, $r^2 = 0.020$) treatments (Table I.12). There were no differences in frequencies of erect dayflower among summer-burned ($F_7 = 1.016$, P = 0.360, $r^2 = 0.169$), winter-burned ($F_7 = 0.348$, P = 0.581, $r^2 = 0.065$), and unburned ($F_7 = 0.011$, P = 0.921, $r^2 = 0.002$) treatments (Table I.13).

Woody Vegetation Sampling

Average percent cover of all woody plants in summer-burned ($F_7 = 0.467$, P = 0.519, $r^2 = 0.072$) and winter-burned ($F_7 = 2.90$, P = 0.139, $r^2 = 0.326$) plots did not vary

among treatments (Table I.14). However, percent canopy cover of woody plants decreased in the unburned plots over time ($F_7 = 5.89$, P = 0.051, $r^2 = 0.495$, Table I.14, Figure I.12). I detected no differences in average diameter of the canopies of all woody clumps in summer burned ($F_7 = 2.54$, P = 0.162, $r^2 = 0.297$) and winter-burned ($F_7 =$ 0.124, P = 0.736, $r^2 = 0.020$) plots (Table I.14). However, in the unburned plots ($F_7 =$ 5.413, P = 0.058, $r^2 = 0.474$), average diameter of canopies all woody plant clumps increased over time (Table I.14, Figure I.14). I detected that the average number of all woody plant clumps per intercept in the summer-burned ($F_7 = 3.722$, P = 0.102, $r^2 =$ 0.383) treatments was approaching significance (Table I.14, Figure I.14). However, average number of woody plant clumps per intercept decreased in both winter-burned ($F_7 =$ 4.420, P = 0.080, $r^2 = 0.424$) and unburned ($F_7 = 6.943$, P = 0.039, $r^2 = 0.536$) treatments (Table I.14, Figure I.14)

No trends for percent cover of sandsage brush were detected in summer-burned treatments ($F_7 = 1.387$, P = 0.283, $r^2 = 0.188$, Table I.15). However, canopy cover of sandsage brush was significantly reduced in winter-burned ($F_7 = 4.189$, P = 0.087, $r^2 = 0.411$) and unburned ($F_7 = 13.643$, P = 0.010, $r^2 = 0.695$) treatments (Table I.15, Figure I.15). No trends were detected for average diameter of canopies of sandsage clumps for summer-burned ($F_7 = 0.489$, P = 0.511, $r^2 = 0.075$), winter-burned ($F_7 = 0.141$, P = 0.720, $r^2 = 0.023$), and unburned ($F_7 = 0.003$, P = 0.955, $r^2 = 0.001$) treatments (Table I.15). Average number of sandsage clumps per intercept was significantly reduced in the summer-burned ($F_7 = 3.863$, P = 0.097, $r^2 = 0.392$), winter-burned ($F_7 = 4.769$, P = 0.072,

 $r^2 = 0.443$), and unburned (F₇ = 7.169, P = 0.037, $r^2 = 0.544$) treatments (Table I.15, Figure I.16)

I detected no significant trends for average percent cover of mesquite in summerburned ($F_7 = 0.031$, P = 0.866, $r^2 = 0.005$), winter-burned ($F_7 < 0.001$, P = 0.991, $r^2 < 0.001$), and unburned ($F_7 = 0.700$, P = 0.435, $r^2 = 0.105$) treatments (Table I.16). I detected no significant trends for average diameter of mesquite clumps in summer-burned ($F_7 = 0.560$, P = 0.482, $r^2 = 0.085$), winter-burned ($F_7 = 0.827$, P = 0.398, $r^2 = 0.121$), and unburned ($F_7 = 0.336$, P = 0.583, $r^2 = 0.053$) treatments (Table I.16). Average number of mesquite clumps per intercept was significantly increased in summer-burned ($F_7 =$ 42.331, P < 0.001, $r^2 = 0.876$), winter-burned ($F_7 = 13.148$, P = 0.011, $r^2 = 0.687$), and unburned ($F_7 = 96.629$, P < 0.001, $r^2 = 0.942$) treatments (Table I.16, Figure I.17).

DISCUSSION

Herbaceous Vegetation

Fire is used to alter canopy cover of both herbaceous and woody vegetation, reduce accumulated litter, increase nutrient cycling essential for plant growth, and increase radiation reaching the surface (Mushinsky and Gibson 1991). Intensity, timing, and frequency of fire occurrence vary independently of each other and affect the environment in numerous ways. Fires reduce litter and biomass, therefore altering nutrient, energy, and water fluxes. These changes can affect the long-term productivity of the system (Frost and Robertson 1987). It is suggested that late season fires promote development and growth of perennial grasses while early season fires reduces perennials and promotes annuals (Afolayan 1978). However my results suggested that both burning treatments in the Rolling Plains of Texas were beneficial at shifting the herbaceous plant community and maintaining woody vegetation over time in the presence of a prolonged drought.

My results indicated that treatment did not generally alter community metrics (richness, evenness, diversity) for herbaceous vegetation. However, comparison of early season indices of similarity demonstrated that the presence of burning shifted the species composition of the herbaceous plant community over time. Therefore, burning is shifting the plant community regardless of weather patterns, but it may be ameliorating the effects of long-term drought on other communities. This may be because climate, including droughts, seasonal dryness, and winds, allows fires to burn extensively and helps to suppress woody vegetation that might have otherwise developed in unwanted areas, thus creating less competition for herbaceous plant species to acquire required nutrients for growth (Stubbendieck et al. 2007). It is also suggested that prescribed fire may increase resistance to drought (van Mantgem et al. 2016). Reduced litter density following a prescribed fire also reduces competition for resources amongst the remaining vegetation, so that these remaining species are more resistant to additional stressors, such as drought. My results also support the results of other studies that suggest that prescribed fire is effective at removing accumulated herbaceous litter because of the significant increase in surface area of bare ground in order to promote new growth and allow for the ecological succession of these landscapes (Sharrow 1975, Pase and Knipe 1977, Sharrow and Wright 1977, Dunwiddie 1991).

In my case, prescribed summer burning was effective at reducing grass cover, which was similar to the results of other studies that suggested that perennial grass species tended to decrease in the presence of a summer-burned treatment (Scifres and Duncan 1982, Ruthven and Synatzske 2002). In a similar study conducted in an Oklahoma Tallgrass Prairie, canopy cover of all grasses, except switchgrass decreased a year after a summer prescribed fire (Adams et al. 1982). Likewise, within my study, individual species' responses to treatment were variable. In the early season, blue grama, fringed signalgrass, and sand dropseed were not affected by either burning treatment, but sideoats grama frequency increased with both summer and winter burning relative to no burning. Similarly, in a Kansas Tallgrass Prairie study, sideoats grama, a perennial grass, also increased in the presence of winter and early – mid spring fires, with the composition of the other perennial grasses remaining stable, indicating their persistence as subdominants in the ecosystem (Towne and Owensby 1984). In the late season, however, sideoats grama and sand dropseed were not affected by burning treatments, but blue grama frequency significantly decreased in the summer burning and no burning treatments and fringed signal grass frequency increased with summer burning. This may be explained by the findings of Towne and Owensby (1984), in which aggregates of perennial grasses decrease under late spring burning.

Fire may favor certain species, which then displaces other perennial grass species, which was the case for my study. However, the same holding for individual species' response to burning did not hold true for forbs within my study. Forb species were not affected by any treatment and overall forb cover decreased regardless of treatment. Both

forb and grass responses to season/time of burn can be interpreted by their phenological state at the time the burn takes place. In a winter burn study conducted in Texas coastal prairie, it was determined that later winter fires increased cool season forb production and enhanced grass production. In contrast, early winter burns were conducted before most forbs initiated growth (Hansmire et al. 1988). In the Kansas Flint Hills, winter burns were effective at reducing forb cover, which was supported by my study (Towne and Owensby 1984). Similar studies in south Texas also suggested that species such as cool season annual forbs may remain unaffected by summer burns due to irregular rainfall patterns (Ruthven and Synatzke 2002).

Overall, these observations may be explained by increasing frequency and intensity of droughts in the area, which may have nullified any potential positive responses from burning. This seems particularly likely in my case given that control plots behaved similarly to both burning treatments. Similar studies suggest that below average rainfall patterns and or below average precipitation occurring after a burn treatment can result in no treatment effect or a decrease in grass and forb productivity (Reynold and Bohning 1956, Ruthven and Synatzske 2002). Therefore, rainfall pattern variability may be a confounding factor effecting overall production, which could mask treatment effects (Kennan 1972).

In general, it seems that the determining factor of the potential effects of fire on populations and community compositions in grassland ecosystems is the interaction of the fire with both past and future events, such as drought, rainfall, and grazing occurring, particularly during the post-fire recovery phase (Frost and Robertson 1987). Drought, precipitation patterns, and herbivory affect potential fuel loads available, and in turn, fire intensity and the individual plants degree of recovery. Therefore, the effects of fire on species composition cannot be seen independently from the influences of these other variables. These integrated factors may complicate the interpretation of the results of studies on the seasonality of fire in grassland ecosystems (Frost and Robertson 1987).

Woody Vegetation

Fires in xeric grasslands and savannahs typically occur frequently, but are rarely intense enough to kill established woody species. Although canopy cover of woody plants may be reduced, density may increase as a result of increased recruitment from seeds stimulated by fire to germinate (Sweet 1982, Hodgkinson et al. 1984).

My results indicated that the unburned treatment was effective at reducing average percent cover of all woody plants as compared to the winter-burned treatment. However, the winter-burned treatment was approaching significance in also reducing percent cover of all woody plants as compared to the summer-burned treatment, which indicated no treatment effect at all. This suggests that the existing drought conditions may have induced woody plant mortality in the encroached landscapes. Bidwell et al. (2013) suggested that fire intensity affects the plant's response to fire. Low-intensity fires causes the wooded areas to shift towards a savannah, whereas high-intensity fires may have the ability to shift the woody vegetation into a sprout thicket if mature stands of trees are topkilled, which was consistent with my data. Winter burns are typically lower intensity fires, whereas summer burns are a higher intensity because of the warmer temperatures that exist during that time of year. The unburned treatment also resulted in a significant reduction of average number of woody plant clumps per intercept as compared to the winter burned treatment.

My results indicated that the winter burned treatment resulted in a reduction of average number of woody clumps per intercept as compared to the summer-burned treatment, which resulted in no treatment effect. However, the summer burned treatment was approaching significance in reduction of the average number of woody plant clumps per intercept. A similar long-term study conducted in Kruger National Park concluded that density of individual woody individuals was unresponsive to fire. However, plots that were dominated by small trees was highly responsive to fire regime, suggesting that savannahs are demographically resilient to fire, but may be structurally responsive to fire (Higgins et al. 2007). This may explain my results that in general, woody vegetation was less responsive in burned treatments as compared to unburned treatments. However, the winter burned treatment was more responsive than the summer burned treatment because winter burned plots generally had smaller woody species. This, coupled with existing climatic conditions allowed for a significant decrease of woody plants in the unburned plots because the existing drought conditions induced woody plant mortality in the encroached landscapes.

Average diameter of canopy for individual woody plant clumps did not vary between winter-burned and summer burned treatments. Interestingly, in the unburned treatments, average diameter of canopy for individual woody plant clumps significantly increased, suggesting that the presence of fire is not altering the sizes of woody vegetation clumps, but it is maintaining them especially during a drought period. Therefore, presence of fire might be negating potential negative effects of drought on woody vegetation clump sizes. In general, it appears that the effects of summer burning are potentially ameliorating the effects of the drought more so than in the winter-burning treatment. The effects of winter burning also appear to be ameliorating the effects of the drought as compared to the unburned treatment.

Burning results in the reduction of fuel loads and nutrient pool sizes, increases nutrient turnover rates within the soil, and redistributes these nutrients within the soil profile (Fisher and Binkley 2000). Presence of burning is then causing rapid regrowth of woody species because of increased nutrient cycling. Woody vegetation is also potentially responding to the onset of drought conditions by maintaining reserves needed for growth. Thus, once burning occurs, woody plants respond and regrowth occurs at a quick rate because of nutrient cycling as an effect of fire and a lack of precipitation, which is consistent with my results.

My results indicated that the unburned treatment had a stronger influence at reducing the percent cover of sandsage brush as compared to the winter-burned treatment and the winter-burned treatment was effective at reducing the percent cover of sandsage brush as compared to the summer-burned treatment, which indicated no treatment effect. This trend of the unburned control plots being more significant in reduction of woody species than the treated plots may also be explained as an artifact of the distribution of woody vegetation within the plots, and the resolution we have in terms of detecting the woody vegetation with the sampling methods used could also create potential problems. The average number of sandsage clumps per intercept was significantly reduced in the unburned, winter-burned, and summer-burned treatments, which could lead us to believe that climate variation may be more important to consider than treatment, because regardless of treatment, sandsage declined. Therefore, weather (drought) is taking more of a toll that exceeds the impacts of fire. It also seems to suggest that the effects of the drought are being more pronounced in the treated plots relative to the control plots. However, when examining other woody species, the average number of mesquite clumps per intercept was significantly increased in the unburned, winter-burned, and summerburned treatments.

Woody species, such as prickly pear, sandsage brush, and honey mesquite occurred in the Rolling Plains ecoregion historically (Wright et. Al 1976). As a result, these species have become difficult to manage because they are often resilient to fire and tend to resprout after a fire occurs (Heirman and Wright 1973, Ruthven et al. 2003). In studies conducted in South Texas rangelands, both summer and winter burning were effective at reducing honey mesquite, which could then potentially promote increases in herbaceous vegetation yields (Hansmire et al. 1988, Scifres and Hamilton 1993, Ruthven et al. 2003). It is suggested that summer burning may be effective at reducing honey mesquite if it is followed by significant rainfall and winter burning following periods of drought is recommended if the management goal is to reduce overall woody vegetation canopy cover (Ruthven et al. 2003). However, this was not the case with my study. Not only did a drought occur for large portion of this project, which could negate potential positive effects of fire, but all treatment plots have been continually grazed since 2007/2008.

Mesquite is a shrub or tree of the legume family that reproduces only by seed and not vegetatively. Mesquite pods therefore depend on foraging animals for seed dispersal. Dissemination of mesquite seed occurs when seeds are consumed and distributed fecally by cattle and many wildlife species such as javelina, coyotes, and deer (Fisher et al. 1959, Mooney et al. 1977, Ansley et al. 1997). Germination of mesquite is also enhanced when seeds are passed through the animal's digestive system (Archer 1989). Archer (1995) suggested that maximum mesquite seedling establishment is significantly related to overgrazing and periods of drought because competing herbaceous plant vigor and cover are reduced, which may help explain the results of my study. Recent studies also suggested that reoccurring growing season and dormant season burns often fail to kill mature established stands of mesquite (Ansley et al. 2001). Therefore, if controlling honey mesquite encroachment is the management goal, fires may need to occur during the seedling and growth stages to prevent further spreading.

After 2 years of data collection, Poole (2009) found that summer prescribed fire was effective at exposing more bare ground, which was consistent with what I found over the 16 year time period on the same project. However, he found that exposure of bare ground was most likely the result of the reduction of litter, which contradicts what I found, as litter cover did not vary among my treatments. Poole (2009) also found that grass cover significantly decreased in the summer burned plots as compared to the winter burned plots, which was consistent with my long-term results. However, individual grass species' responses to burning varied across both the short-term and long-term studies. Poole (2009) found no differences among treatments for frequency of blue grama and fringed signal grass, which was consistent with my early season results. In addition, I found that sideoats grama frequency increased in summer burning treatments. Whereas, in my late season of sampling, frequency of blue grama and fringed signal grass significantly increased in the summer-burned treatments. For forbs, Poole (2009) found that erect dayflower and narrowleaf globernallow increased in frequency and silver leaf nightshade decreasing in frequency with summer burning, which contradicts my results as individual species of forbs showed no trends.

Poole (2009) concluded that sideoats grama favorably responded to winter burning with an increase in frequency, which is what I found with my early season sample results. However, he found that blue grama frequency decreased in in the presence of winter burning and I found that blue grama appeared to be unaffected by winter burning. The short-term study demonstrated that winter burning did not affect forb cover, which was consistent with my results, however, forb cover was approaching significance in decreasing in the winter burned plots over time in the long-term study. Poole (2009) had individual forb species' responses to winter burning with peppergrass and plaintain being negatively affected and erect dayflower increasing, while I found no trends with those species. However, erect dayflower was approaching significance with my study.

For woody vegetation, Poole (2009) found that prescribed summer burning appeared to be effective at reducing canopy cover of sand sagebrush, honey mesquite, prickly pear, and yucca. I found no significant trends for the reduction of sand sagebrush and honey mesquite in the summer burned treatments. Poole (2009) concluded that winter burning was also significant at reducing honey mesquite cover, while I found no trends for honey mesquite cover with winter burning.

Overall, it appears that long-term studies are important for consideration as these studies may provide a different picture than their short-term studies. A majority of ecological research focuses on short-term results, but most key changes occurring within the environment take place over prolonged periods. Thus, long-term studies provide robust datasets and the results are critical to understand demographic fluctuations on populations, to identify conservation problems, and to determine key priorities to apply needed conservation measures. Because a majority of the important questions in ecology deal with predicting long-term ecosystem responses, testing the precision of these ecological predictions by observing the future is essential.

Management Implications

In grassland ecosystems, prescribed burning can be used to maintain woody plant diversity, increase herbaceous vegetation preferred by wildlife and livestock, reduce woody vegetation encroachment, and enhance wildlife habitat. In general, winter burns have been efficient at reducing shrub competition by top-killing/seed removal, controlling winter annuals, and improving forage quality. Summer burns are often more damaging because vegetation is typically drought stressed and therefore, highly flammable. A combination of increased temperatures, highly flammable fuel loads, and low humidity contributes to the intensity of summer burns. Therefore, summer prescribed burns can be effectively used for controlling vegetation. Depending on management goals, both burning treatments may be used in conjunction with one another. In summary, prescribed burning involves the combination of appropriate climactic and range conditions and seasonal timing.

Summer and winter prescribed burning in a sandsage ecosystem within the Rolling Plains of Texas were effective at increasing bare ground, reducing grass cover, and maintaining sizes of woody clumps. Therefore, a combination of both burning treatments is recommended for suppressing unwanted woody vegetation, increasing herbaceous plant community compositions preferred by both wildlife and cattle. Interpretation has often been made difficult because of the long-term severe drought that has persisted on the site over the last few years. However, a variable fire regime may maximize biodiversity and reduce potential for undesired changes. Other considerations that should be taken into account are the unpredictable climactic patterns in this region. Long-term periods of drought are common which could severely effect production of fuels that are necessary for a fire to occur. Therefore, continued monitoring of herbaceous and woody vegetation communities should elucidate treatment effects that are currently masked by drought effects, particularly if precipitation patterns and range conditions improve.

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Table I.1. Season, treatment, r^2 , and p-values derived from a regression between percent bare ground and sample year on the Matador WMA in Cottle County, Texas 2005 -2009 and 2018 - 2019.

Season	Treatment	\mathbf{r}^2	p-value
Early	Unburned	0.528	0.101
	Winter burned	0.367	0.201
	Summer burned	0.690	0.040
Late	Unburned	0.495	0.118
	Winter burned	0.444	0.148
	Summer burned	0.146	0.454

Table. I.2. Season, treatment, r², and p-values derived from a regression between percent grass and sample year on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.594	0.072
	Winter burned	0.250	0.311
	Summer burned	0.857	0.007
Late	Unburned	0.391	0.184
	Winter burned	0.062	0.633
	Summer burned	0.031	0.737

Table. I.3. Season, treatment, r^2 , and p-values derived from a regression between percent litter and sample year on the Matador WMA in Cottle County, Texas, 2005 –

Season	Treatment	r^2	p-value
Early	Unburned	0.131	0.480
	Winter burned	0.153	0.443
	Summer burned	0.114	0.511
Late	Unburned	0.169	0.417
	Winter burned	0.068	0.616
	Summer burned	0.142	0.460
Late	Unburned Winter burned Summer burned	0.169 0.068 0.142	0.417 0.616 0.460

2009, 2018–2019

Table I.4. Season, treatment, r², and p-values derived from a regression between percent forb and sample year on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2018 – 2019.

Season	Treatment	\mathbf{r}^2	p-value
Early	Unburned	0.464	0.136
	Winter burned	0.528	0.101
	Summer burned	0.515	0.107
Late	Unburned	0.129	0.484
	Winter burned	0.291	0.268
	Summer burned	0.196	0.378

Table I.5. Early Season treatment, r², and p-values derived from regression between herbaceous vegetation community metrics (richness, evenness, Shannon's diversity, modified Simpson's diversity) and sample year on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019.

Metric	Treatment	r^2	p-value
Richness	Unburned	0.228	0.278
	Winter burned	0.055	0.611
	Summer burned	0.114	0.459
Evenness	Unburned	0.003	0.901
	Winter burned	0.001	0.957
	Summer burned	0.086	0.522
Shannon's Diversity	Unburned	0.097	0.495
	Winter burned	0.014	0.798
	Summer burned	0.098	0.495
Modified Simpsons	Unburned	0.091	0.510
Diversity	Winter burned	0.001	0.933
	Summer burned	0.146	0.397

Table I.6. Season, treatment, r², and p-values derived from a herbaceous vegetation regression between Jaccard's index of similarity to 2019 and sample year on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.094	0.553
	Winter burned	0.530	0.100
	Summer burned	0.710	0.035
Late	Unburned	0.021	0.782
	Winter burned	0.451	0.143
	Summer burned	0.344	0.220

Table. I.7. Late Season treatment, r^2 , and p-values derived from regression between herbaceous vegetation community metrics (richness, evenness, Shannon's diversity, modified Simpson's diversity) and sample year on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 - 2019.

Metric	Treatment	r^2	p-value
Richness	Unburned	0.001	0.953
	Winter burned	0.010	0.827
	Summer burned	0.000	0.977
Evenness	Unburned	0.304	0.199
	Winter burned	0.233	0.273
	Summer burned	0.022	0.752
Shannon's Diversity	Unburned	0.045	0.647
	Winter burned	0.064	0.585
	Summer burned	0.003	0.901
Modified Simpsons	Unburned	0.066	0.577
Diversity	Winter burned	0.056	0.609
	Summer burned	0.069	0.566

Table I.8. Season, treatment, r^2 , and p-values derived from a regression between sideoats grama frequency and sample year on the Matador WMA in Cottle County, Texas, 2005 - 2009, 2018 - 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.246	0.257
	Winter burned	0.709	0.017
	Summer burned	0.713	0.016
Late	Unburned	0.164	0.367
	Winter burned	0.354	0.158
	Summer burned	0.181	0.340

Table I.9. Season, treatment, r^2 , and p-values derived from a regression between blue grama frequency and sample year on the Matador WMA in Cottle County, Texas, 2005 - 2009, 2018 - 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.140	0.407
	Winter burned	0.019	0.767
	Summer burned	0.194	0.321
Late	Unburned	0.590	0.043
	Winter burned	0.175	0.349
	Summer burned	0.695	0.019

Table I.10. Season, treatment, r², and p-values derived from a regression between fringed signalgrass frequency and sample year on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.075	0.550
	Winter burned	0.000	0.986
	Summer burned	0.057	0.604
Late	Unburned	0.373	0.145
	Winter burned	0.044	0.649
	Summer burned	0.574	0.048

Table. I.11. Season, treatment, r², and p-values derived from a regression between sand dropseed frequency and sample year on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.045	0.644
	Winter burned	0.000	0.972
	Summer burned	0.137	0.412
Late	Unburned	0.551	0.056
	Winter burned	0.286	0.215
	Summer burned	0.047	0.640

Table I.12. Season, treatment, r^2 , and p-values derived from a regression between western ragweed frequency and sample year on the Matador WMA in Cottle County, Texas, 2005 - 2009, 2018 - 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.001	0.938
	Winter burned	0.087	0.518
	Summer burned	0.045	0.644
Late	Unburned	0.020	0.759
	Winter burned	0.011	0.818
	Summer burned	0.086	0.522

I.13. Season, treatment, r², and p-values derived from a regression between dayflower frequency and sample year on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2018 – 2019.

Metric	Treatment	r^2	p-value
Percent cover of all woody	Unburned	0.495	0.051
plants	Winter burned	0.325	0.139
	Summer burned	0.072	0.519
Mean diameter of canopy for	Unburned	0.474	0.058
individual woody plant clumps	Winter burned	0.020	0.736
	Summer burned	0.297	0.162
Number of woody plant	Unburned	0.536	0.038
clumps per intercept	Winter burned	0.424	0.080
	Summer burned	0.382	0.101

Table I.14. Treatment, r², and p-values derived from a regression between percent cover of all woody plants, mean diameter of canopy for individual woody plant clumps, number of woody plant clumps per intercept and sample year measured using the line-intercept method on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2011, and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.001	0.925
	Winter burned	0.308	0.195
	Summer burned	0.301	0.201
Late	Unburned	0.002	0.921
	Winter burned	0.065	0.581
	Summer burned	0.168	0.359
Table I.15. Treatment, r², and p-values derived from a regression between percent cover of sandsage brush, mean diameter of canopy for individual sandsage clumps, number of sandsage clumps per intercept and sample year measured using the line-intercept method on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2011, and 2018 – 2019.

Metric	Treatment	r^2	p-value
Percent cover of Sandsage	Unburned	0.694	0.010
brush	Winter burned	0.411	0.086
	Summer burned	0.187	0.283
Mean diameter of canopy for	Unburned	0.000	0.954
individual Sandsage clumps	Winter burned	0.022	0.720
	Summer burned	0.075	0.510
Number of Sandsage	Unburned	0.544	0.036
clumps per intercept	Winter burned	0.442	0.071
	Summer burned	0.391	0.096

Table I.16. Treatment, r², and p-values derived from a regression between percent cover of mesquite, mean diameter of canopy for individual mesquite clumps, number of mesquite clumps per intercept and sample year measured using the line-intercept method on the Matador WMA in Cottle County, Texas, 2005 – 2009, 2011, and 2018 – 2019.

Metric	Treatment	r^2	p-value
Percent cover of honey	Unburned	0.104	0.434
mesquite	Winter burned	0.000	0.990
	Summer burned	0.005	0.866
Mean diameter of canopy for	Unburned	0.053	0.582
individual honey mesquite	Winter burned	0.121	0.398
clumps	Summer burned	0.085	0.482
Number of honey mesquite	Unburned	0.941	0.000
clumps per intercept	Winter burned	0.686	0.011
	Summer burned	0.875	0.000



Figure I.1. Blowout map of my study site indicating the location of Cottle County within Texas (A), the Matador Wildlife Management Area within Cottle County (B), Headquarters Pasture within the Matador WMA (C), and the design of the study plots within Headquarters Pasture (D).



Figure I.2. Diagram of my specific study plots within Headquarters Pasture on the Matador WMA within Cottle County, Texas. Headquarters pasture was divided into a 3 x 5 randomized complete block design; each block containing 3 -18 ha plots, resulting in 15 total plots. Each plot within a block was randomly assigned 1 of 3 treatments: summer burned (S), winter burned (W), and unburned (C)



Figure I.3. Diagram of transect sampling regime for measuring canopy cover of woody vegetation within each plot. Three 200-m north and south parallel transects spaced 100-m apart. 12 30-m east and west transects spaced 40-m apart lie perpendicular to the north and south running main transects. Woody canopy cover and frequency were measured along the 30-m transects at the Matador WMA in Cottle County, Texas during each year of sampling.



Figure I.4. Early season regression of percent bare ground sampled using a quadrat frame on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019.
Summer-burned treatment (solid line) and unburned treatment (hollow line) were significant, however, winter burning treatment was not significant so a regression is not presented.



Figure I.5. Early season regression of percent grass sampled using a quadrat frame on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Summerburned treatment (solid line) was significant and unburned treatment (hollow dotted line) was approaching significance, however winter-burned treatment was not significant so no regression is presented.



Figure I.6. Early season regression of percent forbs sampled using a quadrat frame on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Summer-burned treatment (black dotted line) and the winter-burned treatment (gray dotted line) were approaching significance. However unburned treatment was not significant so no regression is presented.



Figure I.7. Early season regression of Jaccard's index of similarity to 2019 of herbaceous vegetation on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Summer-burned treatment (black line) and winter-burned treatment (shaded line) were significant. However unburned treatment was not significant, so no regression is presented.



Figure I.8. Early season regression of sideoats grama frequency sampled using a quadrat frame on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 - 2019. Summer-burned treatment (solid black line) and winter burned treatment (shaded line) were significant. However, unburned treatment was not significant so no regression is presented.



Figure I.9. Early season regression of blue grama frequency sampled using a quadrat frame on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Summer-burned treatment (solid black line) and unburned treatment (hollow black line) were significant. However, winter burned treatment was not significant so no regression is presented.



Figure I.10. Late season regression of fringed signalgrass frequency sampled using a quadrat frame on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Summer-burned treatment (solid black line) was significant. However, unburned treatment and winter burned treatment were not significant so no regression is presented.



Figure I.11. Late season regression of sand dropseed frequency sampled using a quadrat frame on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Unburned treatment (hollow line) was significant. However, summer burned treatment and winter burned treatment were not significant so no regression is presented.



Figure I.12. Average percent cover of all woody plants sampled using the line-intercept method on the Matador WMA in Cottle County, Texas 2005 – 2009 and 2018 - 2019. Unburned treatment (hollow line) was significant. Summer burned treatment and winter burned treatment were not significant so no regression is presented.



Figure I.13. Average diameter of all woody plants sampled using the line-intercept method on the Matador WMA in Cottle County, Texas 2005 – 2009, 2011, and 2018 – 2019. Unburned treatment (hollow line) was significant. However, summer- burned treatment and winter – burned treatment were not significant so no regression is presented.



Figure I.14. Average number of woody plant clumps per intercept sampled using the lineintercept method on the Matador WMA in Cottle County, Texas 2005 - 2009, 2011, and 2018 - 2019. Unburned treatments (hollow line) winter burned treatment (gray line) were significant. Summer – burned treatment (dotted black line) was approaching significance.



Figure I.15. Average percent cover of sandsage brush sampled using the line-intercept method on the Matador WMA in Cottle County, Texas 2005 – 2009, 2011, and 2018 – 2019. Unburned treatments (hollow line) and winter burned treatments (gray line) were significant. However, summer burned treatments were not significant so no regression is presented.



Figure I.16. Average number of sandsage brush clumps per intercept sampled using the line-intercept method on the Matador WMA in Cottle County, Texas 2005 – 2009, 2011, and 2018 – 2019. Unburned treatments (hollow line), winter burned treatments (gray line), and the summer burned treatments (solid black line) were significant.



Figure I.17. Average number of mesquite clumps per intercept sampled using the line-intercept method on the Matador WMA in Cottle County, Texas 2005 – 2009, 2011, and 2018 – 2019. Unburned treatments (hollow line), winter burned treatments (gray line), and the summer burned treatments (solid black line) were significant.

CHAPTER II

LONG- TERM EFFECTS OF FIRE SEASONALITY ON SMALL VERTEBRATES IN A ROLLING PLAINS ECOSYSTEM

INTRODUCTION

Habitat variables have a robust influence on the abundance and distribution of wildlife at different spatial scales, and likewise, these organisms may directly affect the structure of their environments (Cody 1981, Vinton and Collins 1989). These community dynamics are driven by interactions produced by a complex feedback system. Therefore, the use of natural disturbances used in combination, such as grazing and fire as management practices, are needed to maintain local and regional faunal biodiversity within these ecosystems (Plumb and Dodd 1993, Steuter et al. 1990).

Prescribed fire is used as a management tool to achieve various objectives, such as regulating fuel load accumulation, increasing nutrient cycling essential for plant growth, and reducing undesirable woody vegetation. Fire is essential for the persistence and development of prairie ecosystems (Axelrod 1985). Regular burning of prairies increases flora productivity (Briggs and Knapp, 1995), decreases accumulated litter (Hulbert 1988), and can remove excess woody vegetation (Heisler et al. 2003). Without fire, native grasslands can potentially become colonized by woody vegetation rapidly. Fire suppression results in an invasion of woody vegetation across various landscapes and fire is the main factor keeping unwanted woody species from these native ecosystems (Saur 1950, Stewart 1956).

Prescribed fire is also a critical tool for wildlife management. It can influence the quantity and quality of cover and food by altering the habitat through succession (Leopold 1933, Komarek 1963, Wade and Lunsford 1989). Grazers and browsers will regularly fixate on an area after the burn because the forage is more palatable, available, and nutritious. In Texas, rangelands have been greatly improved and are typically grazed within 3 to 7 months post burns (Wright and Bailey 1982).

The Rolling Plains are covered by mesquite-grasslands of a savannah nature and encompass the southwestern portion of Oklahoma and the northcentral range of Texas (Hamilton 1962). Although it is considered to be primarily composed of honey mesquite (*Prosopis glandulosa*) and grama (*Bouteloua* spp.) grasses, the region displays a mosaic of plant communities, comprised of mixed-grass savannahs and riparian corridors. Historically, this area is suggested to have had a fire frequency of 5 to 10 years, which was dependent upon available fuel loads (Wright and Bailey 1982). Following settlement, fire suppression coupled with improper grazing techniques exhausted naturally occurring grasses, removed fuel or biomass critical for burns, and allowed woody plant species encroachment. With a lack in use of fire, wildlife can potentially decline in numbers and diversity over time (Wright and Bailey 1982). Therefore, fire is an ecosystem process that is essential to fire-dependent ecosystems and species in these areas have evolved adaptations to respond positively to fire. Fires often cause short-term increases in many important wildlife foods. The long-term effects of fire on animal communities depend on the extent of change to the habitat structure. When fire intensity and frequency from pre-European settlement patterns changes, critical habitat for wildlife species often declines.

Habitats are constantly changing via succession; therefore, some form of maintenance is needed to sustain suitable habitat for many wildlife populations. Thus, the value of prescribed fire for wildlife must be considered along with the diversity of species, the native flora types, weather patterns, succession of vegetation, and planned fire intensity of that region (Wright and Bailey 1982). Understanding the role of fire in structuring the flora of a habitat is critical for land management choices that may prevent the decline or loss of threatened species and the native vegetation.

Previous studies conducted throughout various ecosystems have indicated that the presence of fire affects both the community composition and abundances of species either indirectly through altering the habitat, or directly through mortality as a result of the fire (Wilgers and Horne 2006). However, the impacts of wildlife on vegetation structure in grassland communities has been the primary focus as opposed to how vegetation may affect wildlife abundances and distributions, particularly non-game wildlife. And, despite the numerous studies concerning the effects of prescribed burning on wildlife, responses to fire of herpetofauna have received little attention (Harlow and Van Lear 1981, 1987; Means and Campbell 1981). Not only does the physiology and natural history of this group of vertebrates make them of valuable use for research, but also concerns over potentially declining populations increases the need for exploration into the effects of fire on herpetofauna for management practices (Pechmann and Wilbur 1994, deMaynadier and Hunter 1995). There is also evidence that small vertebrate communities are being

ignored and declining primarily because management is becoming more ecosystemoriented rather than single species focused. This creates problems because management practices for conservation can average broadly across species with differing requirements (Lindenmayer et al. 2007). Therefore, filling in this data gap with longer-term studies will help us better-manage these species that are potentially in trouble and being ignored.

Both direct and indirect effects of prescribed fire on small vertebrates may occur. Historically, fires were a prominent natural disturbance within various ecosystems (Wahlenburg 1946, Hunter 1990, Rogers 1996). Because fire has the utility to restore or maintain vegetation within habitats, herpetofauna and small mammals historically adapted to these ecosystems and benefited indirectly from these treatments (Daubenmire 1968, Russel et al. 1999). Burn studies within chaparral ecosystems of southern California indicated that diversity of reptile species was significantly correlated with vegetation density (Hanes 1971). Similarly, overall herpetofaunal diversity has increased in fire-dependent savannahs of Australia (Catling and Newsome 1981, Braithwwaite 1987, Friend 1993, Pianka 1996). However, direct effects or mortality of fire on small vertebrates, although seldom, may occur. Direct effects often include a response of movement away from fire, and many species of herpetofauna and small mammals seek cover in the soil or burrows (Speake et al. 1979). On the other hand, more mobile species simply disperse from burns (Komareck 1969, Means and Cambell 1981, Patterson 1984). However, overall mortality rates of small vertebrates from fire are often suggested to be relatively low.

Because there is a lack of data on the effects of seasonal prescribed fire and little has been published concerning the beneficial effects of prescribed fire on nongame wildlife in the Rolling Plains of Texas. I initiated a project to explore the long-term effects of fire seasonality on small vertebrates in a sand sage prairie ecosystem within the Rolling Plains of Texas. My objective was to determine the effects of summer and winter prescribed burns on the abundance, richness, evenness, and diversity of herpetofauna and small mammals in the Rolling Plains of Texas.

STUDY AREA

Matador Wildlife Management Area

My study site lies within the 11,405-ha Matador Wildlife Management Area (WMA) about 10 km north of the town of Paducah in Cottle County, Texas (Figure I.1). The property was purchased by the state of Texas in 1959 with Pittman-Robertson funds and is managed as a research and demonstration area by the Wildlife Division of Texas Parks and Wildlife Department. The area is comprised of grass savannahs and riparian corridors along the Pease River drainage. The area's climate is identified as a subtropical climate with dry winters, and relatively hot, humid summers with an average annual precipitation of 54 cm. Historically, the rainy season primarily occurred from May to June and the growing season averages 219 days (Richardson et al. 1974). The area is susceptible to periods of drought with one year out of every 10 receiving less than 33 cm of rain. Conversely, annual rainfall exceeds 76 cm one year out of every 10. Little benefit is acquired during these extremely wet years because heavy downpours result in an excessive runoff (Richardson et al. 1974). The terrain is rough in the west and levels off in the east with elevations ranging from 488 and 640 m above sea level. Topography is characterized by rolling plains to steep slopes and canyons (Richardson et al. 1974).

The dominant woody vegetation on Matador WMA includes sand sagebrush (*Artemisia filifolia*), honey mesquite, eastern cottonwood (*Populus deltoides*), salt-cedar (*Tamarix* sp.), western soapberry (*Sapindus saponaria*), netleaf hackberry (*Celtis reticulata*), and redberry juniper (*Juniperus pinchotii*). The primary grasses found on the area include sideoats grama (*Bouteloua crependula*), sand dropseed (*Sporobolus cryptandrus*), purple three-awn (*Aristida purpurea*), little bluestem (*Schizachyrium scoparium*), and blue grama (*Bouteloua gracilis*). Common forbs found include western ragweed (*Ambrosia psilostachya*), lamb's quarters (*Chenopodium album*), marestail (*Erigeron canadensis*), and plaintain (*Plantago* spp., Hodge 2000, Hall 2005)

METHODS

Burning and Experimental Design

My study site consisted of 273-ha of sand sagebrush grasslands in Headquarters Pasture (Figure II.1). Headquarters pasture was divided into a 3 x 5 randomized complete block design; each block containing 3 18 – ha plots, resulting in 15 total plots (Figure II.3). Each plot within a block was randomly assigned1 of 3 treatments: winter-burned, summer-burned, or unburned (Figure II.2). The burning regime for the pasture has been maintained by the Texas Parks and Wildlife Department and each burned plot has been treated 3-4 times since the initial project. Summer burns are conducted between July and August and winter burns are conducted between December and February largely depending on manpower, weather conditions, and fuel loads available.

The decision was made beginning in 2007/2008 that continuous cattle grazing was to be initiated on the study site. Grazing was not restricted to a plot-by-plot basis. Therefore, the availability of grazing was equal across all study plots. The pasture was stocked at 16 animal units (cow/calf).

The initial project was started in 2004, and data for small vertebrates was collected over the years of 2005 - 2009. I came along in 2018 and 2019 and replicated their methods and the methods of data collection are consistent with the ones I describe below.

Small Vertebrate Sampling

Herpetofauna - Herpetofauna were sampled using y-shaped drift fence arrays with pitfall traps placed in the center of each plot as described by Ruthven et al. (2002). Drift fence arrays were monitored for 14 consecutive days during the early season (May-June) and late season (July-August) of 2004 – 2009 and 2018 – 2019. Species and cohort specific toe clip code were recorded for all herpetofauna captured to allow determination of recaptures.

Small Mammal Drift Fence Arrays – Small mammals were sampled using yshaped drift fence arrays with pitfall traps as described by Ruthven et al. (2002) placed in the center of each plot. Drift fence arrays were monitored for 14 consecutive days during the early season (May-June) and late season (July-August) of 2005, 2006, 2018, and 2019. Species, sex, hindfoot length, and tail length were recorded for small mammals captured.

Small mammal Sherman Live Trapping – Small mammals were also sampled utilizing Sherman live traps (35 cm x 8 cm x 9 cm) set in transects (Simpson et al. 1996) in conjunction with drift fence arrays. Sherman live trapping was conducted during the spring and summer during 2005 and 2006 and during the summer in 2018 and 2019. One block of 3 plots was sampled during each trapping session. Each plot had a total of 106 traps set in 3 north to south running transects (Figure II.3). All traps were baited with peanut butter and oat balls wrapped in wax paper. Traps were checked each morning and closed upon checking and reopened in the evening for 4 consecutive evenings, totaling 424 traps nights per plot. Captured individuals were identified, sexed, and marked by hair clipping to allow determination of recaptures. Hindfoot, tail, and ear length were also recorded to assist with species identification.

In my first year of sampling (2018), Sherman live trapping resulted in 6,360 trap nights of sampling across all plots. However, only 10 individual small mammals were captured. My second year of sampling (2019) resulted in 2,544 trap nights with only 2 individual small mammals captured. Because of the small sample size and low capture rate as compared to effort, I was not able to compare potential treatment effects on small mammals captured from Sherman live trapping, and, therefore, Sherman live trapping was discontinued. Similar patterns happened following the 2005 year of sampling and continued from 2006 – 2009. Therefore, we have no long-term trend data for mammal populations during this study.

Small Vertebrate metrics

Common Species for Small Vertebrates - Species comparisons for all taxonomic groups were done for species in which sample sizes were large enough to make valid comparisons (>10% of total individuals). Therefore, abundance of individual small vertebrate species comparisons were only done for six-lined racerunner (*Cnemidophorus sexlineatus*), Great plains skink (*Eumeces obsoletus*), plains threadsnake (*Leptotyphlops duclis*), prairie lizard (*Sceloperus undulatus*), western narrowmouth toad (*Gastrophryne olivacea*), northern pygmy mouse (*Baiomys taylori*), and plains harvest mouse (*Reithrodontomys montanus*). Abundances of individuals for common species were regressed against sample year.

Community Metrics - Community metrics were also used to evaluate the longerterm responses to the different burning regimes. Small vertebrate species diversities were calculated using Shannon-Weiner diversity indices. Shannon-weiner diversity was calculated as follows:

$$H' = -\Sigma pi(lnpi)$$

Modified Simpson's diversity was calculated as follows:

Mod
$$D = 1 - \Sigma p i^2$$

Where pi = proportion of the ith species in the sample (Pielou 1975, Magurran 1988). Herbaceous vegetation evenness was calculated as follows:

$$Evenness = \frac{H'}{H'_{max}}$$

Where H' = Shannon-Weiner diversity and H' $_{max} = \ln(S)$, where S = species richness (Magurran 1988). I then compared overall small vertebrate species richness,

evenness, and diversity using a linear regression. I regressed year of sampling against community metrics to evaluate the longer-term responses to the different burning regimes. Analyses were conducted for early and late season data separately to avoid the potentially confounding influence of trapping season on treatment effect. For small vertebrates, I also calculated a Jaccard's indices of similarity to compare small vertebrate species documented in each year to the species composition I detected in 2019. Jaccard's index was calculated as follows:

$$J = j/(a+b-j)$$

Where a = number of species captured in year 1, b = number of species captured in year 2, and j = number of species captured in both years. I regressed Jaccard's index of similarity to 2019 against each year to look for changes in species composition over time (Magurran 1988). Because analyses were exploratory, I set $\alpha = 0.1$ for all comparisons.

RESULTS

Individual Species Responses – Early Season

There were 210 array-nights per season of sampling per year and there were 1,584 individuals captured across 23 different species of herpetofauna (Table II.1, Table II.1 Cont.) and 876 individuals captured across 6 different species of small mammals (Table II.2).

In the early season of sampling, individual small vertebrate species abundances of the Great Plains skink did not differ among summer-burned ($F_7 = 1.033$, P = 0.356, $r^2 = 0.171$) and unburned treatment ($F_7 = 0.828$, P = 0.404, $r^2 = 0.142$), but winter-burned

treatment was approaching significance ($F_7 = 2.874$, P = 0.151, $r^2 = 0.365$, Table II.2). Abundance of plains threadsnake did not vary among summer-burned ($F_7 = 0.202$, P = 0.672, $r^2 = 0.039$), winter-burned ($F_7 = 0.325$, P = 0.593, $r^2 = 0.061$), and unburned ($F_7 = 1.000$, P = 0.363, $r^2 = 0.167$, Table II.3) treatments. Abundance of prairie lizard did not vary among summer-burned ($F_7 = 0.134$, P = 0.729, $r^2 = 0.026$), winter-burned ($F_7 = 1.116$, P = 0.339, $r^2 = 0.183$), and unburned ($F_7 = 1.793$, P = 0.238, $r^2 = 0.264$, Table II.4) treatments. Western narrowmouth toad abundance did not vary among winter-burned ($F_7 = 1.781$, P = 0.240, $r^2 = 0.263$) and unburned ($F_7 = 0.004$, P = 0.950, $r^2 = 0.001$ treatments but was approaching significance in increasing in the summer-burned ($F_7 = 3.951$, P = 0.104, $r^2 = 0.441$) treatments in the early season of sampling (Table II.5). However, total abundance of six-lined racerunners significantly increased over time in summer-burned ($F_7 = 27.227$, P = 0.003, $r^2 = 0.845$) and unburned ($F_7 = 12.613$, P = 0.016, $r^2 = 0.716$) treatments as compared to the winter-burned treatment ($F_7 = 2.698$, P = 0.161, $r^2 = 0.350$), which was approaching significance (Table II.6, Figure II.4).

Individual comparisons for the abundance of the northern pygmy mouse did not differ among summer-burned ($F_7 = 1.316$, P = 0.303, $r^2 = 0.208$), winter-burned ($F_7 = 1.203$, P = 0.322, $r^2 = 0.208$), and unburned ($F_7 = 1.665$, P = 0.253, $r^2 = 0.249$) treatments in the early season of sampling (Table II.10). However, abundance of the plains harvest mouse decreased over time in the unburned treatment ($F_7 = 4.65$, P = 0.083, $r^2 = 0.482$) and tended to decrease in the summer-burned ($F_7 = 2.57$, P = 0.169, $r^2 = 0.339$) and winter-burned ($F_7 = 3.61$, P = 0.116, $r^2 = 0.419$) treatments as well (Table II.11, Figure II.5).

Community Responses – Early Season

In the early season, species richness significantly decreased over time in the winter burned plots (F_7 =4.973, P= 0.076, r² = 0.499) as compared to summer burned (F_7 = 0.003, P = 0.962, r² = 0.001) and unburned plots (F₇ = 1.092, P= 0.344, r² = 0.179, Table II.7, Figure II.5). No treatment effect was detected for evenness in the summerburned (F₇= 0.051, P= 0.830, $r^2 = 0.010$), winter-burned (F₇ = 0.010, P=0.923, $r^2 =$ 0.002), and unburned plots (P= 1.859, $r^2 = 0.271$, Table II.8, Figure II.6). Shannon-Weiner diversity significantly decreased over time in the winter burned plots ($F_7 = 5.002$, P=0.076, $r^2 = 0.500$) with no significance detected in the summer-burned (F₇ = 0.003, P= $0.961, r^2 = 0.001$) and unburned plots (F₇ = 1.122, P= 0.338, r^2 = 0.183, Table, II.9, Figure II.7). No treatment effect was detected for modified Simpson's diversity in the summer burned ($F_7 = 0.023$, P = 0.885, $r^2 = 0.005$) and unburned ($F_7 = 1.505$, P = 0.275, $r^2 = 0.231$) treatments as compared to the winter burned treatment (F₇ = 2.908, P = 0.149, $r^2 = 0.368$), which was approaching a significant increase in the early season of sampling. No treatment effect was detected for total abundance of small vertebrates in the summer burned (F₇ = 514, P = 0.506, r^2 = 0.093), winter burned (F₇ = 1.647, P = 0.256, r^2 = 0.248) and unburned ($F_7 = 1.698$, P = 0.249, $r^2 = 0.254$) treatments in the early season of sampling (Table II.10, Figure II.8).

However, Jaccard's index of similarity demonstrated that in the early season, species composition of the small vertebrate community did shift overtime in the summer burned treatment ($F_7 = 4.630$, P = 0.098, $r^2 = 0.536$) as compared to the winter burned treatment ($F_7 = 3.626$, P = 0.130, $r^2 = 0.475$), which was approaching significance in decreasing the species composition, and the unburned treatment ($F_7 = 0.142$, P = 0.726, $r^2 = 0.034$), which showed no trend in shifting the species composition of the small vertebrate community over time (Table II.15, Figure II.9).

Individual Species Responses – Late Season

In the late season, individual herpetofauna species abundance of the Great Plains skink did not differ among winter-burned ($F_7 = 0.114$, P = 0.749, $r^2 = 0.022$), unburned $(F_7 = 0.116, P = 0.747, r^2 = 0.023)$, and the summer-burned $(F_7 = 0.002, P = 0.963, r^2 < 0.023)$ 0.001) treatments (Table II.2). Prairie lizard abundance did not differ among summerburned (F₇ = 1.416, P = 0.288, $r^2 = 0.221$), winter-burned (F₇ = 0.716, P = 0.436, $r^2 =$ 0.125), and unburned ($F_7 = 1.347$, P = 0.298, $r^2 = 0.212$, Table II.4) treatments. Western narrowmouth toad abundance did not differ among summer-burned ($F_7 = 0.358$, P = 0.576, $r^2 = 0.067$), winter-burned (F₇ = 0.368, P = 0.570, r^2 = 0.069), and unburned (F₇ = 0.379, P = 0.565, r² = 0.070, Table II.5) treatments. Abundance of the six-lined racerunner increased over time, regardless of treatment (summer: $F_7 = 46.791$, P = 0.001, $r^2 = 0.903$; winter: $F_7 = 13.950$, P = 0.014, $r^2 = 0.736$; unburned: $F_7 = 17.478$, P = 0.009, $r^2 = 0.778$, Table II.6 Figure II.10). Plains threadsnake abundance increased in the presence of a winter-burned ($F_7 = 14.818$, P = 0.012, $r^2 = 0.748$) treatment, with no treatment effect detect in the summer burned ($F_7 = 0.807$, P = 0.410, $r^2 = 0.139$) and unburned treatment ($F_7 = 0.186$, P = 0.684, $r^2 = 0.036$, Figure II.11).

No treatment effect was detected for the individual comparisons for the abundance of the northern pygmy mouse (summer: $F_7 = 1.752$, P = 0.242, $r^2 = 0.259$; winter: $F_7 = 1.599$, P = 0.261, $r^2 = 0.242$; unburned: $F_7 = 1.441$, P = 0.283, $r^2 = 0.223$)

and the plains harvest mouse (summer: $F_7 = 1.752$, P = 0.350, $r^2 = 0.174$; winter: $F_7 = 1.599$, P = 0.241, $r^2 = 0.261$; unburned: $F_7 = 1.441$, P = 0.295, $r^2 = 0.214$) in the late season of sampling.

Community Responses – Late Season

In the late season of sampling, no treatment effect was detected for the total abundance of individuals in the summer-burned ($F_7 = 0.547$, P = 0.493, $r^2 = 0.099$), winter-burned ($F_7 = 0.617$, P = 0.468, $r^2 = 0.110$), and unburned ($F_7 = 0.531$, P = 0.499, $r^2 = 0.096$) treatments. No treatment effect was detected for species richness in the summer-burned ($F_7 = 0.040$, P = 0.850, $r^2 = 0.008$), winter-burned ($F_7 = 0.240$, P = 0.645, $r^2 = 0.046$), and unburned treatments ($F_7 = 1.408$, P = 0.289, $r^2 = 0.220$, Table II.7, Figure II.12). Over time, no treatment effect was detected for evenness of small vertebrates in the summer-burned ($F_7 = 1.774$, P = 0.240, $r^2 = 0.262$), winter-burned ($F_7 = 0.470$, P = 0.523, $r^2 = 0.086$), and unburned plots ($F_7 = 0.284$, P = 0.617, $r^2 = 0.054$, Table II.3, Figure II.13). No treatment effect was detected for both Shannon's diversity (summer: $F_7 = 0.280$, P = 0.619, $r^2 = 0.053$; winter: $F_7 = 0.003$, P = 0.959, $r^2 = 0.001$; unburned: $F_7 = 2.628$, P = 0.166, $r^2 = 0.345$) and modified Simpson's diversity (summer: $F_7 = 2.628$, P = 0.166, $r^2 = 0.345$) in the late season of sampling (Table II.4, Figure II.14).

However, Jaccard's index of similarity demonstrated that in the late season, species composition of the small vertebrate community did shift overtime in the unburned treatment ($F_7 = 10.480$, P = 0.032, $r^2 = 0.724$) as compared to the winter burned treatment ($F_7 = 521$, P = 0.510, $r^2 = 0.115$), and the summer burned treatment ($F_7 = 0.408$, P = 0.558, $r^2 = 0.093$), which showed no trend in shifting the species composition of the small vertebrate community over time (Table II.15, Figure II.5).

DISCUSSION

Small Vertebrates - Community

Prescribed burning has been suggested to be a useful management tool in conjunction with other applicable treatments to benefit herpetofaunal and small mammal communities by rehabilitating the ecological mosaic of succession on landscapes and plant species compositions required by many species (Means and Campbell 1981, Campbell and Christman 1982, Hunter 1990, Russell et al. 1999). Any mortality that may occur as a direct result from prescribed burning is often outweighed by the many benefits to the habitat, which should increase species diversity over the long-term and at broader scales (Vogl 1973).

Burning on my study site was effective at increasing the surface area of bare ground and the presence of burning shifted the species composition of both the small vertebrate and herbaceous plant communities over time. These changes were similar to other findings in grassland ecosystems (Daubenmire 1968). My results indicated that in the early season, small vertebrate populations decreased or tended to decrease over time with winter burning. However, early season comparison indices demonstrated the relationship for unburned treatments was not significant but the relationship for the summer burned treatment showed an increase in similarity over time, with the winter burned treatment approaching significance in a decrease in similarity over time. Similar results were also reported in Bolivian tropical forests (Fredricksen and Fredricksen 2002). One thing that could be driving the winter burned effect might be due to the relative disappearance of small mammals from the study in the later part of the years. The winter burned trend might be being pulled down purely from the most recent data point as well. So a couple more additional years of data are needed to really clarify the trend. Species richness and Shannon-Weiner small vertebrate species diversity of small vertebrates decreased over time with winter-burning. However, individual species of herpetofauna showed positive trends with summer and winter-burning as compared to small mammals, therefore, community trends may be negatively affected by the declining numbers of small mammal species. Several studies documented increases in herpetofaunal communities and the importance of fire for herpetofauna in xeric pine forests in Florida, with most species being found in open, early-succession habitats with high proportions of bare ground, which was consist with my findings (Means and Campbell 1981, Campbell and Christman 1982, Stout et al. 1988, Greenberg et al. 1994). My results also indicated that the total number of individuals captured declined over time in the unburned plots. This may be explained by the litter-encroached habitat detected in unburned areas. In the absence of fire, many species of herpetofauna will disappear as areas become inhospitable because of dense overgrown vegetation and absence of bare ground (Campbell and Christman 1982, Greenberg et al. 1994). In the late season, no significant trends were detected for small vertebrate community metrics.

Overall, it appears that without frequent fire or under fire-altered regimes (seasons), understory succession may not be significantly stunted to promote grass and

forb establishment with areas of exposed bare ground interspersed throughout, which is critical for small vertebrate communities (Brockway and Lewis 1997, Palis 1997).

My results indicated that in general, summer burning seems to have a greater positive benefit on small vertebrate communities as compared to winter burning. Fire alters live vegetation structure, which, in turn alters suitable habitats available. Prescribed fires also may cause an increase in solar radiation (Means and Campbell 1981). These thermal alterations to the environment are often beneficial to the maintenance of species such as herpetofauna, which are endemic to xeric prairies.

Community metrics did not change over time in the unburned treatment regardless of season, suggesting that these differences are treatment related and not weather related. However, interpretation has been made difficult because of the long-term severe drought that has persisted on the site over the last few years. In general, summerburning seems to have a greater positive benefit on small vertebrate communities over winter-burning. But my results have indicated that both burning treatments improve conditions for small vertebrates relative to lack of burning. My results also suggested that burning has the potential to reduce the overall negative effects of the prolonged drought on the overall total individuals captured.

Individual Species - Herpetofauna

My results indicated that in the early season, burning did not have an effect on the common species, as Great Plains skink, Plains threadsnake, prairie lizard, and western narrowmouth toad did not differ in abundance among treatments. Common species, which did not show change in abundance, could potentially be correlated to the drought
that has persisted on the study site over the past few years.

Where six-lined racerunners occur within prairies, they require patches of open ground and areas where vegetation is sparse because they are an arid-adapted ectothermic species (Fitch 1956). An increased exposure of bare ground to solar radiation leads to warmer soil surface temperatures. In my study, six-lined racerunners increased over time regardless of treatment. A proposed explanation for an increase in six-lined racerunner populations regardless of treatment could be because of the weather's effect on habitat availability. Warmer weather conditions may have been better overall for six-lined racerunners. Therefore, six-lined racerunners were less selective of treatment plots because of reduced herbaceous plant cover and increased surface area of bare ground. Thus, the protracted drought might be increasing bare ground in the control plots, as fire does in the burned plots. In studies conducted in east central Alabama and west central Georgia, adult six-lined racerunners entered hibernation during the late summer with juveniles remaining active into November (Etheridge et al. 1983). However, this was not the case with my study, as daily temperatures remained over $15 - 18^{\circ}$ C, which appears to be essential to bring racerunners out of hibernation (Hoddenbach 1966). This species could also be moving across treatment plots because of the plots' relatively small size and because different treatments lie adjacently to one another. However, this seems an unlikely explanation given the differences in habitat I detected (Chapter I).

My results indicated that in the late season, burning did not have an effect on Great Plains skinks, prairie lizards, and western narrowmouth toad abundance. Studies conducted in central Florida and the South Texas Plains observed increases in prairie lizard abundance in the fall with less captures occurring when air temperatures were high (Davis and Verbeek 1972, Campbell and Christman 1982, Ruthven et al. 2002b). This may be the case for no effect detected on prairie lizard abundances in my study in the rolling plains, where summer temperatures routinely exceed 38°C.

Ruthven et al. (2002b) suggested that the lack in differences in seasonal abundance of the Plains narrowmouth toad could be attributed to precipitation. Therefore, captures will likely be influenced by precipitation and relative humidity, which could explain why captures of amphibians for my study were low because of the persisting drought.

My results suggested that Plains threadsnake abundance increased in the presence of both burning treatments. In general, burning reduces litter and increases soil surface temperatures (Evans 1984). Ants often respond positively to burning because of the increase in soil surface temperatures because they are thermophilic species. Ant abundance may be increasing because of burning, which in turn, benefits Texas threadsnakes because they consume larval ants and termites. Therefore, both burning treatments were beneficial for Plains threadsnakes.

After two years of data collection, Poole (2009) found that in the short-term, summer and winter burning had little effect on herpetofauna. Poole (2009) suggested that burning did not have a significant effect on common species, as prairie lizard and Great Plains skink did not differ in abundance among treatments, which is consistent with what I found with my long-term results.

Poole (2009) indicated that community comparisons for herpetofauna abundance,

species richness, and diversity did not differ among treatments. I also found that small vertebrate community metrics did not generally differ among treatments for the early season of sampling. However, in the early season, I detected that species richness decreased over time in the summer-burned plots. Over time, diversity decreased in the summer treatment in the early season of sampling. In the late season, I found no trends for community comparisons in my late season of sampling.

Individual Species - Small Mammals

The indirect effects of prescribed fire on small mammals often results in changes of resource needs such as food availability and modification of habitats through removal of litter and increased exposure of bare ground (Daubenmire 1968, Simmons 1991). Burning on my study site was effective at increasing the surface area of bare ground and presence of burning shifted the species composition of the herbaceous plant community over time. Kaufman et al. (1990) suggested that differences in abundances of small mammals when comparing burned versus unburned areas were results from recruitment and disbursement. I found no differences from treatment effect and differences in early season to late season sampling for species abundance of small mammals, richness, diversity, or evenness.

It is suggested that omnivorous and granivorous small mammals occur in grasslands of low productivity (short grass), omnivorous and herbivorous small mammals occur in grasslands of somewhat higher productivity (mixed-grass), and herbivorous rodents occur in grasslands of relatively high productivity (tallgrass; French et al. 1976). Grassland productivity, in turn, is affected by climatic variables, disturbances, and topography (Kaufman and Kaufman1989). In general, precipitation is the key climactic factor in grassland productivity (Burke et al. 1991).

Relatively few studies have been conducted on the prolonged effects of drought on small mammal community metrics in grassland ecosystems. However, it has been suggested that these effects of drought on small mammals typically occur in the season following a drought year (French et al. 1976). In studies conducted in the tallgrass Konza Prairie of northeastern Kansas, many species of small mammals abundances were significantly correlated with precipitation. Elliot's short-tailed shrew (*Blarina hylophaga*) was positively correlated with precipitation that occurred from January-August. Whitefooted mice (*Peromyscus leucopus*) abundance was positively correlated with summer precipitation (Kaufman et al. 1995, Kaufman and Kaufman 1989). The abundance of deer mice (*Peromyscus maniculatus*) varied in a mixed-grass prairie with relatively high densities occurring in wet years and low densities observed during dry years (Kaufman 1990).

Overall, it appears that droughts may have more of an effect on small mammal populations than fire does. Direct effects of drought may include lack of free water and indirect effects may include lack of food availability and changes in vegetation. Lack of water affects the demographic process of vegetation within a habitat, and this in turn significantly affects the abundance of small mammals (Kaufman and Kaufman 1997).

Individual comparisons for the abundance of the northern pygmy mouse did not differ among treatments in the early season of sampling. A study in California examined the effects of drought on small mammals and found that there was an overall reduction in small mammal abundance and species diversity during a drought (Desideri 2016), which is consistent with my results. However, abundance of the plains harvest mouse decreased over time in the unburned treatment and tended to decrease in the summer-burned and winter-burned treatments as well. Bradley et al. (2006) suggested that over time, most small mammal populations decrease during severe periods as needed resources become scarce. It is therefore likely that species in my study are responding to the prolonged drought effects rather than the burn effects. And no treatment effect was detected for the individual comparisons for the abundance of the northern pygmy mouse and the plains harvest mouse in the late season of sampling. These results shed light on how dramatic the effects of drought can be on ecosystems and the varied responses by species in these habitats.

Since the initiation of this project, abundance, diversity, richness, and evenness of small mammal species has significantly declined. Individual comparisons for abundant species have also declined. I believe this suggests that populations of small mammals have negatively responded to the onset of drought conditions within the study area. However, interpretation has been made difficult because of the long-term severe drought that has persisted on the site over the last few years. Continued monitoring of these populations should elucidate treatment effects that are currently masked by drought effects, particularly if rainfall patterns and range conditions improve. Similar results were also reported in many other studies, in that, population size is correlated with habitat and food availability and these factors, in turn, are correlated with adequate precipitation

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amounts (Mutze et al. 1991, Brown and Ernest 2002, Bradley et al. 2006)

Poole (2009) found that prescribed burning had little effect on small mammal captures, species richness, diversity, and evenness, which was consistent with my results. His results from both Sherman live trapping and drift fence arrays indicated that abundance of individuals, species richness, evenness, and diversity declined over the two years of sampling (Poole 2009). Poole (2009) found that abundance of northern pygmy mouse and plains harvest mouse decreased between 2005 and 2006. This was consistent with my results because abundance of plains harvest mouse decreased over time in the unburned treatments and tended to decrease in the summer and winter burned treatments in the early season of sampling. However, I found that northern pygmy mouse abundance did not differ among treatments in the early and late seasons and no treatment effect was detected for the individual comparisons for both species in the long-term late season of sampling. Overall, it is likely that the abundance and diversity of small mammals has been declining since the initiation of the project because of the onset of below average precipitation patterns observed.

Long-term studies are regarded as highly indispensable because the ability to detect potential changes in the populations and communities requires observations during variable abiotic factors, such as climactic trends over the longer term. Thus, the natural causes and effects of variability in ecosystems can only be understood through extended studies over longer periods of time than most other studies are typically conducted (Cody and Smallwood 1996). These ecosystem changes in the presence of long-term climactic trends can be seen when examining Poole's (2009) short-term data versus my long-term data, which justifies the need for such studies. With long-term research, the ability to resolve short – term phenomena is enhanced while the ability to discover and address longer-term phenomena is revealed (Cody and Smallwood 1996).

Management Implications

The Rolling Plains Ecoregion is dominated by vast areas of mesquite grasslands and is characterized by variable weather patterns and drought periods. Little data exists on the role of seasonal prescribed burns on small vertebrate species communities in this ecosystem. Therefore, the long-term effects of seasonal prescribed fire on vegetation and small vertebrates can most likely depend on the highly variable climate conditions of this region. Summer prescribed burning appears to be more effective at increasing community metrics for small vertebrates, but both treatments are beneficial to vegetation and small vertebrate communities. However, persisting droughts may negate any positive effects and increase negative results. My results indicated that in general, summer-burning seems to have a greater positive benefit on communities and individual species populations relative to lack of burning. Treatment did not generally alter community metrics for herbaceous vegetation. However, comparison of early season indices of similarity demonstrated that presence of burning shifted the species composition of the herbaceous plant community over time. Therefore, burning is shifting the plant community regardless of weather patterns, but it may be ameliorating the effects of long-term drought on the small vertebrate community. Therefore a combination of both burning treatments is recommended to suppress woody vegetation, and increase vegetation productivity for wildlife and livestock use. Continued monitoring of these populations should elucidate

treatment effects that are currently masked by drought effects, particularly if range conditions improve.

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during the $2004 - 2009$ and 20	18 - 2019 years of sampling.				
Common Name	Scientific Name	Summer burned	Winter burned	Unburned	Total
Western narrowmouth toad	Gastrophryne olivacea	199	146	181	526
Six-lined racerunner	Cnemidophorus sexlineatus	95	100	106	301
Great Plains skink	Eumeces obsoletus	57	74	80	211
Prairie lizard	Sceloporus undulates	36	38	67	141
Plains threadsnake	Leptotyphlops dulcis	34	44	31	109
Green toad	Bufo debilis	39	26	37	102
Plains spadefoot toad	Spea bombifrons	35	19	×	62
Plains black-headed snake	Tantilla nigriceps	4	15	S	24
Texas toad	Bufo speciosus	5	8	ω	16
Woodhouse's toad	Anaxyrus woodhousii	5	${\bf \omega}$	L	15
Texas whiptail	Cnemidophorus gularis	5	L	7	14

Table II.1. Number of individuals captured for herpetofauna using drift fence arrays at Matador WMA in Cottle County, Texas

Table II.1, con't. Number of in	dividuals captured for herpeto	fauna using drift fen	ce arrays at Matad	lor WMA in Co	ottle County,
Texas during the $2004 - 2009$ i	and 2018 – 2019 years of sam	oling.			
Common Name	Scientific Name	Summer burned	Winter burned	Unburned	Total
Texas horned lizard	Phrynosma cornutum	4	3	4	11
Ground snake	Sonora semiannulata	4	2	S	11
Couch's Spadefoot	Scaphiopus couchi	4	4	1	6
Ornate box turtle	Terrapene ornata	L	1	0	8
Western tiger salamander	Ambystoma mavortium	\mathfrak{c}	2	7	7
Coachwhip	Masticophis flagellum	1	3	7	9
Night snake	Hypsiglena torquata	0	0	7	7
Western hognose snake	Heterodon nasicus	0	0	1	1
Milk snake	Lampropeltis triangulum	0	0	1	1
Long-nosed snake	Rhinocheilus lecontei	0	1	0	1
Glossy snake	Arizona elegans	1	2	2	5
Plains leopard frog	Lithobates blairi	1	0	0	1
	Total	539	498	547	1,584

Species	Summer	Winter	Unburned	Total
	Burned	Burned		
Northern Pygmy Mouse	159	188	156	503
Plains Harvest Mouse	89	101	102	292
Least Shrew	16	13	21	50
Merriam's Pocket Mouse	11	5	5	21
House Mouse	0	1	4	5
Plains Pocket Gopher	3	1	1	5
Total	278	309	289	876

Table II.2. Number of individuals of each species captured using drift fence arrays at Matador WMA in Cottle County, Texas during 2005 – 2009 and 2018 – 2019.

Table II.3. Season, treatment, r^2 , and p-values derived from a regression between Great Plains skink abundance and sample year on the Matador WMA in Cottle County, Texas from 2004 – 2009 and 2018 – 2019.

Season	Treatment	\mathbf{r}^2	p-value
Early	Unburned	0.142	0.404
	Winter burned	0.365	0.151
	Summer burned	0.171	0.356
Late	Unburned	0.023	0.963
	Winter burned	0.022	0.749
	Summer burned	<0.001	0.963

Table II.4. Season, treatment, r^2 , and p-values derived from a regression between plains threadsnake abundance and sample year on the Matador WMA in Cottle County, Texas from 2004 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.167	0.363
	Winter burned	0.061	0.593
	Summer burned	0.039	0.672
Late	Unburned	0.036	0.684
	Winter burned	0.748	0.012
	Summer burned	0.139	0.410

Table II.5. Season, treatment, r^2 , and p-values derived from a regression between prairie lizard abundance and sample year on the Matador WMA in Cottle County, Texas from 2004 – 2009 and 2018 – 2019.

Season	Treatment	\mathbf{r}^2	p-value
Early	Unburned	0.264	0.238
	Winter burned	0.183	0.339
	Summer burned	0.026	0.729
Late	Unburned	0.212	0.298
	Winter burned	0.125	0.436
	Summer burned	0.221	0.288

Table II.6. Season, treatment, r^2 , and p-values derived from a regression between western narrowmouth toad abundance and sample year on the Matador WMA in Cottle County Texas from 2004 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.001	0.950
	Winter burned	0.263	0.240
	Summer burned	0.441	0.104
Late	Unburned	0.070	0.565
	Winter burned	0.069	0.570
	Summer burned	0.067	0.576

Table II.7. Season, treatment, r^2 , and p-values derived from a regression between sixlined racerunner abundance and sample year on the Matador WMA in Cottle County Texas from 2004 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.716	0.016
	Winter burned	0.350	0.161
	Summer burned	0.845	0.003
Late	Unburned	0.777	0.009
	Winter burned	0.736	0.014
	Summer burned	0.903	0.001

Table II.8. Season, treatment, r^2 , and p-values derived from a regression between northern pygmy mouse abundance and sample year on the Matador WMA in Cottle County, Texas from 2005 – 2009 and 2018 – 2019.

Season	Treatment	\mathbf{r}^2	p-value
Early	Unburned	0.249	0.253
	Winter burned	0.193	0.322
	Summer burned	0.208	0.303
Late	Unburned	0.223	0.283
	Winter burned	0.242	0.261
	Summer burned	0.259	0.242

Table II.9. Season, treatment, r^2 , and p-values derived from a regression between plains harvest mouse abundance and sample year on the Matador WMA in Cottle County, Texas from 2005 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.482	0.083
	Winter burned	0.419	0.115
	Summer burned	0.339	0.169
Late	Unburned	0.214	0.295
	Winter burned	0.261	0.241
	Summer burned	0.174	0.350

Table II.10. Season, treatment, r^2 , and p-values derived from a regression between small vertebrate species richness and sample year on the Matador WMA in Cottle County, Texas from 2004 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.179	0.344
	Winter burned	0.499	0.076
	Summer burned	0.001	0.962
Late	Unburned	0.220	0.289
	Winter burned	0.046	0.645
	Summer burned	0.008	0.850

Table II.11. Season, treatment, r^2 , and p-values derived from a regression between small vertebrate species evenness and sample year on the Matador WMA in Cottle County, Texas from 2004 – 2009 and 2018 - 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.271	0.231
	Winter burned	0.002	0.922
	Summer burned	0.010	0.830
Late	Unburned	0.054	0.617
	Winter burned	0.086	0.523
	Summer burned	0.262	0.240

Table II.12. Season, treatment, r², and p-values derived from a regression between small vertebrate species Shannon – Weiner diversity indices and sample year on the Matador WMA in Cottle County, Texas from 2004- 2009 and 2018 - 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.183	0.338
	Winter burned	0.500	0.076
	Summer burned	0.001	0.961
Late	Unburned	0.345	0.166
	Winter burned	0.001	0.959
	Summer burned	0.053	0.619

Table II.13. Season, treatment, r², and p-values derived from a regression between small vertebrate species Modified Simpson's diversity indices and sample year on the Matador WMA in Cottle County, Texas from 2004- 2009 and 2018 - 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.231	0.275
	Winter burned	0.368	0.149
	Summer burned	0.005	0.885
Late	Unburned	0.228	0.279
	Winter burned	0.016	0.785
	Summer burned	0.003	0.902

Table II.14. Season, treatment, r^2 , and p-values derived from a regression between small vertebrate species abundance and sample year on the Matador WMA in Cottle County, Texas from 2004 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.254	0.249
	Winter burned	0.248	0.256
	Summer burned	0.093	0.506
Late	Unburned	0.096	0.499
	Winter burned	0.110	0.468
	Summer burned	0.099	0.493

Table II.15. Season, Treatment, r², and p-values derived from a small vertebrate regression between Jaccard's index of similarity to 2019 and sample year on the Matador WMA in Cottle County, Texas 2004 – 2009 and 2018 – 2019.

Season	Treatment	r^2	p-value
Early	Unburned	0.034	0.726
	Winter burned	0.475	0.130
	Summer burned	0.536	0.098
Late	Unburned	0.724	0.032
	Winter burned	0.115	0.510
	Summer burned	0.093	0.558



Figure II.1. Blowout map of my study site indicating the location of Cottle County within Texas (A), the Matador Wildlife Management Area within Cottle County (B), Headquarters Pasture within the Matador WMA (C), and the design of the study plots within Headquarters Pasture (D).



Figure II.2. Diagram of my specific study plots within Headquarters Pasture on the Matador WMA within Cottle County, Texas. Headquarters pasture was divided into a 3 x 5 randomized complete block design; each block containing 3 -18 ha plots, resulting in 15 total plots. Each plot within a block was randomly assigned 1 of 3 treatments: summer burned (S), winter burned (W), and unburned (C).

35 Traps 36 Traps	35 Traps 	↑ N
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Figure II.3. Diagram of Sherman live trap transect sampling for small mammals within each plot. Three 200-m north and south running parallel transects 100-m apart consisted of 106 traps. Each trap within a transect was approximately 7 paces apart with each trap's opening facing north.


Figure II.4. Early season regression of six-lined racerunner abundance sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Summer – burned treatment (black line) and unburned treatment (hollow line) were significant. Winter burned treatment (gray dotted line) was approaching significance.



Figure II.5. Early season regression of plains harvest mouse frequency sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Unburned treatment (hollow line) was significant.
Summer-burned (black dotted line) and winter-burned treatment (gray dotted line) were approaching significance.



Figure II.5. Early season regression of small vertebrate richness sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Winter-burned treatment (gray line) was significant. However, summer-burned treatment and unburned treatment were not significant so no regression is presented.



Figure II.6. Early season regression of small vertebrate evenness sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Summer-burned , winter-burned treatment, and unburned treatment were not significant so no trend is presented.



Figure II.7. Early season regression of small vertebrate Shannon's diversity sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County,
Texas 2005 – 2009 and 2018 – 2019. Winter-burned treatment (gray line) was significant. However, summer-burned treatment and unburned treatment were not significant so no regression is presented.



Figure II.8. Early season regression of small vertebrate abundance sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Unburned treatment (hollow line) was significant.
Summer-burned treatment, winter-burned treatment, and unburned treatement were not significant so no trend is presented



Figure II.9. Early season regression of Jaccard's index of similarity to 2019 sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County,
Texas 2005 – 2009 and 2018 – 2019. Summer-burned treatment (solid black line) was significant. Winter-burned treatment (dotted gray line) was approaching significance, and unburned treatement was not significant so no trend is presented



Figure II.10. Late season regression of six-lined racerunner abundance sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2004 – 2009 and 2018 – 2019. Summer burned treatment (black line), winter burned treatment (gray line), and unburned treatment (hollow line) were significant.



Figure II.11. Late season regression of plains threadsnake abundance sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2004 – 2009 and 2018 – 2019. Winter burned treatment (gray line) was significant. However, summer burned treatment and unburned treatment were not significant so no regression is presented.



Figure II.12. Late season regression of small vertebrate richness sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2004 – 2009 and 2018 – 2019. Summer- burned treatment, winter – burned treatment, and unburned treatment was not significant so no regression is presented.



Figure II.13. Late season regression of small vertebrate evenness sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2004 – 2009 and 2018 – 2019. Summer-burned, winter-burned, and unburned treatments were not significant so no regression is presented.



Figure II.14. Late season regression of small vertebrate Shannon's diversity sampled via drift fence arrays at the Matador Wildlife Management Area in Cottle County,
Texas 2004 – 2009 and 2018 – 2019. Summer-burned, winter-burned, and unburned treatments were not significant so no regression is presented.



Figure II.15. Late season regression of Jaccard's index of similarity to 2019 sampled via

drift fence arrays at the Matador Wildlife Management Area in Cottle County, Texas 2005 – 2009 and 2018 – 2019. Unburned treatment (hollow line) was significant. However, summer-burned and winter-burned treatment were not significant so no regression is presented.