MAMMALIAN COMMUNITY RESPONSES TO TIME SINCE BURNING ON A MANAGED ROLLING PLAINS LANDSCAPE

by

Erin Stiede

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ABSTRACT

Fire is a natural disturbance in the Rolling Plains ecosystem. When applied properly, fire removes unwanted litter, decreases woody plant densities, and increases herbaceous plant production. Plant abundance, density, and biomass will differ by time since an area burns. Animals utilize plants for resources such as diet and cover. Therefore, it is expected that animals will use habitats differently depending on the time since a burn has occurred. This has the potential to help biologists and landowners to more efficiently manage native species. My objective was to look at the influence of time since burning on the activity of mammals. I used camera traps in 16 different locations with 0-15 years post-burn on Matador Wildlife Management Area in Cottle County, Texas, in 2018-2020. Cameras were spaced throughout the property in uplands, lowlands, and drainages. I calculated an activity index for each species for each month of data. Each species activity indices were then used to calculate Shannon Diversity Index in activity, total activity, and species evenness in activity. Species richness was also calculated per month per camera location. I then regressed the 5 most abundant species' activity indices against days since burning. Community metrics were also regressed against days since burning. During the duration of the study 919,884 images were collected over 8,498 camera days. I detected 17 mammal species on the property. The 5 most abundant species were coyote (*Canis latrans*), wild boar (*Sus scrofa*), white-tailed deer (*Odocoileus virginianus*), mule deer (*Odocoileus hemionus*), and cottontails (*Sylvilagus* spp.). Coyotes and wild boar activities were not statistically significantly influenced by days since burning. White-tailed deer increased in activity in areas with an increase in days since burning. Mule deer also increased in activities as days since burning increased. Cottontail activity decreased as days since burning increased. Total mammal activity increased as days since burning increased. Shannon Diversity Index for activity and species evenness of activity both decreased as days since burning increased. Finally, there was no statistical significance between days since burning and species richness. In conclusion, in the short-term, prescribed fire influenced mammalian activity on a managed Rolling Plains landscape. I suspect prolonged drought influenced the ability to detect clearer patterns. In conclusion, there is evidence that suggests species are using both recently burned areas and areas that have gone numerous years of growing seasons post-fire. This indicates a need for a mosaic management plan to best manage the entire community.

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Approved:

Dr. Richard T. Kazmaier Chairman, Thesis Committee	Date
Dr. James B. Johnson Member, Thesis Committee	Date
Brad Simpson Member, Thesis Committee	Date
Head, Department of Life, Earth, and Environmental Sciences	Date
Dean, College of Agriculture And Natural Sciences	Date
Dean, Graduate School	Date

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CHAPTER I

MAMMALIAN COMMUNITY RESPONSES TO TIME SINCE BURNING ON A MANAGED ROLLING PLAINS LANDSCAPE

INTRODUCTION

Fire is a natural and important disturbance in many ecosystems globally. Widespread prairie fires occurred naturally in the Great Plains of North America because of factors such as topography, climate, and vegetation type (Jackson 1965). In Serengeti National Park, fire played a major role in shaping grasslands and savannas (Eby et al. 2014). Fire-induced heterogeneity on the landscapes of southeast Australia affects different components of biota (Chia et al. 2015). Thus, fire is an integral part of ecosystems for wildlife, including plant and animal species. Likewise, suitable habitat with adequate food resources and concealment cover are renewed by fire (Smith 2000, Cherry et al. 2018).

Early settlers in the Great Plains noted violent lightning storms causing prairie fires (Jackson 1965, Drewa et al. 2001, Cherry et al. 2018). Natural wildfires were formally the major method in controlling brush in the region (Blydenstein 1957). However, after human settlement, fire suppression became the standard and woody species, such as honey mesquite (*Prosopis glandulosa*), spread because of decreased fire intensity and time between burns (Ansley et al. 2015). Fire suppression is defined as the area that did not burned because the fire was extinguished (Baker 1993). For a proper fire, comprehensive whole-system responses are indicative to avoid woody species spread, exotic establishment, and succession (Pinchak et al. 2010, Pastro et al. 2011). It is extremely hard to predict how far a fire would carry naturally before fire suppression because of immediate and delayed responses (Baker 1993). For example, in the Great Plains, the only historic firebreaks present were rivers, meaning a single ignition could have traveled for hundreds of km (Jackson 1965). Additionally, fires burned unevenly, thus creating spatial heterogeneity leaving refuge patches for wildlife species (Bond et al. 2002, Chia et al. 2015, Valkó et al. 2016, Hill et al. 2017). For example, white-tailed deer (*Odocoileus virginianus*) in southern Alabama used streambeds and other moist sites as refuges from fire (Ivey and Causey 1984).

Anthropogenic actions have influenced the probability of fires and spread, by influencing ignitions, fighting fire spread, and modifying fuel load (Ruffault and Mouillot 2015). Thus, fire suppression and prevention practices have led to a sharp decrease in fire activity, leading to the need for managers and landowners to prescribe fire. Prescribed fire is different from wildfire because it is conducted under predetermined abiotic and biotic conditions to accomplish a management goal or objective (Texas Parks and Wildlife Department 2011). In the Rolling Plains ecoregion of Texas, prescribed fire has been used by landowners and managers for over 70 years for the ongoing war with woody plants. Landowners' original goal was to kill all trees on the rangeland, but prairie trees, such as honey mesquite, are fire-adapted species that can often resprout after being top-killed by fire (Ansley and Jacoby 1998). Therefore, if prescribed appropriately, burning is a well-suited tool to maintain these woody species in habitats such as mesquite savannahs, while benefiting wildlife species as well.

Long-term fire suppression modeling clearly shows the need for spatially explicit prescribed burns to manage landscapes (Baker 1993). Woody encroachment is one objective prescribed fire can confront. Woody encroachment is defined as a process that includes the recruitment of new species and the expansion in cover of existing shrubs or trees (Heisler et al. 2003). Grassland ecosystems worldwide have experienced major dominate species shifts as woody plant species have expanded (Heisler et al. 2003). Recruitment of these woody species have been increasing with an increased time between fires or low fire frequency rates (Heisler et al. 2003). Reduction in fire frequency along with grazing are the proximate causes of grassland species shifts, along with atmospheric carbon dioxide enrichment, climate change, and nitrogen deposition (Heisler et al. 2003, Allred et al. 2011). High fire frequency does not always eliminate woody vegetation but can prevent recruitment of new species and expansion of existing species (Heisler et al. 2003).

Fire severity is also a key component to be considered when prescribing a burn, with important landscape features such as cover type and pre-fire fuel loads (Cherry et al. 2018). Also, fuel moisture content and fuel continuity during burning can affect fire intensity, severity, and spread (Hill et al. 2017). For example, if you want a fire with low burn severity it is recommended to burn in the morning with cool and humid conditions (Hill et al. 2017).

Other means of brush control such as mechanical (for example, aeration and grubbing) or applying herbicides can be used to decrease woody vegetation for goals such as favoring herbaceous vegetation. Although these practices can produce the same results as burning, they are generally more expensive and require heavy machinery (Rogers et al. 2004, Ruthven et al. 2008). For example, in aeration treatment plots in southern Texas brushlands, maintenance by fire versus aeration did not differ in the results of habitat characteristics that benefited white-tailed deer density (Rogers et al. 2004). Therefore, prescribed burning can be the most economical means of decreasing woody encroachment, and maintaining the natural landscape (Dills 1970, Sharrow and Wright 1977, Wood 1988, Teague et al. 2008, Hill et al. 2017).

Climate change and weather variability can have unpredictable effects on fire frequency and intensity of wildfire (Rogers et al. 2004, Chia et al. 2015). Thus, with prescribed fire, unusual post-fire weather patterns have the potential to impact management goals and alter the landscape. In western Texas, grass recovery post-fire is determined by rain (Wright 1974). During dry years, soil moisture is the limiting factor for plants, and burning may have no beneficial effect during this time (Sharrow and Wright 1977). For example, sideoats grama (*Bouteloua curtipendula*) is typically harmed after fire unless rainfall is adequate or above average for that year, in western Texas (Wright 1974).

The primary advantage of proper prescribed burning is for the benefit of plant communities. The effect a fire has on a community is dependent on the amount of structural change in vegetation (Smith 2000). Productivity of a rangeland goes down over time since burning was applied, as litter and dead vegetation accumulate (Vinton and Collins 1997). This stresses the need for appropriate fire return intervals to maintain a particular woody structure. Fire return intervals are determined by the type of ecosystem and function of the landscape, modeling as close to natural return rates for maintenance, and more aggressive or targeted during key plant life stages for maximum removal (Ansley et al. 2015). Post-fire, tobosagrass (*Hilaria mutica*) and other herbs reached equilibrium, and the same measurements as the controls, by the 5th growing season in the Rolling Plains of Texas (Neuenschwander et al. 1978). Too frequent burning may deplete soil nitrogen and reduce future plant growth (Sharrow and Wright 1977). In the Rolling Plains of Texas, fire alone did not reduce prickly pear (*Opuntia* spp.) canopy cover (Blair et al. 1993). Thus, fire plus herbicide or other management actions are commonly used to enhance or prolong effectiveness of woody plant elimination (Lochmiller et al. 1991, Blair et al. 1993).

Prescribed fire has the potential to benefit plant species. Inorganic nitrogen levels elevated in burned areas immediately post-fire in an area where fire suppression was prevalent (Ojima et al. 1994). Fire enhanced production of lower quality root material which enhanced root nitrogen immobilization (Ojima et al. 1994). In an area 2 months post-fire, there were higher amounts of copper, potassium, magnesium, and nitrogen in green leaf material relative to unburned areas, which made burned habitat preferable to herbivores (Eby et al. 2014). In southern mixed prairie grasslands, little bluestem (*Schizachyrium scoparium*) doubled in production the first year after burning (Wright 1974). Additionally, plant species richness increased significantly in burned plots during both dormant season fires and summer fires in New Mexico shortgrass prairies (Brockway et al. 2002).

Prescribed fire also has the potential to directly and indirectly affect animal species. Prescribed fires decrease the likelihood of catastrophic fires and improve habitat quality for wildlife and livestock (Main and Richardson 2002, Pastro et al. 2011). With fire being a natural disturbance, many species do not seem alarmed by it. White-tailed deer have been seen feeding 20 m from fire with no alarm (Ivey and Causey 1984). Raccoons (*Procyon lotor*) in east-central Minnesota did not change activity level in an area even after a part of it was recently burned (Sunquist 1967). Fire may have an impact on animal behavior, but mostly effects food resources and cover availability (Main and Richardson 2002, Allred et al. 2011).

Benefits arise in burned and unburned areas on the landscape. Fire reduces vegetation height so prey can see predators better and have a better chance at getting away (Eby et al. 2014). However, predators, like coyotes (*Canis latrans*), have selected for recently burned areas where prey increased and may be easier to see and chase (Stevenson et al. 2019). Fires in Blackland Prairie directly affected small mammals by forced emigration, reduced reproduction effort, and high mortality (Kirchner et al. 2011). Animals have been seen avoiding harshly burnt areas, meaning fire severity has a strong influence on animal use of an area (Ivey and Causey 1984, Chia et al. 2015). In general, prescribed burns are generally considered to improve wildlife habitat (Hobbs and Spowart 1984).

Archibald et al. (2005) defined “magnet effect” whereby herbivores area actively attracted to recently burned areas for foraging. This has been supported by many other studies. For example, white-tailed deer expanded their home ranges into areas that were recently burned in a tropical pyric landscape (Cherry et al. 2018). Likewise, white-tailed deer and mule deer (*Odocoileus hemionus*) both increased their activity in burned areas compared to unburned (Davis 1967, Vogl and Beck 1970, Klinger et al. 1989, Main and Richardson 2002). American bison (*Bison bison*) in the Tallgrass Prairie Preserve of Oklahoma also exhibited the magnet effect using recently burned areas preferentially (Allred et al. 2011). The magnet effect is supported by other studies concluding that burned areas had an increased nutritional quality for herbivores (Eby et al. 2014): mountain sheep (*Ovis canadensis*) and mule deer (Hobbs and Spowart 1984), white-tailed deer (Dills 1970, Ivey and Causey 1984, Masters et al. 1993, Mixon et al. 2009), and collared peccaries (*Pecari tajacu*; O’Brien et al. 2005).

Food is not always the driving resource in post-fire selection. In longleaf pine (*Pinus palustris*) forests of North Carolina, lactating female white-tailed deer selected unburned and over 1-year post burned areas for more cover for their young to avoid predation, regardless of the better browse in the recently burned areas (Lashley et al. 2015). Animals need different types of cover for different seasonal and daily requirements. White-tailed deer used closed vegetation during the day and open vegetation at night (Beier and McCullough 1990). Collared peccary bedding sites had more thermal cover versus their foraging site, because of thermal regulation and predator avoidance (O’Brien et al. 2005). Forests provide cover for white-tailed deer fawns while adult does are foraging (Augustine et al. 1998). In the Alaskan Taiga, snowshoe hares (*Lepus americanus*) and lynx (*Lynx canadensis*) increased their abundance in burned areas, however snowshoe hares also selected for areas of refugia in the unburned areas (Paragi et al. 1997).

Fire does not benefit all animal species. Fire had no difference in mule deer fawn survival or population size in the chaparral (Klinger et al. 1989). Pygmy mice (*Baiomys taylori*) and cotton rats (*Sigmodon hispidus*) had negative responses to fire by decreasing in abundance in a tallgrass blackland prairie (Kirchner et al. 2011). Cottontail rabbits (*Sylvilagus floridanus*) were negatively affected by fire from increased avian predation compared to unburned sites in east-central Mississippi (Bond et al. 2002). Fire had a negative influence on vegetation visual obstruction on the dung beetle assemblage in southern Great Plains (Smith et al. 2019). In the short-term, winter and/or summer burns had little effect on herpetofauna diversity at a site in southern Texas (Ruthven et al. 2008).

In conclusion, fire is a natural and important disturbance for many ecosystems. Understanding responses of plants and animals in relation to burns can help managers and landowners more effectively manage native species. It is important to look at mammal activity following the time since an area was burned and the possible cause and effect for management purposes. I hypothesized that mammal activity would increase in diversity, richness, abundance, and evenness following a fire and decrease in that area in the years following the burn.

STUDY AREA

This project was conducted at Matador Wildlife Management Area in Cottle County, Texas. It is located about 43 km south of the town of Childress and 15.9 km north of Paducah (Figure 1). This property was purchased by the state of Texas in 1959 with Pittman-Robertson funds and is managed by the Wildlife Division of Texas Parks and Wildlife Department. This property contains 11,406 ha of managed property for research and demonstration purposes. Matador Wildlife Management Area also hosts many recreational activities such as birding, public youth shoots, fishing, and hunting. The property is continually active in hosting numerous hunts open to the public which include species such as white-tailed deer, mule deer, and wild turkey (*Meleagris gallopavo*).

This property is in the Rolling Plains ecoregion, containing habitats such as shinnery oak (*Quercus havardii*) sandy soil uplands, honey mesquite lowlands and uplands, upland breaks dominated red-berry juniper (*Juniperus pinchotii*) with shallow limestone or gypsum soils, and bottomland floodplains of the Middle Pease River (Ferguson et al. 2015, Romo 2018, Poole 2009). Typically, Matador Wildlife Management Area has dry winters and hot summers with an average annual precipitation of about 56 cm (Richardson et al. 1974).

Matador Wildlife Management Area, being a research and demonstration area, is involved in using practices and tools to maintain and restore wildlife habitat. The managers conduct different rangeland improvement actions to share with local landowners. These may include herbicide treatment on honey mesquite and prickly pear

(*Opuntia* spp.) to mechanical brush manipulation like grubbing, aeration, and manual tree removal. Matador Wildlife Management Area also rotationally grazes cattle in most of their pastures.

There also is an active prescribed burning program on the property. The property is split into 14 major pastures and several pastures are typically burned in either summer or winter every year. The average fire return interval in any specific pasture is about 4 years. They also have a few small areas that are unburned as demonstration controls.

METHODS

Sixteen different locations were selected within Matador Wildlife Management Area with a fire interval from 0-15 years post burning (Figure 2). A trail camera was set up at each location secured to a plywood stand or tree. Deployed cameras originally were Bushnell Trophy cameras (Bushnell Corp., Overland Park, KS, USA) and were replaced by Browning Elite Trail cameras (Browning Trail Cameras, Prometheus Group, LLC. Birmingham, AL, USA) in the second year of the study. The cameras were replaced for a more effective model type, and evenly distributed when changing camera types. Plywood stands measured 61 cm tall and 30.5 cm wide. The stands were secured into a “Z-shaped” figure with metal straps (Figure 3). They consisted of 2-horizontal 30.5 cm by 30.5 cm square sections attached to a 61-cm-long vertically aligned section, creating a base and top cover that shaded the camera (Cancellare 2018). Cameras on both stands and trees were secured in a metal security box (Browning Trail Cameras, Prometheus Group, LLC.

Birmingham, AL, USA or CAMLOCKbox, Green Bay, WI, USA), for best short-term stability. Then they were locked with a keyed lock or cable lock to prevent theft.

Sixteen camera traps were stratified between lowlands, uplands, and drainages with different fire frequencies (Figure 4). Camera traps are an effective tool for generating detections of species, especially in studies with a large number of species (Wearn and Glover-Kapfer 2019). Camera traps have also been used in wildlife studies looking at influences of stages post-fire (Main and Richardson 2002). Additionally, game cameras have been used to study fires and monitor fire conditions (Ridenour and Gray 2010). Cameras were placed along game trails to maximize chances of detecting species in the area. Cameras were pointed north when possible to reduce glare and enhance animal identification. Cameras were placed at least 200 m from the nearest road or two-track, and they were placed at least 500 m from another camera.

Each location's Global Positioning System coordinate was recorded with a Garmin handheld unit (Garmin Ltd., Olathe, KS, USA) and averaged to produce an error rate of <5 m for each location. The sites were manipulated with hand loppers and a weed eater to clear the vegetation around the camera to minimize false triggers. Cameras were set up to take 3 pictures rapid-fire creating an event, followed by a 10-second interval before being able to trigger again. Cameras were checked every couple of months to replenish batteries and replace SD cards. Although cameras were placed on game trails, they were unbaited.

The photos were then sorted through Daminion (Daminion Software, Pacific Business Centre, Seattle, WA, USA), a network-based digital photo manager to produce an automated catalogue for data entry. Then the data from the photos were organized in Microsoft Excel (Microsoft Corporation, Redmond, WA, USA) by filename, time, and date for each camera and time period. For each camera, every image was scored for how many species and individuals were in the photo. Each image was represented by a row: camera number, filename, date, and time. Each mammal species that was encountered was given a column. For each mammal species that was detected in an image, the number of individuals of that species on each image was recorded under its respective column. Photos of mice and bats did not generally consist of enough characteristics to be identified to species, so all mice species were categorized into “mouse” and all bat species were considered “bat”. Photos of cottontails (*Sylvilagus* spp.) and woodrats (*Neotoma* spp.) did not generally consist of enough species characteristics, so they were identified to genera. If a mammal was detected, but not able to be identified properly it was scored “unknown”. An image containing an animal not of interest was scored in a “miscellaneous” column and identified. For example, reptiles, birds, and amphibians were not of interest to this study so they were identified as “miscellaneous.” Additionally, cows and humans were not included in the analysis, so they were also classified as “miscellaneous.” Domestic cattle and humans were removed, because I was interested in the activity of only wild species. Finally, if nothing was in the picture, such as the camera was triggered by vegetation, the image was scored as a “clear”.

Linear regression was used to look at the relationship to activity of mammals to time since burning. For this analysis, the images needed to be condensed into events by selecting 1 photo out of the 3 from each event that had the most species and individuals of mammals present. Then, the events were condensed into an activity index. Meaning every species per image for every month and camera was converted into an activity index using a specific formula. This value is meant to represent the importance or level of activity of a specific camera location for mammals. An activity index was needed to do this analysis because individuals were not able to be identified by the photos. Additionally, not all cameras were operating during the entire month, so number of cameras days was also incorporated into the activity index to standardize effort. The activity index equation for each species for each month was:

$$Activity\ Index = \left[\frac{E}{C}\ x\ I\right]\ x\ 10$$

Where, E= Number of Events a Species was Detected

C= Camera Days

I = Average Number of Individuals of the Species per Event

With the activity index, linear regression analyses were conducted for the activity of the most abundant mammal species on the site against days since burned. I selected the top 5 species that composed most of the events recorded.

I also examined community responses. To do this, each species activity indices were then used to calculate a Shannon Diversity Index ($H' = -\sum pi \ln pi$; Magurran 2004) in activity for each month. Total activity, species richness (S; derived from the number of species in each event per month), and species evenness $\left[H_{max} = \frac{H'}{\ln(S)}\right]$ in activities were also calculated (Magurran 2004). Images categorized as “unknown” were excluded from analysis. For each month, I calculated the number of days since burn, by subtracting the day of the most recent burn for that area by the mid-date of each month.

Then these activity indices were regressed against days since burned. For each camera, months recording less than 20 days out of the month were excluded from analysis. Additionally, we excluded months with a result of zero for our variables. Finally, since this project was set up to explore the trends and relationships of these variables, I set alpha = 0.1.

RESULTS

Cameras were active for 2 years from February 2018 - March 2020. During the duration of the study 919,884 images were collected over 8,498 camera days. Total images consisted of 90.2% "clear" triggers, 5.8% mammals of interest, 4% "miscellaneous" species, and only 0.01% of the total images captured (106 images) were labeled unknown specimens. The two most numerous reasons an image was scored as clear were from moving vegetation 87.7 % and cows knocking over the camera or pushing it out of the field of view 11.1%. For miscellaneous images captured, domestic cattle had the highest amount consisting of 76.8% for animals not of interest. Other miscellaneous specimens encompassed birds, reptiles, and insects contributing for 21.4% of the miscellaneous scored images.

Seventeen species of wild mammals were detected on camera during the study period (Table 1). There were 5 species that accounted for a total of 90% of the overall activity on the cameras: coyote (6.6%), wild boar (*Sus scrofa;* 12.5%), white-tailed deer (54.2%), mule deer (9.3%), and cottontails (6.9%).

Coyotes were detected on all 16 cameras at one point in time during the study. Coyotes occurred 67% of the months per active cameras. Coyote events per month ranged from 0 – 348 events. Coyote activity ($F_{184} = 0.101$, p = 0.751, $r^2 < 0.001$) was not statistically related to days since burning (Figure 5).

Wild boars were detected on all the cameras at one point in time during the study. Wild boars occurred 38% of the time while the cameras were active. Wild boar events per month ranged from 0 – 744 events. Wild boar activities ($F_{104} = 1.939$, p = 0.167, $r^2 = 0.018$) were not statistically related influenced by days since burning (Figure 6).

White-tailed deer were detected on all the cameras at one point in time during the study. White-tailed deer occurred 88.4% of the time while the cameras were active. This species ranged from 0 – 1286 events per month during the time the cameras were active. White-tailed deer increased in activity ($F_{243} = 2.874$, p = 0.091, $r^2 = 0.012$) in areas with an increase in days since burning (Figure 7).

Mule deer were detected on 15 out of the 16 cameras at one point in time during the study. Mule deer occurred 42.4% of the time while the cameras were active. Mule deer ranged from 0 – 186 events per month during the time the cameras were active. Mule deer increased in activity ($F_{116} = 15.999$, p < 0.001, $r^2 = 0.122$) as days since burning increased (Figure 8).

Cottontails were detected in 13 out of the 16 camera locations at one point in time during the study. Cottontails occurred 22.1% of the time while the cameras were active. They also ranged from 0 – 186 events per month during the time the cameras were active.

Cottontail's activity ($F_{60} = 4.461$, p = 0.039, $r^2 = 0.070$) decreased as days since burning increased (Figure 9).

Total mammal activity ($F_{269} = 3.067$, p = 0.081, $r^2 = 0.011$) was influenced by days since burning, increasing in activity as days since burning increased (Figure 10). Shannon Diversity Index for activity ($F_{247} = 9.190$, p = 0.003, $r^2 = 0.036$) decreased as days since burning increased (Figure 11). Species evenness of activity ($F_{247} = 9.049$, p = 0.003, $r^2 = 0.036$) significantly decreased as days since burning increased (Figure 12). Finally, species richness was not influenced by days since burning ($F_{269} = 1.051$, p = 0.306, $r^2 = 0.004$; Figure 13).

DISCUSSION

Fire is needed to renew suitable habitat for many wildlife species and communities (Smith 2000). Fire is especially important in a habitat such as the Rolling Plains of Texas where woody species have invaded because of suppression of naturally occurring fires. Fire maintains early successional plant communities (Bond et al. 2002). Without these fires, wildlife could lack certain resources such as cover types and food abundance. The results from my data suggest that the mammal community on this property were more active in the areas that have not been burned for a longer period. This is not uncommon; many species need different cover needs for different times of the year or day. For example, white-tailed deer used areas with more closed vegetation during the day (Beier and McCullough 1990). Collared peccaries selected for more dense cover areas because of dietary resources and predator avoidance (O'Brien et al. 2005). In

Florida however, Main and Richardson (2002) reported the opposite of a significant increase of use by total wildlife of recently burned areas. Thus, in my system, cover may be a more important factor in determining mammalian activity than nutrition is.

Mammal diversity and evenness of activity in this study were the highest in areas that were recently burned, but the predictability of these trends were weak. This suggests the trends are being driven by certain species utilizing areas for specific needs and the community response fire specific patterns are less detectable. Also, the small sample size may have influenced the capability to detect differences. When hypothesizing a taxa's response to a management application it is hard to understand the benefit/ hinderance on a large-scale, it is more important to deduce how specific species are reacting (Pastro et al. 2011).

Results for this landscape indicated white-tailed deer tended to spend most of their time in areas that had older growth or have not been burned recently. Masters et al. (1993) came to the same conclusion emphasizing that the woody browse production that increased with longer fire intervals contributed to a major portion of the deer diets in fall and winter in Oklahoma. Also supporting these findings are studies that have found white-tailed deer selecting for dense vegetation for greater cover requirements (Dills 1970, Brunjes et al. 2006). Forests provide cover for white-tailed deer fawns while adult does are foraging (Augustine et al. 1998). Cover is needed for lactating white-tailed deer does to avoid predation by coyotes of neonates (Lashley et al. 2015). However, most of the literature supports the opposite of my data that deer activity increases in recently burned areas. For example, in Pushmataha County, Oklahoma, fire increased forage quality and quantity, allowing deer diets to consist of higher quality forage (Masters et al. 1993). An area in Wisconsin, 8 years post-wildfire, still had evidence of white-tailed deer using the burned over the unburned areas (Vogl and Beck 1970). Fire is needed to maintain early successional stages of the landscape for important food resources, but those important nutritional requirements vary seasonally, with age, and with reproduction status in white-tailed deer (Masters et al. 1993, Mixon et al. 2009, Kramer et al. 2003). Even though my data suggests trends of deer activity increased in areas with a longer fire interval, the strength of the relationship is weak, and this activity of deer may not be explained solely by time since burning.

Mule deer had less activity in the recently burned areas, similar to white-tailed deer in my study. Mule deer on this landscape had a tendency to increase their activity in the area of less disturbance. In other studies, mule deer and white-tailed deer occur sympatrically in areas with considerable overlap, but segregate spatially (Brunjes et al. 2006, Dennison et al. 2016). Other research has found that white-tail deer and mule deer densities both increased in burned areas 2-3 years post burning (Dills 1970, Klinger et al. 1989). This preference for multiple years of growing season post burn may be driving the trend. Mule deer also choose areas of higher elevation than white-tailed deer (Brunjes et al. 2006). That factor may have not been sampled well enough with the broad "upland" habitat of my study. For example, in higher elevation montane plant community and grasslands, prescribed burns increased nutritional quality of winter diets of mule deer (Hobbs and Spowart 1984).

Collared peccary was an interesting species of detection on this property. In previous studies, collared peccaries used areas not burned compared to burned in Tonto National Forest, 60 km northeast of Phoenix, Arizona (O'Brien et al. 2005). In this study, collared peccaries lacked recruitment during drought, because of low quality diet (O'Brien et al. 2005). Collared peccaries were not of high detection in my study so I cannot speculate on their trends. However, it would be interesting to see what happens to the population at the Matador Wildlife Management Area because the area is active in reducing prickly pear densities with fire and herbicide. Prickly pear is the primary component of the collared peccary diet (Eddy 1961). Therefore, if this site is burning areas with prickly pear more often, this might lead to reduced collared peccary activity or occupancy.

Cottontail activity was likely to stay in open areas and habitat that was recently burned. Similarly, European rabbits (*Oryctolagus cuniculus*) colonized burnt areas after wildfire for the open habitat (Rollan and Real 2011). Likewise, herbicide and burning had a positive influence on population density of cottontail rabbits in the Cross Timbers of Oklahoma (Lochmiller et al. 1991). I found that cottontails at the Matador Wildlife Management Area had less activity in the older growth area that have gone longer without burning. This was also seen in Lochmiller et al. (1991) where they detected cottontail rabbits avoiding mature hardwood overstory and mixed brush habitat. Rollan and Real (2011) explained that woody cover hinders rabbit movement. This might be an explanation for their trends because recently burned areas also have reduced vegetation height, so predators like mammals are easier to see, and the cottontails have a better chance of getting away (Eby et al. 2014). Opposing that view, avian predation was greater in the burned treatments compared to the unburned treatments at the Black Prairie Wildlife Management Area in east-central Mississippi (Bond et al. 2002). This might indicate cottontail needing different cover types for different predators. At the Matador Wildlife Management Area, I detected 2 predators of the cottontail using my cameras, the coyote and bobcat. The area has other predators recorded at the site that were not detected in my study, such as the avian predators. Predator avoidance behavior does play a role in habitat use by prey species (Brunjes et al. 2006). However, Bond et al.'s (2002) study found predation equal in burned and unburned areas. The trends I detected in this study had weak predictability, therefore the activities of predators may be an alternate hypothesis to explore for cottontail activity.

Speculation of these activity trends may put emphasis on predator-prey dynamics. To gain insight on predator-prey interactions we need to first understand habitat selection (Dennison et al. 2016). Cover could be more important for predator avoidance for white-tailed deer and mule deer, more so than spending more time in recently burned areas to forage. There were 2 main predators detected at the site: coyotes and bobcats. Coyotes did not spend an increased amount of activity in any specific burn-type. This could mean they are generalist and go wherever the food is. Prey availability and distribution are important for the survival and land use of predators (Dennison et al. 2016). Stevenson et al. (2019) found that coyotes selected for recently burned areas where prey had increased. It is important to understand apex predators and predator-prey interactions on a property because they could indirectly affect lower tropic levels like vegetation, in addition to fire (Stevenson et al. 2019).

Weather can also play a major role when examining mammal trends. For herbivores, a dry season or drought can influence food availability (Wright 1974). The United States is projected to see an increased frequency of heat, drought, and false springs in the near future with worsening climate change conditions (Martinuzzi et al. 2016). Highly variable weather patterns have had a history in throwing off hypotheses leading to the need of long-term studies (Rogers et al. 2004). Abundance of small mammals in the short grass prairie of the Texas Panhandle were positively related to the amount of precipitation during the previous season regardless of burning type, leading to the conclusion that fire has a minor role in the short-term when compared to climate change (Priesmeyer et al. 2014).

Prolonged drought may prevent application of desired management actions like prescribed fires (Martinuzzi et al. 2016). High accounts of deer activity are associated with fair weather conditions. Therefore, deer activity is more closely related to climate variation than fire (Vogl and Beck 1970). Declines in mule deer and white-tailed deer have been associated with below average rainfall in the Sonoran Desert (Anthony 1976). Drought switched diet of mule deer and white-tailed deer to evergreen and drought-resistant forage instead of more highly preferred species (Anthony 1976). Managers and landowners need information about future weather conditions to better prepare for the worsening negative effects on the biota (Martinuzzi et al. 2016).

In conclusion, in the short-term, my studied concluded that prescribed fire has the potential to influenced mammalian activity on a managed Rolling Plains landscape. There is evidence with my results that recent fires may negatively impact total mammal activity, so frequent burning may harm the community as a whole. However, fire is needed to maintain the mesquite savannah habitat in the long-term. In other systems like the tallgrass Blackland Prairie regions, small mammals responded negatively to fire, but in the long-term fire is needed to maintain their grassland ecosystem (Bond et al. 2002, Kirchner et al. 2011). Also, certain species like the cottontail thrive in open areas and depend on fire to maintain productive rangelands.

Primary limitations of this study advocate for longer study periods than I used. However, the camera traps used were a productive tool in detecting a wider range of animals compared to more traditional methods (Espartosa et al. 2011). From 1998-2008 published work including methods of camera trapping and use of cameras as a research tool increased 50% (Rowcliffe and Carbone 2008). Cameras do have limitations with what they captured but are a growing trend in species specific and community studies.

MANAGEMENT IMPLICATIONS

Prescribed fire did influence the mammal community on a managed property in the Rolling Plains of Texas. This influence suggests that patterns of the whole community is using the areas that have gone numerous years of growing season post-fire. Also, some species are directly using the recently burned areas as well. Therefore, a mosaic management burn plan would benefit the most species in this type of habitat. For example, species like white-tailed deer and mule deer need a mosaic of open and dense vegetation in proximation to food and cover (Brunjes et al. 2006). Post-fire rangeland environments influence each species differently and by studying the long-term trends of the species, managers can better conserve native wildlife habitat.

It is also important to make sure the fire burned enough for structural changes such as litter removal and top killing. It is also beneficial to burn unevenly to leave areas of refugia in the middle of a burned area, for cover. Thus, adjustment of the fire return interval should easily allow managers to achieve the desired pattern of the landscape. However, longer term research on the effects of prescribed fire in a mesquite savanna, especially in drought years, is needed to fully evaluate the effects burning has on the wildlife population in the Rolling Plains of Texas.

LITERATURE CITED

Allred, B. W, S. D. Fuhlendorf, D. M. Engle, and R. D. Elmore. 2011. Ungulate preference for burned patches reveals strength of fire–grazing interaction. Ecology and Evolution 1: 132-144.

Ansley, R. J., B. A. Kramp, and D. L. Jones. 2015. Honey mesquite (*Prosopis glandulosa*) seedlings responses to seasonal timing of fire and fireline intensity. Rangeland Ecology & Management 68: 194-203.

Ansley, R. J. and P. W. Jacoby. 1998. Manipulation of fire intensity to achieve mesquite management goals in north Texas. Pages 195-204 *in* Proceedings of the 20th Tall Timbers Fire Ecology Conference, 1998, Tallahassee, Florida, USA.

Anthony, R. G. 1976. Influence of drought on diets and numbers of desert deer. The Journal of Wildlife Management 40: 140-144.

Archibald, S., W. J. Bond, W. D. Stock, and D. H. K. Fairbanks. 2005. Shaping the landscape: fire-grazer interactions in an African savanna. Ecological Applications 15: 96-109.

Augustine, D. J., L. E. Frelich, and P. A. Jordan. 1998. Evidence for two alternate stable states in an ungulate grazing system. Ecological Applications 8: 1260-1269.

Baker, W.L. 1993. Spatially heterogeneous multi-scale response of landscapes to fire suppression. Nordic Society Oikos 66: 66-71.

Beier, P. and D. R. McCullough. 1990. Factors influencing white-tailed deer activity patterns and habitat use. Wildlife Monographs 109: 3-51.

Blair, B. K., C. M. Britton, and D. N. Ueckert. 1993. Pricklypear control with fire and herbicides on Texas Rolling Plains. Texas Journal Agriculture Natural Resources 6: 87- 98.

Blydenstein, J. 1957. The survival of velvet mesquite (*Prosopis juliflora* var. *velutina*) after fire. Journal of Range Management 10: 221-223.

Bond, B. T., L. W. Burger, Jr., B. D. Leopold, K. D. Godwin, and J. C. Jones. 2002. Short-term responses of eastern cottontails to prescribed fire in east-central Mississippi. Pages 187-197 *in* Proceedings of the 56th Annual Conference of the Southeastern Association of Fish and Wildlife Agencies, 26 October—30 October 2002, Baltimore, Maryland, USA.

Brockway, D. G., R. G. Gatewood, and R. B. Paris. 2002. Restoring fire as an ecological process in shortgrass prairie ecosystems: initial effects of prescribed burning during the dormant and growing seasons. Journal of Environmental Management 65: 135- 152.

Brunjes, K. J., W. B. Ballard, M. H. Humphrey, F. Harwell, N. E. McIntyre, P. R. Krausman, and M. C. Wallace. 2006. Habitat use by sympatric mule and white-tailed deer in Texas. Journal of Wildlife Management 70:1351-1359.

Cancellare, M. 2018. Habitat-specific variation in mammal communities using camera traps in western Texas. Thesis, West Texas A&M University, Canyon, USA.

Cherry, M. J., R. B. Chandler, E. P. Garrison, D. A. Crawford, B. D. Kelly, D. B. Shindle, K. G. Godsea, K. V. Miller, and L. M. Conner. 2018. Wildfire affects space use and movement of white-tailed deer in a tropical pyric landscape. Forest

Ecology and Management 409: 161- 169.

Chia, E. K., M. Bassett, D. G. Nimmo, S. W. J. Leonard, E. G. Ritchie, M. F. Clarke, and A. F. Bennett. 2015. Fire severity and fire-induced landscape heterogeneity affect arboreal mammals in fire-prone forests. Ecosphere 6: 190.

Davis, J. 1967. Some effects of deer browsing on chamise sprouts after fire. American Midland Naturalist 77: 234-238

Dennison, C. C., P. M. Harveson, and L. A. Harveson. 2016. Assessing habitat relationships of mountain lions and their prey in the Davis Mountains, Texas. The Southwestern Naturalist 61: 18-27.

Dills, G. G. 1970. Effects of prescribed burning on deer browse. The Journal of Wildlife Management 34: 540-545.

Drewa, P. B. and K. M. Havstad. 2001. Effects of fire, grazing, and the presence of shrubs on Chihuahuan desert grasslands. Journal of Arid Environments 48: 429-443.

Eby, S. L., T. M. Anderson, E. P. Mayemba, and M. E. Ritchie. 2014. The effect of fire on habitat selection of mammalian herbivores: the role of body size and vegetation characteristics. Journal of Animal Ecology 83: 1196-1205.

Eddy, T. A. 1961. Foods and feeding patterns of the collared peccary in southern Arizona. Journal of Wildlife Management 25: 248-25.

Espartosa, K. D., B. T. Pinotti, and R. Pardini. 2011. Performance of camera trapping and track counts for surveying large mammals in rainforest remnants. Biodiversity and Conservation 20: 2815-2829

Ferguson, G. W., W. H. Gehrmann, A. M. Brinker, G. C. Kroh, and D. C. Ruthven. 2015.

Natural ultraviolet-B exposure of the Texas horned lizard (*Phrynosoma cornutum)* at a north Texas wildlife refuge. The Southwestern Naturalist 60: 231-239.

Heisler, J. L., J. M. Briggs, and A. K. Knapp. 2003. Long-term patterns of shrub expansion in a C4-dominated grassland: fire frequency and the dynamics of shrub cover and abundance. American Journal of Botany 90: 423-428.

Hill, K. C., J. D. Bakker, and P. W. Dunwiddie. 2017. Prescribed fire in grassland butterfly habitat: targeting weather and fuel conditions to reduce soil temperatures and burn severity. Fire Ecology 13: 24-41.

Hobbs, N. T. and R. A. Spowart. 1984. Effects of prescribed fire on nutrition of mountain sheep and mule deer during winter and spring. The Journal of Wildlife Management 48: 551-560.

Ivey, T. L. and M. K. Causey. 1984. Response of white-tailed deer to prescribed fire. Wildlife Society Bulletin 12: 138-141.

Jackson, A. S. 1965. Wildfires in the Great Plains grasslands. Pages 241- 259 *in* Proceedings of the 4th Tall Timbers Fire Ecology Conference, 18 March—19 March 1965, Tallahassee, Florida, USA.

Kirchner, B. N., N. S. Green, D. A. Sergeant, J. N. Mink, and K. T. Wilkins. 2011. Responses of small mammals and vegetation to prescribed burn in a tallgrass blackland prairie. The American Midland Naturalist 166: 112- 125.

Klinger, R.C., M.J. Kutilek, and H.S. Shellhammer. 1989. Population responses of black-tailed deer to prescribed burning. The Journal of Wildlife Management 53: 863-871.

Kramer, K., T. A. Groen, and S. E. van Wieren. 2003. The interacting effects of ungulates and fire on forest dynamics: an analysis using the model FORSPACE. Forest Ecology and Management 181: 205-222.

Lashley, M.A., M.C. Chitwood, R. Kays, C.A. Harper, C.S. DePerno, C.E. Moorman. 2015. Prescribed fire affects female white-tailed deer habitat use during summer lactation. Forest Ecology and Management 348: 220- 225.

Lochmiller, R. L., J. F. Boggs, S. T. McMurry, D. M. Leslie, Jr., and D. M. Engle. 1991. Response of cottontail rabbit populations to herbicide and fire applications on Cross Timbers rangeland. Journal of Range Management 44: 150-155.

Magurran, A. E. 2004. Measuring biological diversity. Blackwell Science, Malden, Massachusetts, USA.

Main, M. B. and L. W. Richardson. 2002. Response of wildlife to prescribed fire in southwest Florida Pine Flatwoods. Wildlife Society Bulletin 30: 213-221.

Martinuzzi, S., A. J. Allstadt, B. L. Bateman, P. J. Heglund, A. M. Pidgeon, W. E. Thogmartin, S. J. Vavrus, and V. C. Radeloff. 2016. Future frequencies of extreme weather events in the National Wildlife Refuges of the conterminous U.S. Biological Conservation 201: 327-335.

Masters, R. E., R. L. Lochmiller, and D. M. Engle. 1993. Effects of timber harvest and prescribed fire on white-tailed deer forage production. Wildlife Society Bulletin 21: 401-411.

Mixon, M. R., S. Demarais, P. D. Jones, and B. J. Rude. 2009. Deer forage responses to herbicide and fire in mid-rotation pine plantations. Journal of Wildlife Management 73: 663-668.

Neuenschwander, L. F., H. A. Wright, and S. C. Bunting. 1978. The effects of fire on a tobosagrass-mesquite community in the Rolling Plains of Texas. The Southwestern Naturalist 23: 315-337.

O'Brien, C. S., H. M. Boyd, P. R. Krausman, W. B. Ballard, S. C. Cunningham, and J. C. deVos. 2005. Influence of wildfire and coyote presence on habitat use by collared peccaries. Wildlife Society Bulletin 33: 865- 875.

Ojima, D. S., D. S. Schimel, W. J. Parton, and C. E. Owensby. 1994. Long- and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry 24: 67- 84.

Paragi, T. F., W. N. Johnson, D. D. Katnik, and A. J. Magoun. 1997. Selection of post-fire series by lynx and snowshoe hares in the Alaskan Taiga. Northwestern Naturalist 78: 77-86.

Pastro, L. A., C. R. Dickman, and M. Letnic. 2011. Burning for biodiversity or burning biodiversity? Prescribed burn vs. wildfire impacts on plants, lizards, and mammals. Ecological Applications 21: 3238- 3253.

Pinchak, W. E., W. R. Teague, R. J. Ansley, J. A. Waggoner, and S. L. Dowhower. 2010. Integrated grazing and prescribed fire restoration strategies in mesquite savanna: III. Ranch scale cow—calf production responses. Rangeland Ecology & Management 63: 298-307.

Poole, M. W. 2009. Effects of summer and winter burning on vegetation and wildlife in a sand sagebrush/ honey mesquite savanna. Thesis, West Texas A&M University, Canyon, USA.

Priesmeyer, W. J., R. S. Matlack, and R. T. Kazmaier. 2014. Precipitation and fire impacts on small mammals in shortgrass prairie. The Prairie Naturalist 46: 11-20.

Richardson, W. E., J. Hajek, E. Blakley, and C. Neitsch. 1974. Soil survey of Cottle County, Texas, United States Department of Agriculture, Soil Conservation Service, in cooperation with the Texas Agricultural Experiment Station. Washington, D. C., USA.

Ridenour, K. and R. Gray. 2010. Using trail cameras to understand fire behavior. Fire Management Today 70: 37-41.

Rogers, J. O., T. E. Fulbright, and D. C. Ruthven. 2004. Vegetation and deer response to mechanical shrub clearing and burning. Journal of Range Management 57: 41-48.

Rollan, A. and J. Real. 2011. Effects of wildfires and post-fire forest treatments on rabbit abundance. European Journal of Wildlife Research 57: 201-209.

Romo, S. 2018. Summer avian community structure response to habitat and landscape variables at Matador Wildlife Management Area. Thesis, West Texas A&M University, Canyon, USA.

Ruffault, J. and F. Mouillot. 2015. How a new fire-suppression policy can abruptly reshape the fire-weather relationship. Ecosphere 6: 1-19.

Rowcliffe, J. M. and C. Carbone. 2008. Surveys using camera traps: are we looking to a brighter future? Animal Conservation 11: 185-186.

Ruthven, III, D.C., R.T. Kazmaier, and M.W. Janis. 2008. Short-term responses of herpetofauna to various burning regimes in the South Texas Plains. The Southwestern Naturalist 53: 480-487.

Sharrow, S.H. and H.A. Wright. 1977. Effects of fire, ash, and litter on soil nitrate, temperature, moisture and tobosagrass production in the Rolling Plains. Journal of Range Management 30: 266-270.

Sharrow, S.H. and H.A. Wright. 1977. Proper burning intervals for tobosagrass in west Texas based on nitrogen dynamics. Society for Range Management 30: 343-346.

Smith, B. W., B. C. Dabbert, and R. M. Verble. 2019. Prescribed fire effects on rangeland dung beetles (Coleoptera: Scarabaeinae, Aphodiinae) in southern Great Plains. Rangeland Ecology & Management 72: 120-125.

Smith, J. K. 2000. Wildland fire in ecosystems: effects of fire on fauna. Rocky Mountain Ock Research Station, Fort Collins, Colorado, USA.

Stevenson, E.R., M.A. Lashley, M.C. Chitwood, J.E. Garabedian, M.B. Swingen, C.S. DePerno, and C.E. Moorman. 2018. Resource selection by coyotes (*Canis latrans*) in a longleaf pine (*Pinus palustris*) ecosystem: effects of anthropogenic fires and landscapes features. Canadian Journal of Zoology 97: 165- 171.

Sunquist, M. E. 1967. Effects of fire on raccoon behavior. Journal of Mammalogy 48: 673-674.

Texas Parks and Wildlife Department [TPWD]. 2011. Wildland Fire Management. <https://tpwd.texas.gov/landwater/land/wildland_fire_management/> Accessed 13 June 2020.

Teague, W. R., W. E. Grant, U. P. Kreuter, H. Diaz-Solis, S. Dube, M. M. Kothmann, W. E. Pinchak, and R. J. Ansley. 2008. An ecological economic simulation model for assessing fire and grazing management effects on mesquite. Ecological Economics 64: 611- 624.

Valkó, O., B. Deák, T. Magura, P. Török, A. Kelemen, K. Tóth, R. Horváth, D. D. Nagy, Z. Debnár, G. Zsigrai, I. Kapocsi, and B. Tóthmérész. 2016. Supporting biodiversity by prescribed burning in grasslands — A multi-taxa approach. Science of the Total Environment 572: 1377- 1384.

Vinton, M. A. and S. L. Collins. 1997. Landscape gradients and habitat structure in native grasslands of the central Great Plains. Pages 3-19 in F. L. Knopf and F. B. Samson editors. Ecology and Conservation of Great Plains Vertebrates. Springer Science + Business Media, LLC, New York, USA

Vogl, R. J. and A. M. Beck. 1970. Responses of white-tailed deer to a Wisconsin wildfire. The American Midland Naturalist 84: 270-273.

Wearn, O. R. and P. Glover-Kapfer. 2019. Snap happy: camera traps are an effective sampling tool when compared with alternative methods. Royal Society Open Science 6: 181748.

Wood, G. W. 1988. Effects of prescribed fire on deer forage and nutrients. Wildlife Society Bulletin 16: 180- 186.

Wright, H. A. 1974. Effect of fire on southern mixed prairie grasses. Journal of Range Management 27: 417-419.

Table 1. Species of mammals detected across all cameras used to explore the influence of days since burning on mammal activity at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

Common name	Zoological name	Total events	Average activity
White-tailed deer	*Odocoileus virginianus*	8953	14.253
Wild boar	*Sus scrofa*	1912	3.276
Mule deer	*Odocoileus hemionus*	1405	2.449
Coyote	*Canis latrans*	1346	1.751
Cottontail	*Sylvilagus* spp.	1383	1.824
Collared peccary	*Pecari tajacu*	436	0.898
Raccoon	*Procyon lotor*	340	0.484
Black-tailed jackrabbit	*Lepus californicus*	258	0.330
Striped skunk	*Mephitis mephitis*	254	0.318
Nine-banded armadillo	*Dasypus novemcinctus*	236	0.307
Bobcat	*Lynx rufus*	128	0.160
Mouse	--	72	0.089
Woodrat	*Neotoma* spp.	38	0.046
American badger	*Taxidea taxus*	33	0.041

Table 1, con’t. Species of mammals detected across all cameras used to explore the influence of days since burning on mammal activity at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

Common name	**Zoological name**	**Total events**	**Average activity**
North American porcupine	*Erethizon dorsatum*	24	0.030
Eastern fox squirrel	*Sciurus niger*	21	0.027
Bat	--	2	0.003

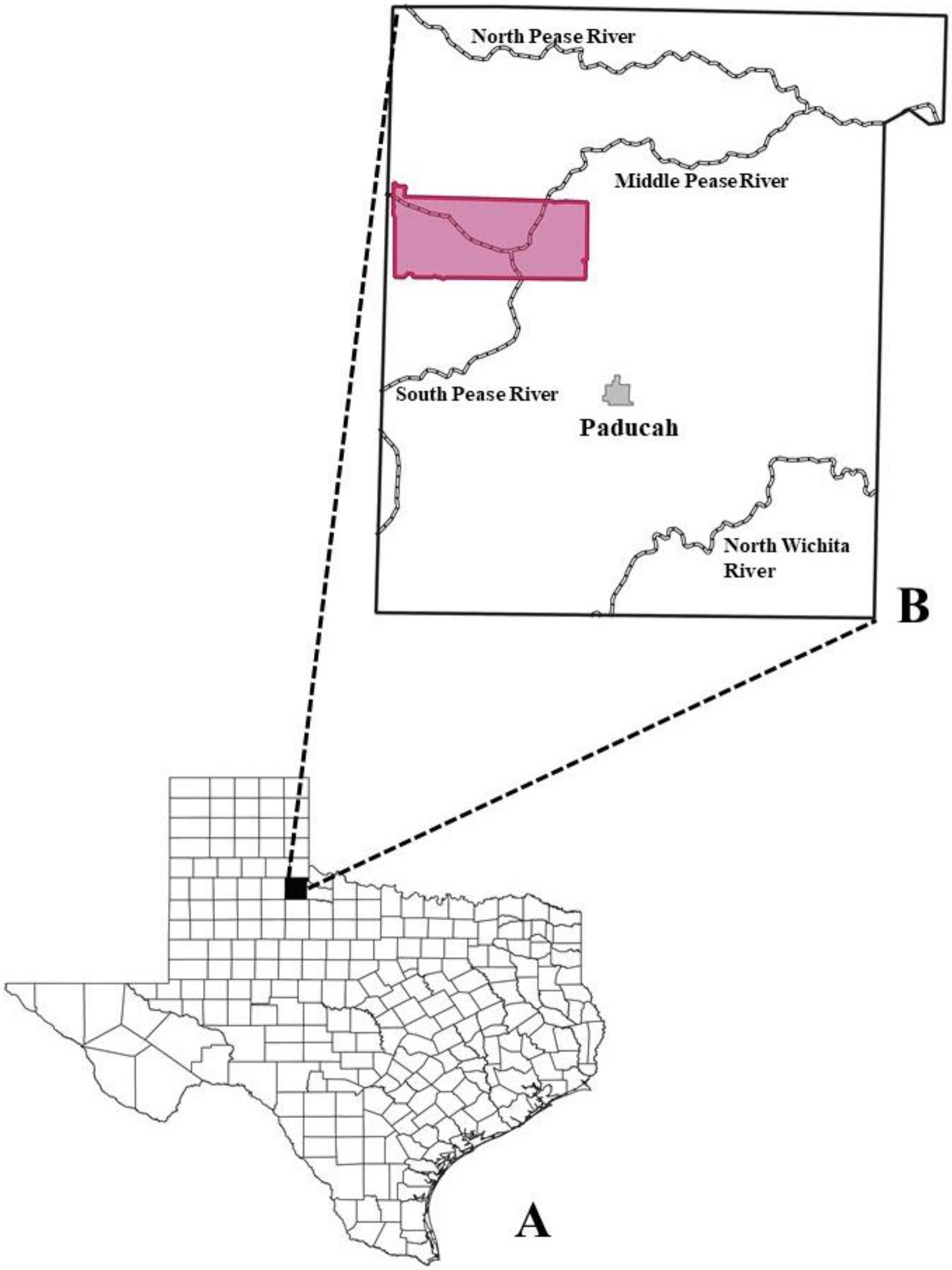

Figure 1. Blowout map indicating, (A) the location of Cottle County within the state of Texas and (B) Matador Wildlife Management Area (shaded in pink) within Cottle County located 15.9 km north of the town of Paducah and within the drainage of the Middle Pease River.

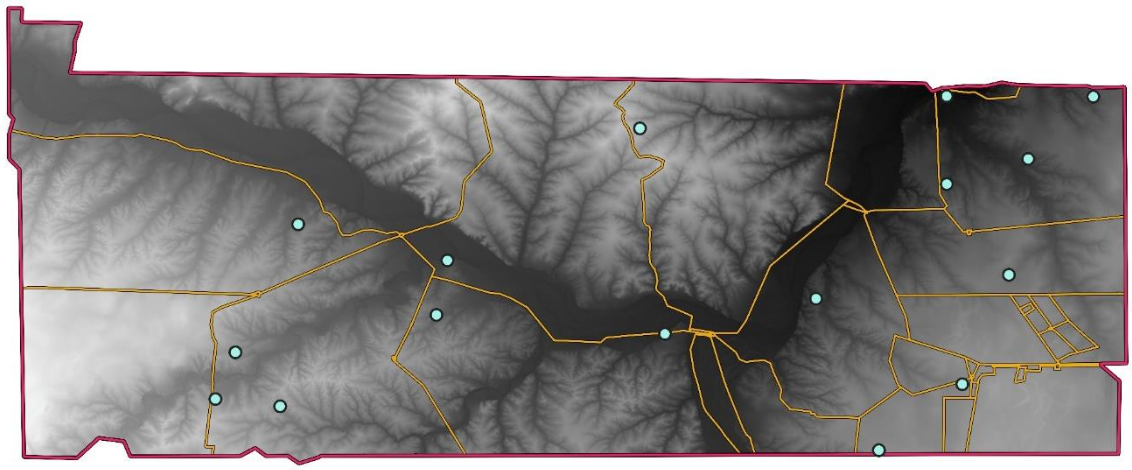

Figure 2. Matador Wildlife Management Area depicting location of the 16 camera traps (blue circles) used to explore mammal activity influence by days since burning, Cottle County, Texas, 2018-2020. Orange lines indicate fence lines between pastures and the background shading is LIDAR to illustrate change in elevation.

Figure 3. (A) Front and (B) side view of the plywood stand that trail cameras were secured to in order to explore mammal activity influence by days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

Figure 4. Representative habitat pictures of Matador Wildlife Management Area of a (A) drainage, (B) open upland, (C) lowland, and (D) shrubby upland where cameras were placed in order to explore mammal activity influence by days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

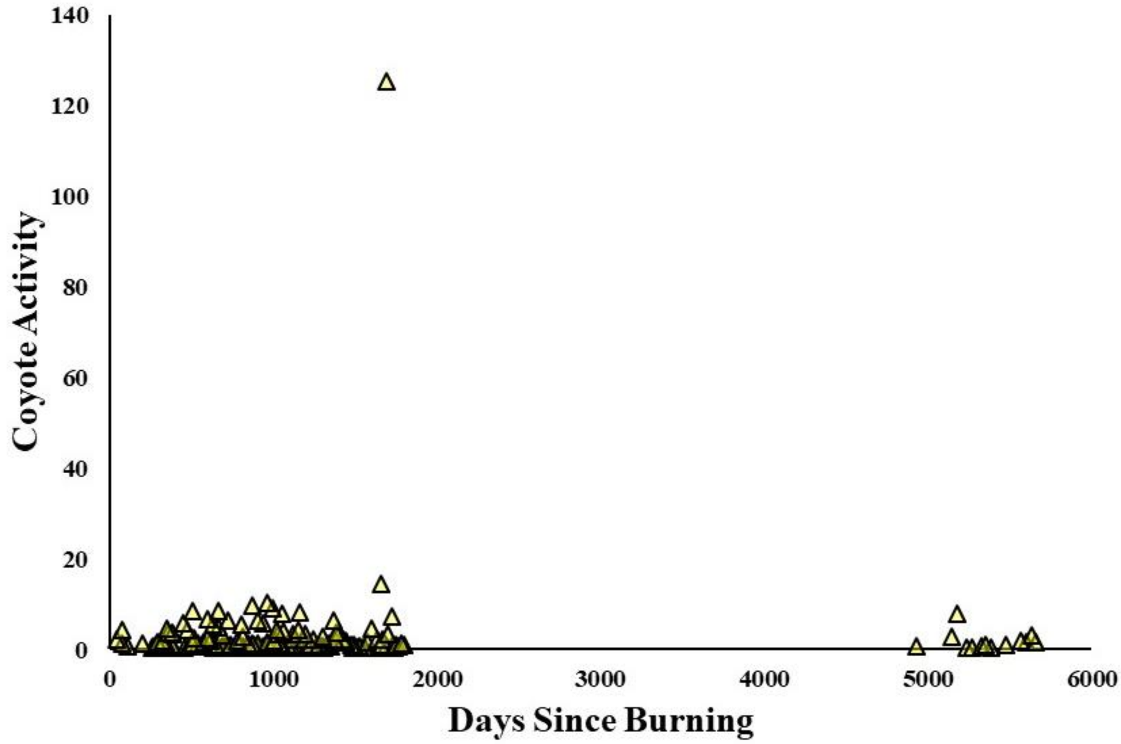

Figure 5. Relationship between coyote *(Canis latrans*) activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020. No regression line is present because the relationship was not significant.

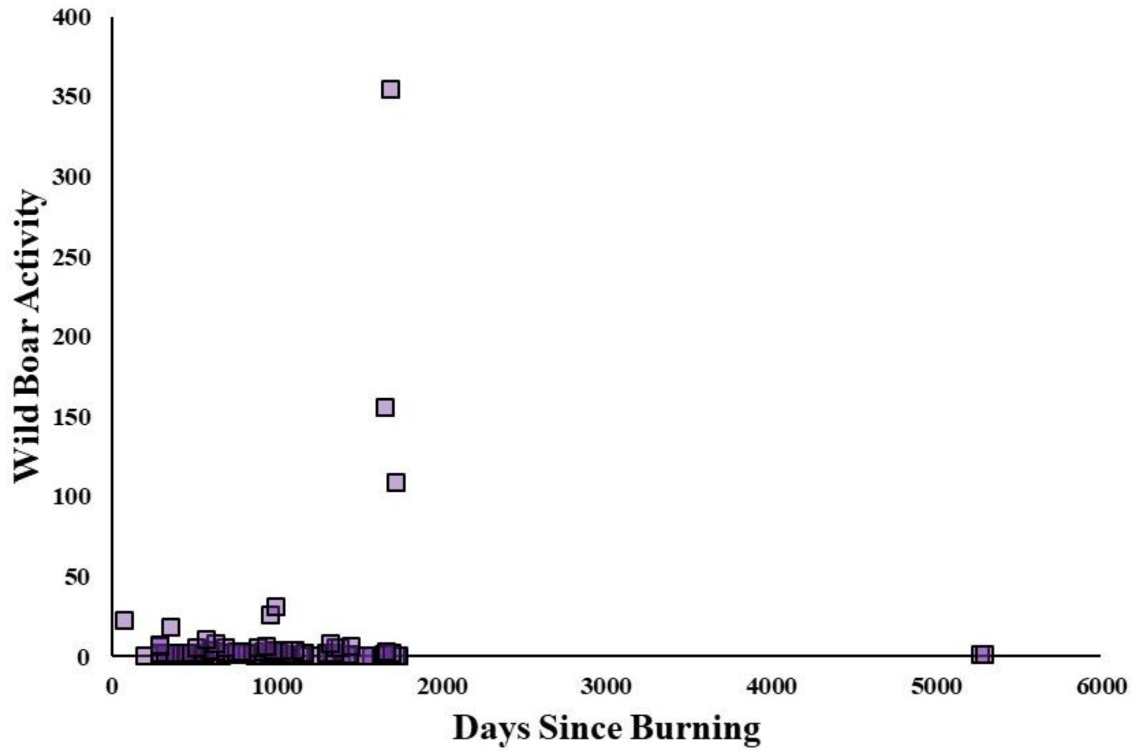

Figure 6. Relationship between wild boar (*Sus scrofa*) activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020. No regression line is present because the relationship was not significant

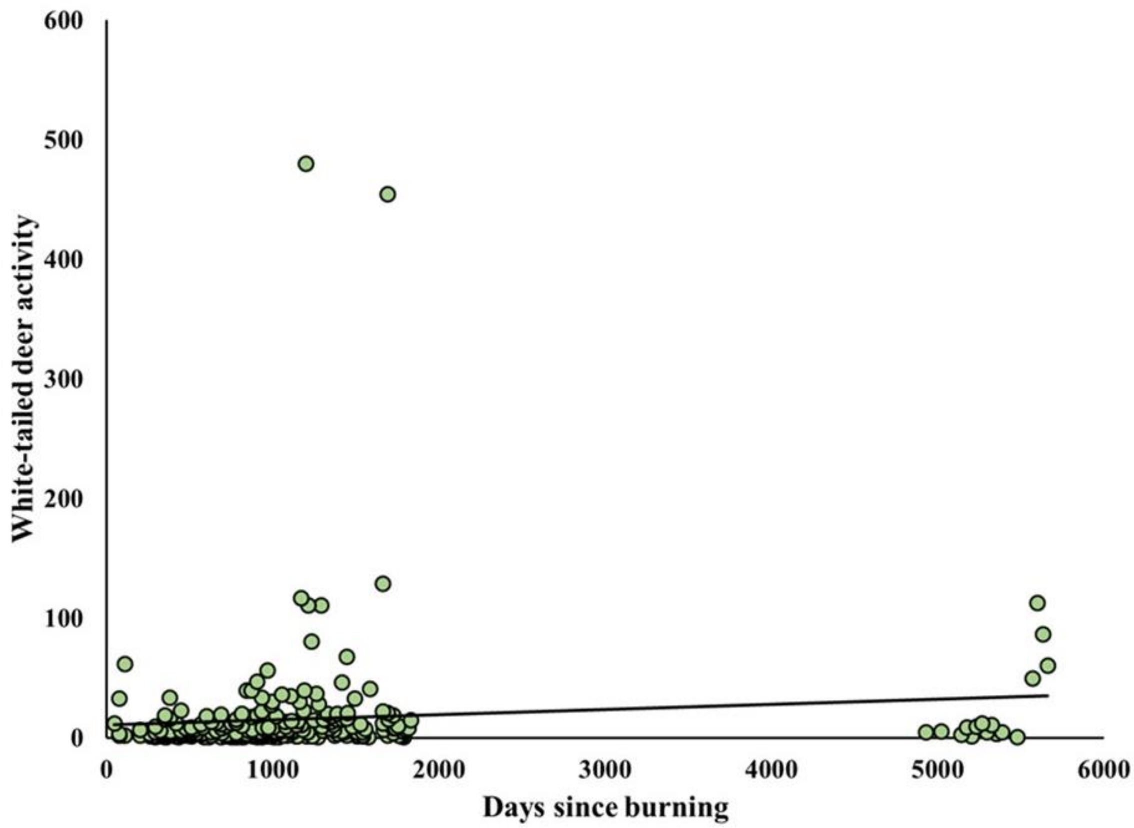

Figure 7. Relationship between white-tailed deer (*Odocoileus virginianus*) activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

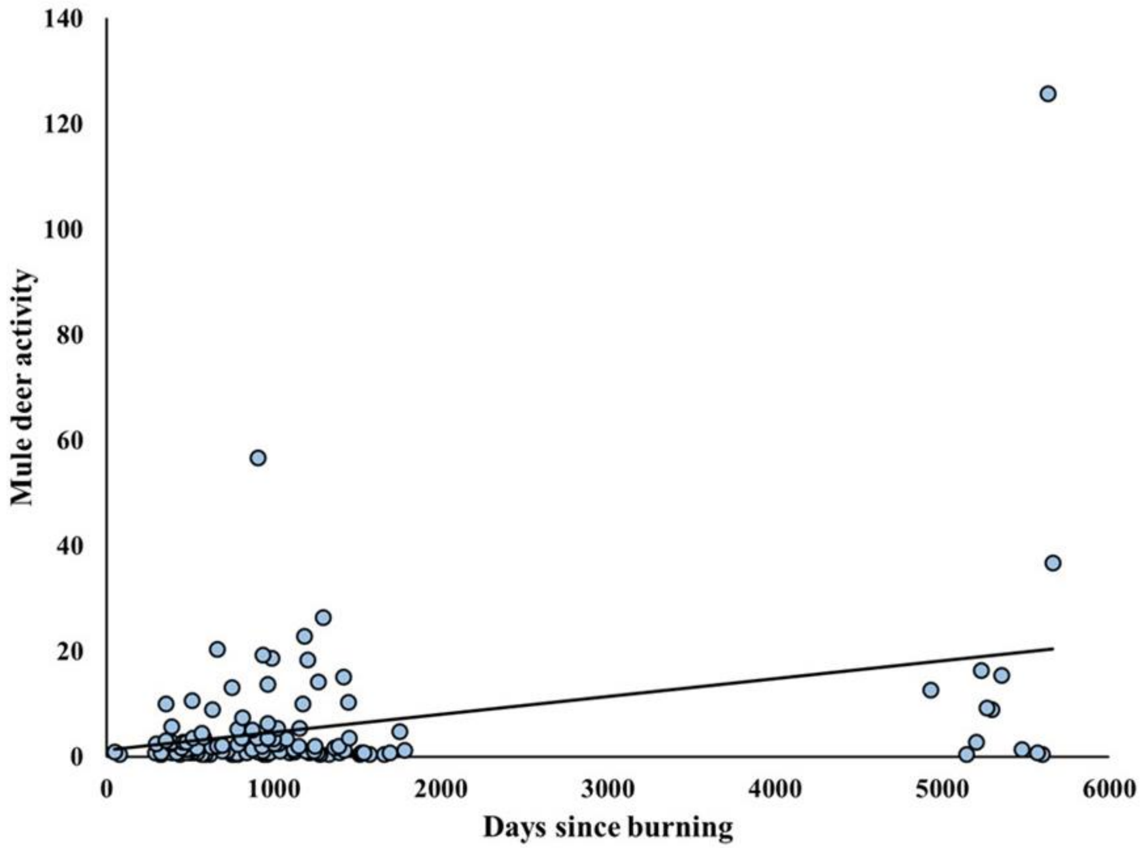

Figure 8. Relationship between mule deer (*Odocoileus hemionus*) activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020

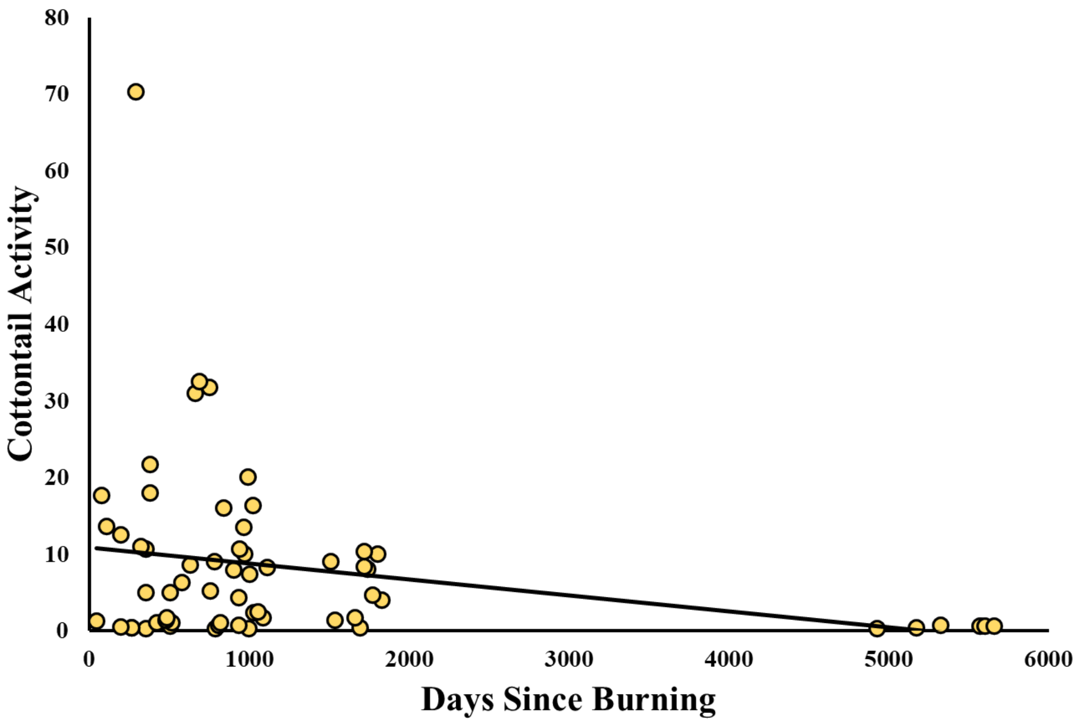

Figure 9. Relationship between cottontail (*Sylvilagus* spp.) activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

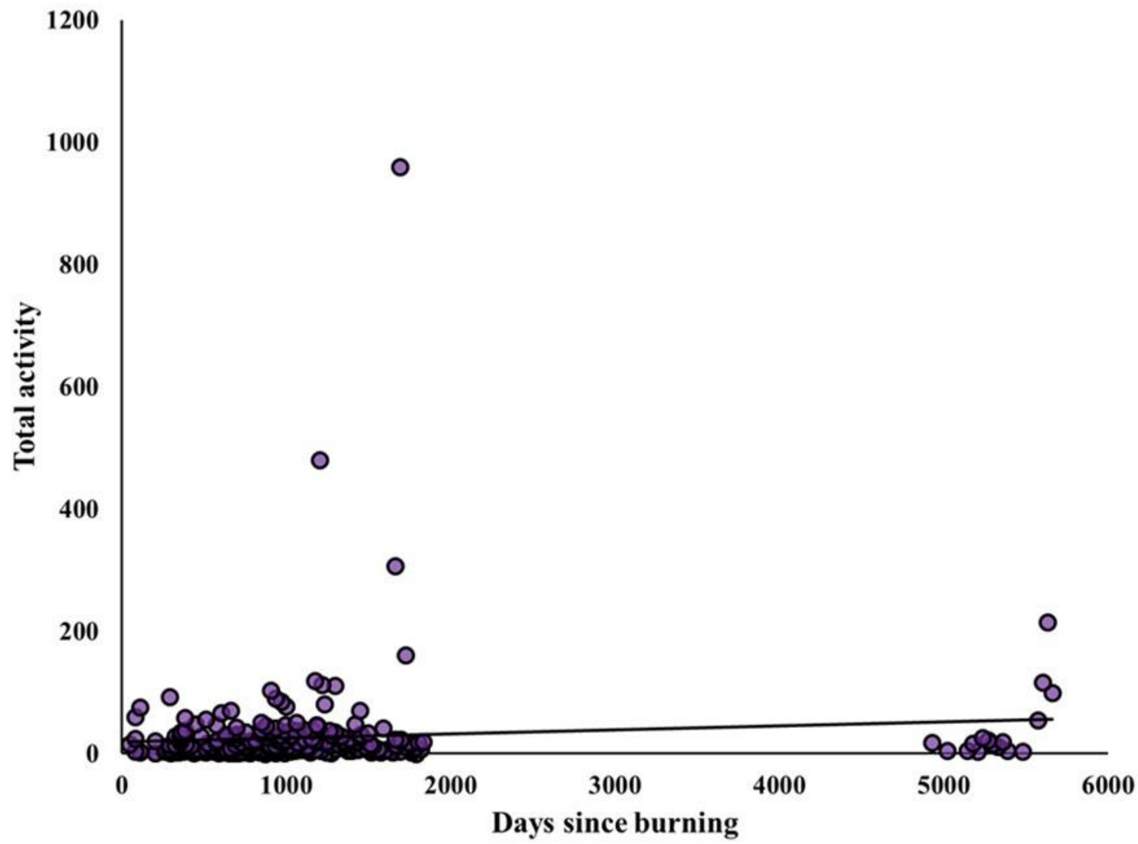

Figure 10. Relationship between total mammal activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

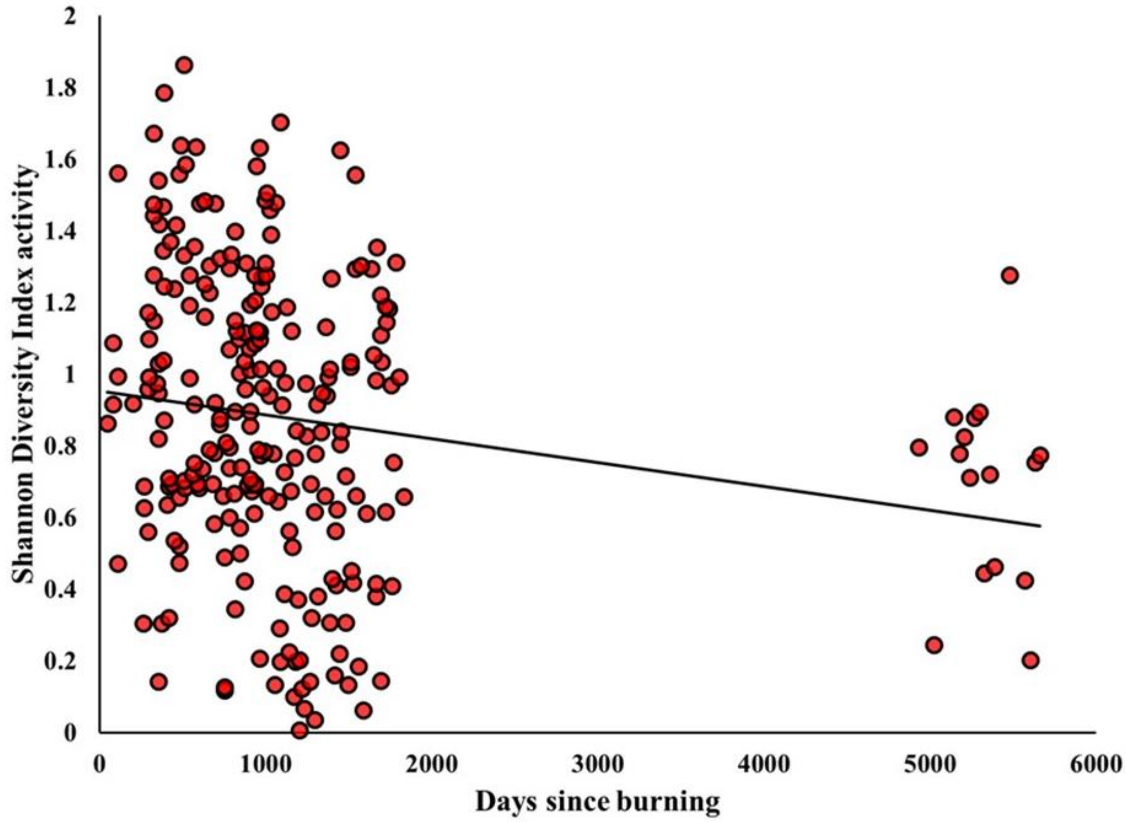

Figure 11. Relationship between Shannon Diversity Index of activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020.

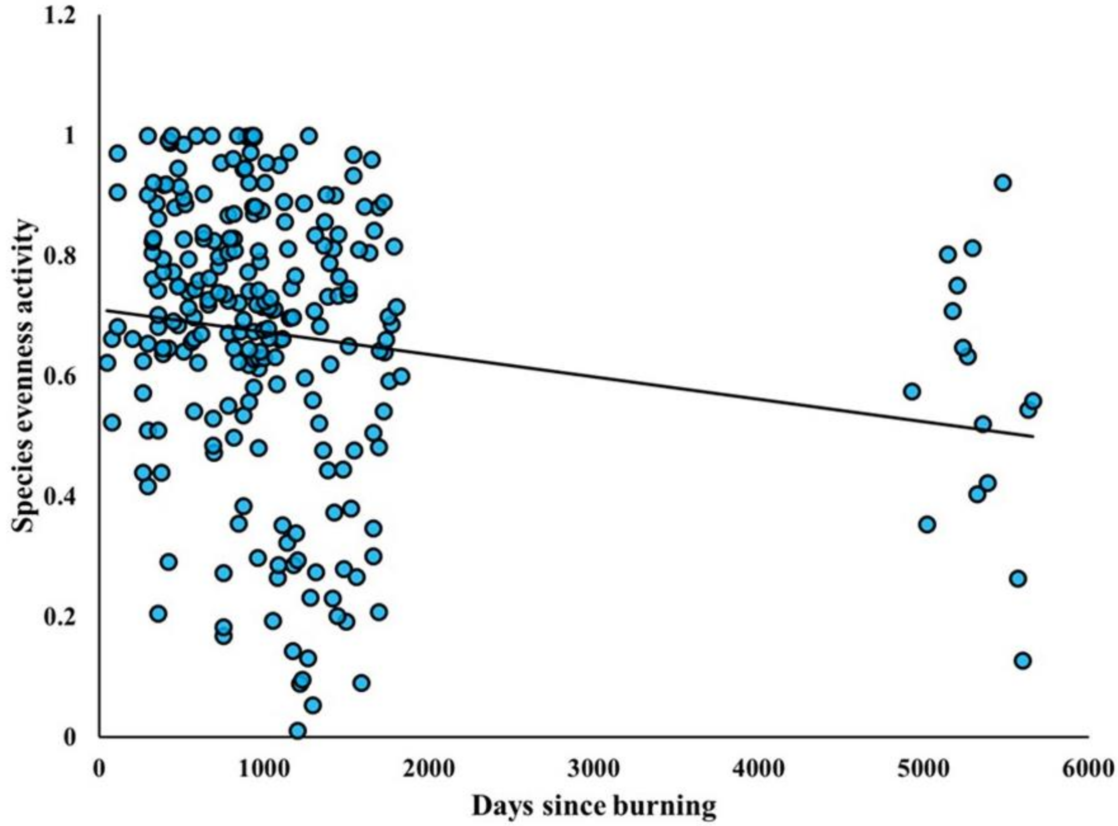

Figure 12. Relationship of species evenness of activity and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018- 2020.

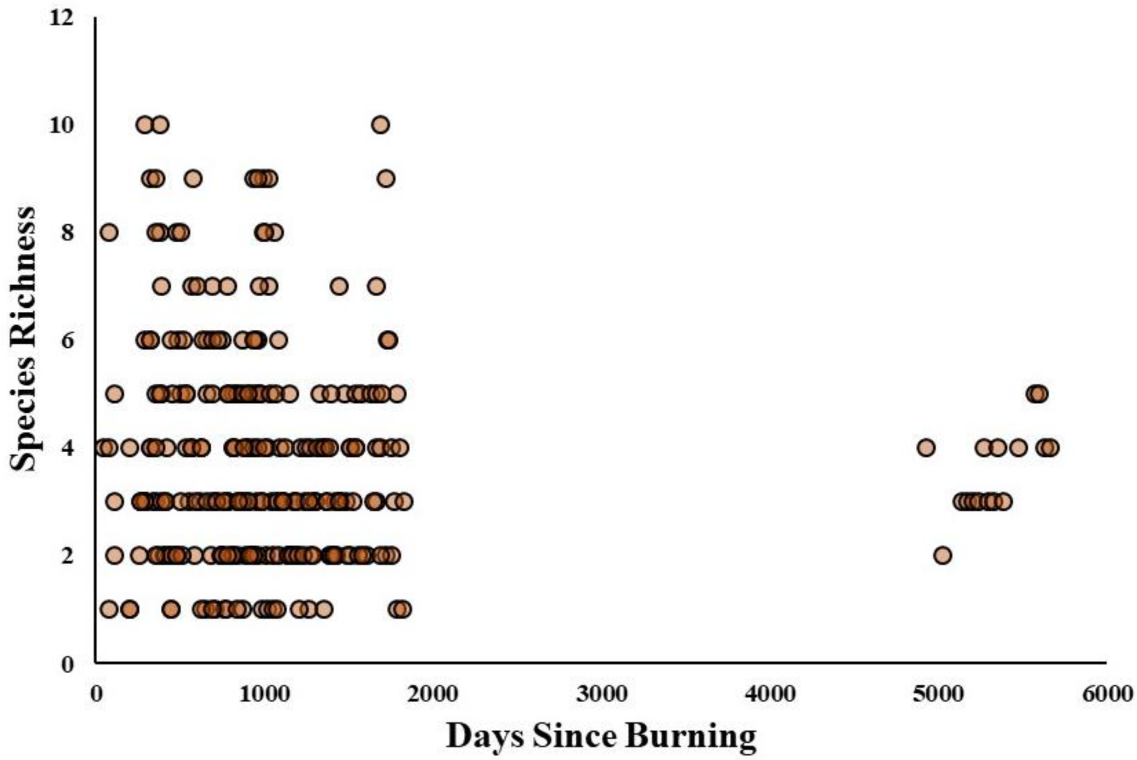

Figure 13. Relationship between species richness and days since burning at Matador Wildlife Management Area, Cottle County, Texas, 2018-2020. No regression line is present because the relationship was not significant.