CONSERVATION ECOLOGY OF THE TEXAS

HORNED LIZARD (*Phrynosoma cornutum*):

COMPARATIVE EFFECTS OF

SUMMER AND WINTER

BURNING

By

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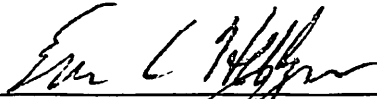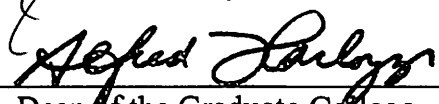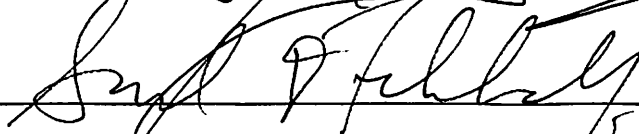Bachelor of Science

Auburn University

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2000
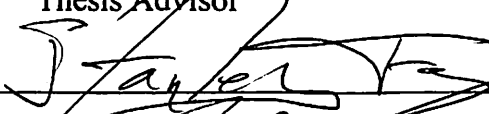
Submitted to the Faculty of the Graduate College of
Oklahoma State University in partial fulfillment of the
requirements for the Degree of
MASTERS OF SCIENCE
July, 2004

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ABSTRACT

The Texas horned lizard (*Phrynosoma cornutum*) has experienced apparent large-scale declines throughout its range, particularly in Texas. I studied the effect of prescribed burning (a habitat management practice of increasing popularity) on the ecology of the Texas horned lizard in a thornscrub savanna. I assessed home range size, woody vegetation selection, and survival rates of horned lizards in 4 treatments. Home ranges in the summer-burned-grazed treatment were smaller than those in the other treatments (winter-burned-grazed, unburned-grazed, and unburned-ungrazed). Survival rates in burned sites were higher than in unburned sites. The survival functions also differed between burning treatments, with survival declining in early summer in the unburned areas and in late summer in the burned areas. Lizards selected for whitebrush (*Aloysia gratissima*) and avoided Texas persimmon (*Diospyros texana*) consistently across treatments. Selection or avoidance of other woody species was not consistent among treatments. Ant activity, used as a surrogate of ant abundance, has been shown previously to be higher on burned sites on our study area. Therefore, more food, or better food-cover interspersion, may explain the higher survival in burned areas and the smaller home ranges in summer-burned areas. I conclude that prescribed burning in a thornscrub savanna provided favorable ecological conditions for Texas horned lizards.

INTRODUCTION

The recent global declines in amphibians (Alford and Richards 1999) and reptiles (Gibbons et al. 2000) have focused increasing attention on their conservation. Habitat

loss and modification are among the primary causes of these declines (Alford and Richards 1999, Gibbons et al. 2000). Therefore, the ecological effects of land-use practices on sensitive species, such as endangered, threatened or candidate species, are of considerable conservation, scientific, and political interest.

Fire has influenced the development of most ecosystems (Komarek 1966, Vogl 1971, Wright and Bailey 1982). It has important effects on ecosystem structure and function. Fire shapes an environment by generating a mosaic of habitats, which is important in creating and maintaining species diversity (Pianka 1989, Griffiths and Christian 1996). Previous studies with lizards have reported changes in species abundance and community composition related to the changes in habitat structure after fire (Fyfe 1980, Mushinsky 1985, Braithwaite 1987, Mushinsky 1992). The season, intensity, and frequency of fires can alter fire effects on lizard communities (Braithwaite 1987, Griffiths and Christian 1996). Effects also vary depending on the life habits of a particular species (Fyfe 1980, Braithwaite 1987, Pianka 1996).

The Texas horned lizard (*Phrynosoma cornutum*) is a species of special concern in the conservation community. Protected by Texas legislative mandate in 1967, the horned lizard has experienced apparent declines throughout its range, particularly in Texas (Price 1990). Factors suggested as causative of this decline include direct and indirect (insecticide use) effects of invasion by red imported fire ants (*Solenopsis invicta*), habitat alteration for other land uses (e.g., agriculture, development), highway mortality, and commercial exploitation (Price 1990, Donaldson et al. 1994).

Horned lizards and their primary food source, harvester ants (*Pogonomyrmex* spp.; Whitford and Bryant 1979), can be affected by habitat changes associated with

burning. Fires generally reduce shrub canopy cover and leaf litter, increasing bare

ground and herbaceous forage (Dunne et al. 1991, McPherson 1995, Wright and Bailey

1982). Hotter summer burns in particular tend to reduce brush cover (Scifres and

Hamilton 1993). The spatial distribution of Texas horned lizards in a central Texas

population was a function of harvester ant presence and open, partially vegetated habitat

(Whiting et al. 1993). Fair and Henke (1997) reported that horned lizards used burned

plots preferentially, perhaps because of ease of movement due to reduced ground litter.

A fire that increases grass or forb production would improve seed availability for

harvester ants, which are granivores (Rissing 1981). Harvester ant queens prefer to

establish new mounds in open areas with little vegetation (DeMers 1993), and the density

of harvester ant colonies is positively related to the frequency of burning (McCoy and

Kaiser 1990).

Horned lizards inhabit arid and semiarid areas and appear to prefer areas with

scant vegetation (Whiting et al. 1993). Burrow et al. (2001) identified the juxtaposition

of open areas and shrub cover as an important habitat variable for the Texas horned

lizard. Open areas are important to horned lizards for foraging (Pianka 1966) and

basking, but shrub cover and leaf litter are also important as thermal cover during the

hottest part of the day and for escape from predators (Burrow et al. 2001, Whitford and

Bryant 1979). Much vegetation in southern Texas is characterized by woody clusters or

shrub patches, generally referred to as mottes, interspersed with open grassy areas

(Whittaker et al. 1979, Archer et al. 1988, Scifres and Hamilton 1993). Given that

horned lizards benefit from a mixture of both habitats, what ratio of open area to shrub

clump is best? Do horned lizards with more edge in their home ranges fare better (e.g., have higher survival) than those that have less?

My work has direct conservation significance to the Texas horned lizard and perhaps other herpetofauna in the South Texas Plains. Burning is an increasingly popular land-use practice in the range of the Texas horned lizard, and my work will shed light on the comparative effects of summer and winter burning on lizard ecology and population status. Previous studies have focused on habitat use by horned lizards at the landscape (Whiting et al. 1993) and microhabitat (Burrow et al. 2001) scales. This study also addressed a different scale of habitat use by preliminary analyses at an intermediate level, the motte. I addressed habitat needs of horned lizards by analyzing the composition of home ranges of horned lizards with regard to interspersion of woody cover and open areas.

OBJECTIVES

The objectives of this study were to compare the ecological effects of summer and winter burning on the Texas horned lizard and to evaluate habitat use of Texas horned lizards at the motte level. The effects of summer and winter burning were studied by measuring lizard home range size, survival rates, woody plant selection, and habitat selection. Habitat use was studied by comparing the composition (e.g., % shrub mottes vs % open area) of lizard home ranges to the composition of the area as a whole.

HYPOTHESIS AND PREDICTIONS

I hypothesized that because fire is a natural component of the southern Texas ecosystem, prescribed burning will have positive effects on Texas horned lizard populations. I predicted that both summer and winter burning would have positive

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effects on horned lizards. Summer burning may be beneficial to the main prey item of horned lizards, harvester ants, by creating more open area, as previously noted by Burrow (2000) for the first and second summers post winter burning. However, summer fires may prove more damaging to herbaceous vegetation in the short term than winter burns (Scifres and Hamilton 1993). Because summer burning may result in more severe fires and increased brush reduction (Scifres and Hamilton 1993), I predicted that cooler winter burns would be better at creating the mosaic of bare ground, herbaceous vegetation, and woody vegetation in close proximity that is most beneficial to horned lizards (Burrow et al. 2001). I predicted that lizards in summer-burned areas would have smaller home ranges and higher survival rates than those in unburned areas due to the opportunity to find prey and cover in a smaller area as was noted by Burrow (2000) for winter burning. I predicted that lizards in summer-burned areas would have larger home ranges and lower survival rates than lizards in winter-burned areas because summer burning may create more open area than the ideal mosaic for horned lizards. I predicted that core areas (50% minimum convex polygon [MCP] home ranges) of lizards would have more woody motte edge per unit area than the surrounding, less-frequently used areas of their 95% MCP home ranges because edge areas provide horned lizards with the needed mixture of open area and shrub cover in close proximity (Burrow et al. 2001).

LITERATURE REVIEW

Ecology of the Texas horned lizard

The Texas horned lizard (*Phrynosoma cornutum*), generally nicknamed "horned toad" or "horny toad," is a 6-10-cm-long familiar inhabitant of the Southwest (Conant and Collins 1998). Many residents of this area remember playing with these heavily

armored but docile creatures as children. Horned lizards occur in arid and semiarid habitats including desert, thornscrub, and grassland (Pianka and Parker 1975), and seem to prefer open areas with less ground cover (Whiting et al. 1993). Two central head spines and a wide, flat body characterize this diurnal species (Conant and Collins 1998). Horned lizards are reluctant to run when approached by a predator, depending instead on their cryptic coloration to avoid detection (Pianka and Parker 1975). Their preferred food source is ants, primarily harvester ants (*Pogonomyrmex* spp., Whitford and Bryant 1979). Horned lizards must eat large numbers of ants to compensate for their low nutritional value (Pianka and Parker 1975). Adaptations to this ant-eating lifestyle include a large stomach capacity for their body size (Pianka and Parker 1975) and a blood factor to detoxify ant venom (Schmidt et al. 1989). The reproductive season of horned lizards lasts from April to July, and the lizards usually lay a single clutch averaging 29 eggs (Ballinger 1974). Burrow (2000) observed incubation lengths from 42 to 50 days.

Earlier work in southern Texas (summarized in Kazmaier et al. in review) documented that lizard home ranges in winter-burned pastures the first year post-burning were smaller than in unburned pastures, but level of grazing did not affect home range size. Summer (15 April - 15 August) survival rates of horned lizards were not affected by burning, but were higher in ungrazed pastures compared to either moderately-grazed or heavily-grazed pastures. Ant abundance and activity were consistently greater in burned pastures, but the effect of grazing varied across years. The smaller home ranges and greater prey abundance in burned pastures suggested a positive effect of winter fire on Texas horned lizards. Grazing effects were less clear. Fair and Henke (1997) reported that lizards used burned plots preferentially, and avoided burned/disked plots

and CRP plots. They suggested that burning and grazing as management strategies might be beneficial to horned lizards by creating open areas for foraging.

Vegetation and Fire

Whether caused by lightning or by humans, fire has historically played an important role in the southern Texas ecosystem (Scifres and Hamilton 1993). Lightning-caused wildfires in southern Texas were likely hot summer burns, whereas timing of human-caused fires varied (Scifres and Hamilton 1993). Before European settlement, fire naturally occurred on the Rio Grande Plains of southern Texas every 5 to 30 years (Wright and Bailey 1982). There has been a gradual increase in the density of woody plants in southern Texas over the last 300 years, in part caused by the reduced frequency and intensity of burning (Scifres and Hamilton 1993). Historically, fires occurred at least every ten years in desert grasslands, preventing the spread of woody shrubs. After 1880, livestock grazing reduced the load of fine fuel that was important for the spread of fires, reducing the frequency of fires (McPherson 1995).

The effects of fire on a plant community vary widely by species and depend on the behavior of the fire and post-fire physical and biological conditions, such as grazing or drought (McPherson 1995). The frequency of fire is also important. Burning too frequently can have adverse effects on vegetation and wildlife. Bock and Bock (1990) recommended that sites should not be re-burned until grasses and herbs have recovered to pre-burn conditions, typically in 3-4 years.

Fire effects may be altered by season. Plants are most damaged by fire during their growing season, but species are generally tolerant of fire when dormant (McPherson 1995). Early-summer fires cause high mortality for most perennial plants, which are

beginning growth (Cable 1965, 1967, 1973), whereas spring, fall, and winter fires mainly cause mortality of herbaceous plants that grow during the cool season (McPherson 1995). Early-summer fires cause more lasting effects than fires in other seasons (Pase 1971, Martin 1983). In general, repeated hot summer fires are most effective at reducing established woody vegetation, eventually increasing the amount of herbaceous understory, whereas cooler winter fires are used to prevent woody vegetation encroachment and encourage herbaceous growth in situations where brush already has been reduced to an acceptable level (Scifres and Hamilton 1993). However, it is important to bear in mind that the effects of fire season are variable and depend on many other factors (e.g., grazing history, successional stage, and weather patterns; Engle and Bidwell 2001).

One of the main uses of prescribed fire in southern Texas is brush reduction and improved forage for cattle (Scifres and Hamilton 1993). Ruthven et al. (2000) found greater forb coverage on winter-burned than unburned sites. Data gathered from an experimental fall burn by Box et al. (1967) demonstrated that fire reduces brush density without harming grass cover and production. Box and White (1969) found that burning without mechanical pretreatment reduced brush cover by 24%. Mechanical pretreatment can aid in successful brush removal (Box and White 1969), but is more costly than prescription burning. Several studies have shown that fire on areas without previous treatments of mechanical brush reduction resulted in an uneven burn, leaving the centers of large mottes intact and unharmed, creating a habitat mosaic (Box et al. 1967, Box and White 1969, Scifres and Hamilton 1993, Ruthven et al. 2000).

Effects of fire on herpetofauna

Fire can have direct and indirect effects on herpetofauna, and these effects vary based on the life habits and habitat needs of each species and on the frequency, intensity, and season of burning. Fire produces a mosaic of habitats within environments, which helps create and maintain species diversity (Pianka 1989, Griffiths and Christian 1996). Previous studies of lizards and fire have found changes in species abundance and community composition related to changes in habitat structure after fire (Fyfe 1980, Mushinsky 1985, Braithwaite 1987, Mushinsky 1992). Direct effects of fire on amphibian and reptile populations are often minor (Kahn 1960, Lawrence 1966, Means and Campbell 1981); however, harsher effects may occur depending on several factors, which I will discuss shortly. Amphibians and reptiles typically are able to burrow under the soil or move away beforehand to avoid the direct heat of fire (Mushinsky 1985). Kahn (1960) noted that adult western fence lizards (*Sceloporus occidentalis*) survived fires by seeking refuge under rocks and in burrows.

The effects of fire on herpetofauna can vary based on the season of burning. Braithwaite (1987) found that lizards in the wet-dry tropics of Australia exhibited a range of relationships to different fire regimes, i.e., some lizards benefit, some are fire-sensitive, and some seem unaffected. Time of year and intensity of the fire were more important than habitat succession after fire in determining composition and abundance of lizard communities in a habitat type that often burns annually. Griffiths and Christian (1996) found no direct mortality of frillneck lizards (*Chlamydosaurus kingii*) from early-season fires, but 29% mortality from high intensity, late dry-season fires.

The rate of recurrence of fire also can alter its effects. Mushinsky (1985) found that burning increased herpetofauna diversity and abundance, and that some fire frequencies (every year or every 7 years) were better than others (every 2 years) for maintaining high diversity. Mushinsky (1992) found that unburned areas or areas burned on 5-7 year cycles had more southeastern five-lined skinks (*Eumeces inexpectatus*) than areas burned on 1- or 2-year cycles because skinks need the leaf litter that fire removes.

The life history and habitat needs of each species also affect their response to fire, as illustrated by Mushinsky's (1992) study on southeastern five-lined skinks. Fyfe (1980) suggested that lizard species that sheltered in ground litter experienced the greatest mortality during fires, based on post-wildfire density estimates. Fire-induced changes in the structure of a habitat are beneficial to some lizard species (Mushinsky 1985) and detrimental to others (Patterson 1984). Griffiths and Christian (1996) found that frillneck lizards (*Chlamydosaurus kingii*) consumed a greater volume and diversity of prey after fires and changed perch tree preference due to changes in habitat structure after fire. Lillywhite and North (1974) noted changes in perch preference in western fence lizards after chaparral fire, likely due to improved basking sites with cover in close proximity. Ford et al. (1999) found no overall impact on herpetofauna from high-intensity prescribed fires in April in the southern Appalachians and noted that fire created a mosaic of vegetation including unburned or mildly affected areas. These latter areas protected moisture-sensitive species. Kahn (1960) also found that diet and reproduction of western fence lizards were the same in burned and unburned areas.

Effects of fire on horned lizards and their prey

Several authors (Ruthven et al. 2000, Fair and Henke 1997) have suggested that winter burns may harm horned lizards during hibernation because horned lizards have been reported to hibernate just below the soil surface or under leaf litter. Soil temperatures decrease rapidly with depth, and during a grassland fire, no significant effects are observed 1 cm below the surface. However, shrubland fires increased soil temperatures up to 50°C at 5-cm soil depth (Wright and Bailey 1982).

Harvester ants, the main prey of horned lizards (Whitford and Bryant 1979, Blackshear and Richerson 1999), can benefit from prescribed fires. Harvester ants are granivores (Rissing 1981), and a fire that increases grass or forb production would improve food availability for the ants. Andersen (1988) found little direct ant mortality due to fire, probably due to safety in underground nests, and he noted an increase in ant abundance and seed predation after fire. Jackson and Fox (1996) found only a minor negative impact on the ant community due to fire and even noted beneficial effects of fire for the ants, such as clearing obstructions to foraging. The density of harvester ant colonies was positively related to the frequency of burning in the sandhill habitats of Florida (McCoy and Kaiser 1990). Harvester ant queens prefer to establish a new mound in open areas with little vegetation (DeMers 1993). Burrow (2000) noted greater activity of harvester ants in burned than unburned sites. Burning also can improve conditions for desert termites (Isoptera; Scifres and Hamilton 1993), another prey of horned lizards (*personal observations*).

Fires generally reduce shrub canopy cover and leaf litter, increasing bare ground and herbaceous forage (Wright and Bailey 1982, Dunne et al. 1991, McPherson 1995).

Open area is important to horned lizards for foraging (Pianka 1966) and basking, but shrub cover and leaf litter also are important as thermal cover during the hottest part of the day and as escape cover from predators (Burrow et al. 2001, Whitford and Bryant 1979). Horned lizards typically bask and feed in the morning, seek shelter in a shrub canopy at midday and early afternoon, and bask again in late afternoon (Whitford and Bryant 1979). A study by Fair and Henke (1997) reported that horned lizards used burned plots preferentially, perhaps because of ease of movement due to reduced ground litter.

Field techniques to assess habitat use by horned lizards

Previous studies have used visual surveys and capture rates to determine use of an area by horned lizards (Whiting et al. 1993, Fair and Henke 1997). Henke (1998) commented on the variable effectiveness of human searchers and noted that a visual census of an area for reptiles can generate misleading results. Henke (1998) also noted that human searchers are less efficient as item abundance declines. A scarce item would appear more so because the search image is lost after a length of time without success. Visual surveys for horned lizards also may be biased by the ease of locating a lizard in an open area as opposed to an area with denser vegetation (Whiting et al. 1993). Radiotelemetry allows consistent tracking of the subject in various habitat types and reduces observer bias.

STUDY AREA

The study area was the 6,150-ha Chaparral Wildlife Management Area (CWMA) in Dimmit and LaSalle counties, Texas. The CWMA was purchased by the state of Texas in 1969 and management authority was given to the Texas Parks and Wildlife

Department (TPWD). Whittaker et al. (1979) described vegetation on the Chaparral Wildlife Management area as mesquite grassland consisting of discrete shrub patches embedded in a grass-dominated matrix. These woody clusters or shrub patches are generally referred to as mottes. Vegetation on the area is mainly honey mesquite (*Prosopis glandulosa*) woodlands or parklands (Burrow et al. 2001). Other important brush species include spiny hackberry (*Celtis pallida*), brasil (*Condalia hookeri*), blackbrush acacia (*Acacia rigidula*), twisted acacia (*Acacia schaffneri*), hogplum (*Colubrina texensis*), Texas persimmon (*Diospyros texana*), prickly pear cactus (*Opuntia engelmannii*), and tasajillo (*Opuntia leptocaulis*, Burrow et al. 2001). Common and scientific names for vegetation follow Hatch et al. (1990). Twenty-year annual precipitation on the area has averaged 66 cm, with peaks in May, September, and October, although county averages are only 53 cm (TPWD, unpublished data).

Since 1996, grazing occurred from 1 October to 30 April in a 1-herd, 13 pasture high intensity-low frequency system. Grazed pastures were stocked at approximately 25 animal-unit days (with AUD = 2 steers for one day) per hectare per year. Two pastures were maintained as ungrazed controls. Grazing was suspended for the 2002-2003 season. Therefore, the area had not been grazed for a year before the 2003 data were collected. A prescribed burning program on CWMA began in 1997. Winter-burned areas were burned in winter 1997-98 and again in 1999-2000. Summer-burned areas were burned in summer 1999. Thus, during my study, all burning treatments were 3-4 years post-burning.

Study sites (50-60 ha) were selected on the CWMA based on similarities in dominant woody species and woody canopy cover. The study sites comprised various

combinations of burning (winter-burned, summer-burned, unburned) and grazing (ungrazed and moderately grazed [ca. 25 AUD/ha]; Figure 1). Treatments were: unburned/moderately grazed (UBG; n = 2), winter-burned/moderately grazed (WBG; n = 2), summer-burned/moderately grazed (SBG; n = 1), and unburned/ungrazed (UBUG; n = 1). Treatments were not randomly allocated to study sites due to management constraints.

METHODS

Field sampling

Horned lizards were captured by hand during fortuitous encounters on roads and in the brush within study plots. Lizards were sexed and implanted with an intra-abdominal passive integrated transponder (PIT; AVID, Norco, California, USA) tag. The fifth toe on the right front foot also was clipped to indicate that the lizard had previously been caught. Body mass, snout-vent length, and total length were determined. Selected adult individuals over 75 mm snout-vent length (SVL) from each of the 6 sites were fitted with $\leq$3-g radio transmitters (150-151 MHz, L and L Electronics, Mahomet, Illinois, USA) in custom-made backpacks. Backpacks were constructed of a beige muslin material and elastic straps dyed to match the substrate color to minimize disruption of the animal's cryptic coloration. The backpack was attached by placing the front strap around the subject's neck and one front leg, and by placing a second strap around the subject's waist. A drop of cyanoacrylate adhesive gel secured the straps to the lizard's chest and lower abdomen. Receiving range of the transmitters was approximately 100 m and could be extended to 200 m by attaching the antenna to the end of a 5-m PVC pole.

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Radiotransmittered individuals were initially relocated twice daily using a handheld two-element Yagi antenna until lizards had accumulated at least 20 locations. Monitoring was then reduced to once daily until the end of the season (15 August). Location was confirmed either by visual observation or by encircling the signal within a shrub clump. UTM coordinates of radiolocations were determined using a hand-held Garmin eTrex GPS unit, and coordinates were recorded with a signal accuracy of ≤ 7 m.

Habitat selection was evaluated at the microhabitat and motte level. X-Y coordinates and microhabitat characteristics (e.g., grass, forb, or shrub species and percent cover, bare ground, distance to nearest shrub) for each location were determined. Woody vegetation transects were conducted on each area to determine woody species available to horned lizards. Three 100-m transects oriented north, southeast, and southwest were sampled at several designated locations (n = 3 for SBG and WBG, n = 6 for UBG and UBUG) within each treatment site. To study motte-level habitat selection, shrub cover in one study site (unburned, ungrazed) was mapped by walking the perimeter of each motte and delineating the drip line of the vegetation using a hand-held Trimble GeoExplorer II GPS unit set to record locations on continuous stream. Dominant woody species, number of species, height of motte, and ground cover were recorded. The dominant species of the motte was chosen as the species that determined the character of the motte (i.e., the species that structured the motte, gave it shape, and provided canopy cover). Number of species was classified as 1, 2-3, 4-5, or ≥ 6 species. Height of motte was classified as < 1 m, 1-2 m, 2-3 m, or >3 m. Ground cover beneath the motte was classified as mixed (from 40:60 to 60:40 litter:herbaceous vegetation ratio), mostly litter (> 60% litter), or mostly herbaceous (> 60% herbaceous vegetation). Mottes < 1 m apart

were combined (based on GPS resolution), unless the characters of the two mottes were very distinct.

Data analysis

Minimum convex polygon (Mohr 1947) and fixed kernel (Worton 1989) home ranges were calculated using the Animal Movement Analysis ArcView extension (Hooge and Eichenlaub 1997). The minimum convex polygon (MCP) is the oldest and most commonly used method of home range estimation (Mohr 1947, Kernohan et al. 2001). It constructs a home range by connecting a series of outer locations to form a convex polygon. It is nonparametric, but is sensitive to outliers (Kernohan et al. 2001). The kernel method places a probability density function over each point and gives a higher-density value where points are concentrated; these densities are shown as contours (Kernohan et al. 2001). This method is also nonparametric, but is better at accommodating outliers (Kernohan et al. 2001).

Home range size of radioed individuals was compared among treatments. Because the Shapiro-Wilk test for normality revealed that the distribution of home range sizes by treatment, sex, and year was not normal and Levene's test revealed that variances were not homogeneous, nonparametric analyses (e.g., Rank transformed ANOVA, Kruskal-Wallis, and Wilcoxon Rank Sum) were used for comparisons of these variables. I conducted ANOVAs on rank transformed and log transformed data to test for an interaction between treatment and year and between treatment and sex. Preliminary analyses for both methods found no interactions between treatment and year or between treatment and sex, therefore data were pooled across year and sex. Data were also pooled across sites containing same treatment. If the Kruskal-Wallis test indicated differences

(P <0.05) among treatments, then the Wilcoxon Rank Sum test was conducted for all possible pairwise comparisons. Individual lizards within each study site served as the experimental units. This type of study design is considered pseudoreplication (Hurlbert 1984), but the importance of this distinction has been questioned (Oksanen 2001), and in this situation it would not be feasible to have true replicates (i.e., multiple 50-60-ha study sites within each treatment). To ensure a reliable home range estimate, only lizards tracked for ≥ 20 locations were included in analyses.

Survival rates were estimated using the Kaplan-Meier Limit Estimator (K-MLE) with the staggered-entry design (Pollock et al. 1989) and the Heisey-Fuller (1985) method. K-MLE and Heisey-Fuller allow newly transmittered individuals to be added to the sample interval at any time. Two methods were used to calculate survival rates to account for the unknown fate of missing (censored) animals. In method 1, all censored animals were assumed live. In method 2, all censored animals were assumed dead. Because timing of mortality precluded the use of Kaplan-Meier survival estimates for comparison by sex and year, the Heisey-Fuller method (Heisey and Fuller 1985) also was used to make certain comparisons.

Several assumptions exist for K-MLE. First, all animals of a particular cohort (e.g., sex or age class) have been sampled randomly. In this study, only adults were monitored. Our captures may be biased toward more mobile animals, which are more visible and presumably more vulnerable to predation. Second, survival times are independent for different animals. Failure of this assumption does not cause bias, but makes estimates appear to have smaller variances than they actually do. The solitary life history of this species makes this assumption valid. Third, capturing the animal or having

it carry a radio collar does not influence its future survival. The 3-g mass of the pack was always ≤10% of the body mass of the lizard, whereas clutch sizes of this species average 28.8 to 30.7 % of total female body mass (Pianka and Parker 1975). Wone and Beauchamp (2003) used transmitters up to 20% of horned lizard body mass and reported that other studies used even heavier telemetry packs with no ill effects on activity or survival. Fourth, censoring (removal from analysis) is random or not related to an animal's fate. Fifth, newly-tagged animals have the same survival function as previously tagged animals (Pollock et al. 1989).

Survival rates of radioed individuals were compared among treatments. I used a log-rank test to test for pairwise differences in the survival function (shape of the curve) of grazing and burning treatments (Pollock et al. 1989). A Z-test statistic also was used to compare the survival curves on the last day of summer monitoring (endpoint survival; Pollock et al. 1989). Survival data were pooled across years and sexes because preliminary analyses indicated no differences in endpoint survival rates between 2002 and 2003 as estimated by method 1 (2002: s = 0.67, σ = 0.01, 2003: s = 0.84, σ = 0.01; Z = -1.14, P= 0.25) or method 2 (2002: s = 0.55, σ = 0.01, 2003: s = 0.35, σ = 0.01; Z = 1.28, P = 0.20) or male and female survival rates as estimated by method 1 (male: s = 0.87, σ = 0.02, female: s = 0.70, σ = 0.01; Z = 1.09, P= 0.28) or method 2 (male: s = 0.65, σ = 0.03, female: s = 0.40, σ = 0.01; Z = 1.41, P = 0.16). Data were also pooled across sites containing the same treatment (i.e. winter-burned-grazed) and across burning and grazing treatments (i.e. burned vs unburned) for various comparisons. I considered P ≤ 0.05 to represent significance in all analyses.

Horned lizard locations from the summers of 2002 and 2003 were sorted by dominant woody plant and by treatment. The total number of locations at each woody species by treatment was tallied. Woody species seldom used (< 5) by lizards were grouped with similar species where possible or added to a category termed "other." Locations with no woody vegetation were excluded. Availability data were gathered from woody vegetation transects conducted during the summer of 2002, which determined woody cover (i.e., cm of cover per 100-m line). Centimeters of cover by each species were averaged across all transects in each treatment to calculate a mean proportion. This proportion was then multiplied by the sample size of lizard locations (excluding "none") in that treatment to estimate the available frequency of each species for that treatment. Availability data were then grouped into the same categories as the use data. Use and availability of each species by treatment were compared using log-linear modeling (PROC CATMOD, SAS Institute Inc. 2001). Use and availability within each treatment was compared by Chi-square analysis for each species that exhibited a treatment by use-availability interaction.

I calculated overall richness and diversity of woody species available and woody species used at all horned lizard locations by treatment. Shannon-Wiener (H'; Shannon and Weaver 1949) and Simpson's (D; Simpson 1949, Krebs 1978) diversity indices were used. Simpson's index is more biased toward abundant species and is often referred to as a dominance index, whereas Shannon-Wiener is more affected by the overall number of species or species richness (Stiling 1999). I also calculated richness and diversity of woody vegetation used on a per lizard basis using 20 randomly-selected locations for each lizard. I then compared richness and diversity among treatments using lizards as

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replicates. Because the Shapiro-Wilk test for normality revealed that the distribution of diversity values by treatment was not normal, nonparametric analyses (e.g., Kruskal-Wallis and Wilcoxon Rank Sum) were used for comparisons. If the Kruskal-Wallis test indicated differences ($P < 0.05$) among treatments, the Wilcoxon Rank Sum test was conducted for all possible pairwise comparisons.

Home range data were overlain on a map of shrub cover to determine the mean motte size, edge density, mean patch edge, mean perimeter-area ratio, and open area-shrub cover ratio within individual home ranges. The composition of 95% MCP home ranges of each lizard was compared to the overall mapped area composition and to their 50% MCP home ranges using paired t-tests. These analyses were conducted using the Patch Analyst 2.2 ArcView extension (Elkie et al. 1999, Rempel and Carr 1999). I planned to regress ecological characteristics of lizards (e.g., survival, home range size, mean daily movement distances) on spatial characteristics of their home ranges (e.g., % edge, % shrub, mean size of motte), but lizard sample size (n=2) was too small for this sort of comparison. This analysis was conducted only on 1 treatment: unburned-ungrazed.

RESULTS

Home Ranges

A total of 47 summer home ranges from 47 lizards were used in home range analyses. Total area used by horned lizards ranged from 0.06 to 23.26 ha for 95% minimum convex polygon and 0.09 to 21.59 ha for 95% fixed kernel (Table 1). Home range size varied by treatment for both 95% MCP ($T_3 = 7.77$, $P = 0.05$) and 95% Kernel ($T_3 = 9.39$, $P = 0.02$) methods (Table 2, Figure 2). Home ranges in the summer-burned-

grazed treatment were smaller than those in the winter-burned-grazed (MCP: S = 108.0, $P = 0.02$; Kernel: S = 110.0, $P = 0.01$) and unburned-ungrazed (MCP: S = 118.0, $P = 0.02$; Kernel: S = 114.0, $P = 0.01$) treatments. Home ranges in the summer-burned-grazed treatment also were smaller than those in the unburned-grazed treatment, but only for adaptive kernel home ranges (S = 105.0, $P = 0.04$). Other treatment comparisons were not different.

Home ranges were larger in 2002 than in 2003 according to MCP (S = 333.0, $P < 0.01$) and Kernel (S = 337.0, $P < 0.01$). This difference was probably due to the larger number of missing lizards in 2003 (12) compared to 2002 (3). Home range size did not vary by sex (MCP: S = 307.0, $P = 0.91$; Kernel: S = 285.0, $P = 0.53$).

Survival Rates

The overall survival rate during the study was $s = 0.74$ [95% CI = 0.59-0.90] (method 1) and $s = 0.44$ [0.28-0.60] (method 2; $n = 51$ lizards) for the Kaplan-Meier method and $s = 0.74$ [95% CI = 0.60-0.91] (method 1) and $s = 0.45$ [0.32-0.64] (method 2; $n = 51$ lizards) for the Heisey-Fuller method. In general, survival rates of lizards were higher on sites 3-4-years post-burning than unburned sites. According to the Kaplan-Meier method, endpoint survival in the burned sites (method 1: $s = 0.93$ [95% CI = 0.79-1.00]; method 2: $s = 0.76$ [0.56-0.97], $n = 19$ lizards) was higher (method 1: $P = 0.01$; method 2: $P < 0.01$) than in all unburned sites (method 1: $s = 0.62$ [95% CI = 0.39-0.84]; method 2: $s = 0.28$ [0.12-0.46], $n = 32$). The shapes of the survival curves also differed between burning treatments (method 1: $\chi^2 = 5.9$, $P = 0.02$; method 2: $\chi^2 = 11.2$, $P < 0.01$; Figure 3), with survival declining in early summer in the unburned areas and in late summer in the burned areas. Burned-grazed sites (see burned sites above) showed a trend

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towards higher survival (method 1: $P = 0.28$; method 2: $P = 0.07$) than unburned-grazed sites (method 1: $s = 0.83$ [95% CI = 0.54-1.00]; method 2: $s = 0.34$ [0.00-0.86], $n = 8$ lizards), but only for method 2. There was a trend towards a difference in the shape of the survival curves for method 2 ($\chi^2 = 3.7$, $P = 0.06$) with survival declining earlier in the summer in the unburned-grazed areas. This trend is an artifact of a single mortality in the unburned-grazed area early in the summer. Endpoint survival in the winter-burned sites (method 1: $s = 1.0$; method 2: $s = 1.0$, $n = 8$ lizards) and the summer-burned site (method 1: $s = 0.83$ [95% CI = 0.54-1.13]; method 2: $s = 0.60$ [0.29-0.91], $n = 11$) could not be statistically compared with Kaplan-Meier because no lizards died in the winter-burned treatment. No difference was found between survival or shape of the survival curve in the summer-burned-grazed sites and the unburned-grazed sites.

Endpoint survival in the grazed sites (method 1: $s = 0.90$ [95% CI = 0.76-1.03]; method 2: $s = 0.67$ [0.47-0.88], $n = 27$ lizards) was higher (method 1: $P = 0.014$; method 2: $P = 0.001$) than in ungrazed sites (method 1: $s = 0.57$ [95% CI = 0.31-0.83]; method 2: $s = 0.24$ [0.06-0.43], $n = 24$), but the shapes of the survival curves did not differ (method 1: $\chi^2 = 0.71$, $P = 0.40$; method 2: $\chi^2 = 0.57$, $P = 0.45$; Figure 4).

Results were similar using the alternative Heisey-Fuller method to estimate survival (Table 3). Survival was greater in burned areas than in unburned areas (method 1: $P = 0.029$; method 2: $P = 0.008$), in winter-burned areas than in summer-burned (method 2 only: $P = 0.006$), in burned-grazed than in unburned-grazed (method 2 only: $P = 0.002$), in grazed areas than ungrazed (method 1: $P = 0.038$; method 2: $P = 0.007$). Survival rates in the summer-burned-grazed sites showed a trend towards higher survival

than unburned-grazed sites (method 2: $P = 0.069$). Other comparisons by treatment, year, or sex were not different.

Lizard Selection of Woody Plants

Lizards selected for *Aloysia gratissima* and avoided *Diospyros texana*, *Prosopis glandulosa*, and *Karwinskia humboldtiana* consistently across treatments (Table 4, Figure 5). Although not significant, there was a trend toward selection for *Colubrina texensis* (Table 4). Selection or avoidance of several other woody species was not consistent among treatments (Table 4). *Acacia* was selected in both burning treatments but avoided in the unburned-grazed treatment (Figure 5). *Condalia hookeri* and *Zanthoxylum fagara* were avoided in both burning treatments (Figure 5). *Opuntia engelmannii* was preferred except in the summer-burned-grazed treatment (Figure 5).

Analysis of use of woody plant diversity provided contrasting results when calculations were made using pooled data vs. the per-lizard basis. Lizards generally used (and had available) a lower richness and diversity of woody species on the 2 burned treatments, based on all lizard locations (Table 5). Species richness, Shannon-Wiener (H) diversity, and Simpson's diversity all differed by treatment ($\chi_3^2 = 12.73$, $P < 0.01$; $\chi_3^2 = 12.88$, $P < 0.01$; $\chi_3^2 = 15.00$, $P < 0.01$, respectively) when calculations were made on a per lizard basis. All 3 measures of diversity for woody species used by horned lizards were greater in the winter-burned-grazed treatment than all other treatments ($P < 0.02$; Figure 6, Table 6). Other treatment comparisons were not different.

Motte

No difference was found between landscape metrics (mean patch size, edge density, mean patch edge, mean perimeter-area ratio, and open area-shrub cover ratio) of

95% MCP lizard home ranges and metrics of core areas (50% MCP home ranges; $P \geq$ 0.45). No difference was found between landscape metrics of 95% MCP home ranges and metrics of the entire mapped area ($P \geq 0.40$). No difference was found between landscape metrics of core areas and the entire mapped area ($P \geq 0.20$), except for a trend towards a larger mean perimeter-area ratio in core areas compared to the whole mapped area ($P = 0.07$; Table 7).

DISCUSSION

Home Range

The results supported my prediction that lizards in summer-burned areas would have smaller home ranges than unburned areas due to the opportunity to find prey and cover in a smaller area. Burrow (2000) noted a similar result for winter burning. However, my prediction that lizards in summer-burned areas would have larger home ranges than lizards in winter-burned areas was not supported. Indeed, my data provided the opposite result, suggesting summer burning does not create too much open area for horned lizards. This result may be due to vegetation recovery since burning. Both summers during this study were relatively wet and cool compared to previous years (Figure 7), resulting in uniformly high herbaceous cover in all study areas. Higher precipitation combined with the timing of the study (3-4 years post-burning), may have eliminated most fire effects on vegetative cover seen in Burrow et al. (2001) after 1-2 years. Therefore, if cover was approximately equal across treatments, the smaller home ranges may be due to greater prey availability.

Home range size is inversely related to resource availability and dispersion for many species, including several lizards (Mares et al. 1976; Litvaitis et al. 1986; Boutin

1990; Lacher and Mares 1996). Ferguson et al. (1983) found that *Sceloporus undulatus garmani* hatchlings given supplemental food established smaller home ranges than hatchlings on control plots. Simon (1975) found a reduction in territory size of *Sceloporus jarrovi* after supplemental feeding and that natural food abundance was inversely correlated with territory size. Although Guyer (1988a,b) did not find decreased home range size for the lizard *Norops humilis* when given supplemental food, he reported increased overlap of home ranges and increased density in the supplemented population. Stamps and Tanaka (1981) found an initial decrease in the size of home ranges of juvenile *Anolis aeneus* after supplemental feeding, but found no difference in home range size after long-term feeding or in relation to food abundance studies in the laboratory. Waldschmidt (1983) found no difference in home range size between lizards given supplemental food and those that were not, but he found that fed lizards' home ranges grew at a slower rate than unfed lizards.

Horned lizards, and their primary food source, harvester ants (Whitford and Bryant 1979), can be affected by changes in vegetation structure caused by fire. Burning may improve resource distribution for horned lizards, and thereby reduce home range size, by creating a mixture of open areas for foraging with scattered patches of woody vegetation for cover, as suggested by Burrow (2000). Fires generally reduce shrub canopy cover and leaf litter, increasing bare ground and herbaceous forage (Wright and Bailey 1982, Dunne et al. 1991, McPherson 1995). Open areas allow for greater ease of movement by horned lizards (Whiting et al. 1993), which improves foraging efficiency (Pianka 1966) on harvester ants. A study by Fair and Henke (1997) reported that horned lizards used burned plots preferentially, perhaps because of ease of movement due to

reduced ground litter. A fire that increases grass or forb production would improve seed availability for harvester ants, which are granivores (Rissing 1981) and provide the open areas that harvester ant queens prefer to establish new mounds (DeMers 1993), thus increasing prey abundance for horned lizards. Also, horned lizards can more easily thermoregulate in open areas due to direct access to solar radiation for basking (Heath 1965).

Burrow (2000) reported home ranges of Texas horned lizards that ranged from 0.02 to 11.05 ha for 95% MCP and 0.02 to 14.63 ha for 95% adaptive kernel at the same site as the present study. Approximate average of home ranges (95% MCP), pooled across treatments, was 1.46 ha in the active season (15 April – 30 June) and 0.49 ha in the inactive season (1 July – 15 August), when calculated from Burrow (2000). These data were similar to my results, except that my results were biased by one very active lizard with a 95% MCP home range of 23.26 ha, which extended the upper portion of my ranges (Table 1). Home range sizes of Texas horned lizards in Arizona (Munger 1984) averaged 1.38 ha for females ($n = 13$) and 2.40 ha for males ($n = 10$), similar to my results (Table 1). Home ranges in my study were noticeably larger than the 0.03 to 1.47 ha ($n = 16$) described by Fair and Henke (1999) for Texas horned lizards in southern Texas, but their home range estimates were based on limited sampling. In a related species, Turner and Medica (1982) reported range size for male and female flat-tailed horned lizards in California as 0.13 ha ($n = 5$) and 0.05 ha ($n = 4$), respectively, whereas Wone and Beauchamp (2003) reported an average of 1.78 ha ($n = 45$) for males and 0.90 ($n = 24$) for females of the same species in the same counties. Smaller home range estimates by Turner and Medica (1982) were likely due to small sample sizes (3-7

captures per lizard) and calculation of home ranges from capture locations as opposed to sampling by radiotelemetry in the second study, which produces more even sampling.

Survival

The data tended to support my prediction that lizards in summer-burned areas would have higher survival rates than unburned areas due to the opportunity to find prey and cover in a smaller area, suggesting a positive burning effect. This contrasts with findings by Kazmaier et al. (in review) for 1-2 years post-winter-burning, in which survival rates were not affected by burning. My prediction that lizards in summer-burned areas would have lower survival rates than lizards in winter-burned areas because summer burning may create more open area than the ideal mosaic for horned lizards also was supported. However, this result was not consistent with patterns in home range size. As mentioned earlier, given the length of time since burning and high precipitation, effects of fire on herbaceous cover were likely eliminated. Therefore, if herbaceous cover was approximately equal across treatments, survival may have been lower on the summer-burned site than the winter-burned site due to differences in shrub cover. Hotter summer burns are speculated to produce greater reductions in brush cover compared to winter burning (Scifres and Hamilton 1993). However, given that Ruthven et al. (2003) did not find any difference in shrub cover between the two methods of prescribed burning on the Chaparral Wildlife Management Area, the survival differences may be due to probable shrub cover structural differences between treatments, which I will discuss below in relation to woody vegetation.

The higher survival in grazed than ungrazed sites is contrary to the results from prior study at Chaparral WMA (Burrow 2000, Kazmaier et al. in review). Grazing was

suspended for the 2002-2003 season, which means that the area had not been grazed for a year before the 2003 data were collected. Time since grazing may have lessened any detrimental grazing effects. The effect of grazing on other lizards is not clear. Reynolds (1979) reported that grazed areas supported more short-horned lizards (*Phrynosoma douglassi*) than ungrazed areas due better basking conditions. However, Jones (1981) found decreased lizard abundance due to the vegetation changes caused by grazing.

Estimates of survival rates of horned lizards are problematic because of the large number of censored (e.g., missing) lizards. Pianka and Parker (1975) implied adult Texas horned lizards had relatively high survival rates, but reported no data. Munger (1986) observed that Texas horned lizards in southeastern Arizona had seasonal survival rates between 35 and 86%, whereas Fair and Henke (1999) estimated 8-month survival rates (Mar-Oct) in southern Texas to be lower (8.9-54.0%). Estimates by Fair and Henke (1999) assumed constant daily survival over time, whereas my estimates were 3-month (15 May-15 August) rates. However, when daily survival rates calculated by Fair and Henke (1999) are converted to 3-month summer survival rates (39-76%), they are similar to my results (45-74%; Heisey-Fuller). For the 1998-2000 period on the Chaparral area, Burrow (2002) reported 4-month summer survival rates ranging from 25% to 62%. In a related species, Munger (1986) reported seasonal survival rates for *Phrynosoma modestum* between 3% and 64%.

Lizard Selection of Woody Plants

Thermal and escape cover may be driving factors in the use of woody vegetation by horned lizards. Previous studies of lizards and fire have found changes in species abundance and community composition related to changes in habitat structure after fire

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(Fyfe 1980, Mushinsky 1985, Braithwaite 1987, Mushinsky 1992). Selection for *Acacia* in the burned areas and avoidance or equal use in unburned areas may be due to changes in their growth form due to burning. Burning often top-kills acacias, causing resprouting from the base (Rasmussen et al. 1983, Ruthven et al. 2003). Resprouting results in a denser canopy at ground level, which would provide better thermal and escape cover for the lizards. *Condalia hookeri* and *Zanthoxylum fagara* appeared to be avoided in winter-burned areas, but this effect may be confounded by their tendency to occur in mottes with other species that might have been recorded as the dominant. *Zanthoxylum fagara* is known to sprout from basal stems after fire (Flinn 1986); therefore avoidance of this species in a burned area is puzzling.

Prosopis glandulosa, although used less than its availability, was still used quite often and was one of the most abundant species on the area. On this study area, *Prosopis glandulosa* is often large enough to avoid top-kill (Ruthven et al. 2003), so it shows limited structural change in regrowth after a fire, and is therefore unlikely to show a change in selection across treatments. This species may appear to be avoided because lizards were using the plants that developed under and around its canopy. It is often the pioneer tree around which other trees and shrubs cluster to form a motte (Archer et al. 1988). When I recorded the woody species covering the lizard, the woody species under the *Prosopis glandulosa* would generally cover more of the lizard and therefore was recorded as dominant. *Diospyros texana* may be avoided because its growth form leaves open ground beneath its canopy (*personal observation*), which provides less thermal protection and allows predators to more easily spot a lizard. When it forms a motte, generally only leaf litter is found beneath it due to its dense canopy.

Aloysia gratissima may be used greater than its availability because it is often found in slightly wetter areas, such as drainages, and it tends to grow in dense thickets (Taylor et al. 1999), which would provide plenty of thermal and escape cover. *Colubrina texensis* has a low, dense canopy, and *Opuntia engelmannii* possessses many spines and fallen pads (Taylor et al. 1999). These structural characteristics provide a good source of cover for small animals (Taylor et al. 1999) and may explain preference for these plants.

I suggest two alternative scenarios to explain the greater richness and diversity of woody vegetation used by lizards in the winter-burned-grazed treatment. One alternative is that horned lizards are using more of the available species to replace another preferred species that is lacking there, or to compensate for less available cover. However, Ruthven et al. (2003) found that woody species richness did not differ between unburned, winter-burned, and winter-summer burned rangelands at this management area.

A second, more compelling alternative follows from observations that fire may change vegetation structure without altering species composition. For example, Harrell et al. (2001) studied the effects of fire on vegetation structure in shinnery oak communities and found decreases in shrub cover, vegetation height, and visual obstruction, leading to increased structural openness in the first and second growing seasons after fire. However, they found no differences in species composition. In addition, the change in growth form (mentioned earlier) of *Acacia* in response to fire illustrates how fire can affect structure. Other shrub species on the study area may show similar responses to fire. Animal species diversity has been positively correlated with plant stuctural diversity (as opposed to species diversity) in several taxa (reviewed in Tews et al. 2004) including birds (MacArthur and MacArthur 1961, MacArthur 1964)

and lizards (Pianka 1966). Studies at the individual-species level conducted in white-footed mice (*Peromyscus leucopus*) have found distinct correlations between foliage structure and mouse density, activity, and habitat use (M'Closkey 1975, M'Closkey and Lajoie 1975).

I propose that structural changes of the habitat in response to fire may provide more thermal and escape cover at the lizard level, allowing lizards to use a greater richness and diversity of woody species in the winter-burned treatment, although a similar species composition was present in all treatments. Summer-burned areas may not show this same trend due to harsher burning conditions. Future studies of horned lizard habitat use should focus on vegetation structural characteristics, such as branch density $\leq$ 1 m from the ground, instead of vegetation species.

Motte

My prediction that horned lizards would have more edge in their core (50% MCP) home ranges than their overall (95% MCP) home ranges was not supported. However, a trend towards a larger mean perimeter-area ratio in 50%MCPs compared to the whole mapped area suggested support for the idea that horned lizards prefer more edge in their areas of greatest use, because perimeter-area ratio is one measure of edge. Smaller or jagged mottes have more edge than large or circular mottes. The juxtaposition of open areas and shrub cover has been identified as an important habitat variable for the Texas horned lizard (Burrow et al. 2001). Unfortunately, due to small sample size (n = 2), I was unable to draw more complex conclusions regarding comparisons of habitat and survival or home range size. If applied to a larger area and a larger sample size, I believe this could be a scale-appropriate technique for analysis of horned lizard habitat use.

Conclusions

Smaller home ranges in the summer-burned treatment and higher survival in burned areas (particularly winter-burned) indicated a beneficial effect of burning but failed to provide a clear answer as to which burning prescription (winter or summer) was more beneficial. Ant activity, used as a surrogate of ant abundance, has been shown previously to be higher on burned sites on our study area. Therefore, more food or better food-cover interspersion may explain the higher survival in burned areas and the smaller home ranges in summer-burned areas. I conclude that prescribed burning in a thornscrub savanna provided favorable ecological conditions for Texas horned lizards. Longer-term data is needed to clarify differences in the effects of summer and winter burning.

LITERATURE CITED

Alford, R. A., and S. J. Richards. 1999. Global amphibian declines: a problem in applied ecology. Annual Review of Ecology and Systematics 30:133-165.

Andersen, A. N. 1988. Immediate and longer-term effects of fire on seed predation by ants in sclerophyllous vegetation in south-eastern Australia. Australian Journal of Ecology 13:285-293.

Archer, S. R., C. J. Scifres, C. R. Bassham, and R. C. Maggio. 1988. Autogenic succession in a subtropical savanna: conversion of grassland to thorn woodland. Ecological Monographs 58:111-127.

Ballinger, R. E. 1974. Reproduction of the Texas horned lizard, *Phrynosoma cornutum*. Herpetologica 30:321-327.

Blackshear, S. D., and J. V. Richerson. 1999. Ant diet of the Texas horned lizard (*Phrynosoma cornutum*) from the Chihuahuan desert. Texas Journal of Science 51:147-152.

Bock, C. E., and J. H. Bock. 1990. Effects of fire on wildlife in southwestern lowland habitats. Pages 50-64 *in* J. S. Krames, technical coordinator. Effects of fire management of southwestern natural resources. USDA Forest Service General Technical Report RM-191.

Boutin, S. 1990. Food supplementation experiments with terrestrial vertebrates: patterns, problems, and the future. Canadian Journal of Zoology 68:203-220.

Box, T. W., J. Powell, and D. L. Drawe. 1967. Influence of fire on South Texas chaparral communities. Ecology 48:955-961.

Box, T. W., and R. S. White. 1969. Fall and winter burning of south Texas brush ranges. Journal of Range Management 22:373-376.

Braithwaite, R. W. 1987. Effects of fire regimes on lizards in the wet-dry tropics of Australia. Journal of Tropical Ecology 3:265-275.

Burrow, A. L. 2000. The effect of prescribed burning and grazing on the threatened Texas horned lizard (*Phrynosoma cornutum*) in the western Rio Grande Plains. M.S. Thesis, Oklahoma State University, Stillwater.

Burrow, A. L., R. T. Kazmaier, E. C. Hellgren, and D. C. Ruthven. 2001. Microhabitat selection by Texas horned lizards in southern Texas. Journal of Wildlife Management 65:645-652.

Cable, D. R. 1965. Damage to mesquite, Lehmann lovegrass, and black grama by a hot June fire. Journal of Range Management 18:326-329.

_____ . 1967. Fire effects on semi-desert grasses and shrubs. Journal of Range Management 20:170-176.

_____ . 1973. Fire effects in southwestern semidesert grass-shrub communities. Proceedings of the Tall Timbers Fire Ecology Conference 12:109-127.

Conant, R., and J. T. Collins. 1998. A field guide to reptiles and ampibians of eastern and central North America. Third edition, expanded. Houghton Mifflin, New York.

DeMers, M. N. 1993. Roadside ditches as corridors for range expansion of the western harvester ant (*Pogonomyrmex occidentalis*). Landscape Ecology 8:93-102.

Donaldson, W., A. H. Price, and J. Morse. 1994. The current status and future prospects of the Texas horned lizard (*Phrynosoma cornutum*) in Texas. Texas Journal of Science 46:97-113.

Dunne, J., A. Dennis, J. W. Bartolome, and R. H. Barrett. 1991. Chaparral response to a prescribed fire in the Mount Hamilton Range, Santa Clara County California. Madrono 38:21-29.

Elkie, P., R. Rempel, and A. Carr. 1999. Patch Analyst User's Manual. Ont. Min. Natur. Resour. Northwest Sci. & Technol. Thunder Bay, Ont. TM–002. 16 pp + Append.

Engle, D. M., and T. G. Bidwell. 2001. Viewpoint: The response of central North American prairies to seasonal fire. Journal of Range Management 54:2-10.

Fair, W. S., and S. E. Henke. 1997. Effects of habitat manipulations on Texas horned lizards and their prey. Journal of Wildlife Management 61:1366-1370.

Fair, W. S, and S. E. Henke. 1999. Movements, home ranges, and survival of Texas horned lizards (*Phrynosoma cornutum*). Journal of Herpetology 33:517-525.

Ferguson, G. W., J. L. Huges, and K. L. Brown. 1983. Food availability and territorial establishment of juvenile *Sceloporus undulatus*. Pages 134-148 *in* R. B. Huey, E. R. Pianka, and T. W. Schoener, editors. Lizard ecology: studies of a model organism. Harvard University Press, Cambridge, Massachusetts.

Flinn, R. C. 1986. Vegetation regeneration in South Texas shrubs. M. S. thesis, Texas A&M University, College Station.

Ford, W. M., M. A. Menzel, D. W. McGill, J. Laerm, and T. S. McCay. 1999. Effects of a community restoration fire on small mammals and herpetofauna in the southern Appalachians. Forest Ecology and Management 114:233-243.

Fyfe, G. 1980. The effects of fire on lizard communities in central Australia. Herpetofauna 12:1-9.

Gibbons, W. J., D. E. Scott, T. J. Ryan, K. A. Buhlmann, T. D. Tuberville, B. S. Metts, J.

L. Greene, T. Mills, Y. Leiden, S. Poppy, and C. T. Winne. 2000. The global

decline in reptiles, déjà vu amphibians. BioScience 50:653-666.

Griffiths, A. D., and K. A. Christian. 1996. The effects of fire on the frillneck lizard

(*Chlamydosaurus kingii*) in northern Australia. Australian Journal of Ecology

21:386-398.

Guyer, C. 1988a. Food supplementation in a tropical mainland anole, *Norops humilis*:

effects on individuals. Ecology 69:362-369.

Guyer, C. 1988b. Food supplementation in a tropical mainland anole, *Norops humilis*:

demographic effects. Ecology 69:350-361.

Harrell, W. C., S. D. Fuhlendorf, and T. G. Bidwell. 2001. Effects of prescribed fire on

sand shinnery oak communities. Journal of Range Management 54:685-690.

Hatch, S. L., K. N. Gandhi, and L. E. Brown. 1990. Checklist of the vascular plants of

Texas. Texas A&M University system, College Station, Texas, USA.

Heath, J. E. 1965. Temperature regulation and diurnal activity in horned lizards.

University of California Publications in Zoology 64:97-136.

Heisey, D. M., and T. K. Fuller. 1985. Evaluation of survival and cause-specific mortality

rates using telemetry data. Journal of Wildlife Management 49:668-674.

Henke, S. E. 1998. The effect of multiple search items and item abundance on the

effectiveness of human searchers. Journal of Herpetology 32:112-115.

Hooge, P. N., and B. Eichenlaub. 1997. Animal movement extension to ArcView.

Version 1.1 Alaska Science Center – Biological Science Office, U.S. Geological

Survey, Anchorage, Alaska, USA.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments.

Ecological Monographs 54:187-211.

Jackson, G. P., and B. J. Fox. 1996. Comparison of regeneration following burning, clearing or mineral sand mining at Tomago, NSW: II. Succession of ant assemblages in a coastal forest. Australian Journal of Ecology 21:200-216.

Jones, B. K. 1981. Effects of grazing on lizard abundance and diversity in western Arizona. The Southwestern Naturalist 26:107-115.

Kahn, W. C. 1960. Observations on the effect of a burn on a population of *Sceloporus occidentalis*. Ecology 41:358-359.

Kernohan, B. J., R. A. Gitzen, and J. J. Millspaugh. 2001. Analysis of animal space use and movements. Pages 125-166 *in* J. J. Millspaugh and J. M. Marzluff, editors. Radio tracking and animal populations. Academic Press, New York.

Komarek, E. V. Sr. 1966. The meteorological basis for fire ecology. Proceedings of the Tall Timbers Fire Ecology Conference 5:85-125.

Krebs, C. J. 1978. Ecology: the experimental analysis of distribution and abundance. Second edition. Harper & Row, New York.

Lacher, T. E., Jr., and M. A. Mares. 1996. Availability of resources and use of space in eastern chipmunks, *Tamias striatus*. Journal of Mammalogy 77:833-849.

Lawrence, G. E. 1966. Ecology of vertebrate animals in relation to chaparral fire in the Sierra Nevada foothills. Ecology 47:278-291.

Litvaitis, J. A., J. A. Sherburne, and J. A. Bissonette. 1986. Bobcat habitat use and home range size in relation to prey density. Journal of Wildlife Management 50:110-117.

Lillywhite, H. B., and F. North. 1974. Perching behavior of *Sceloporus occidentalis* in

recently burned chaparral. Copeia 1974:256-257.

MacArthur, R. H. 1964. Environmental factors affecting bird species diversity. American Naturalist 98:387-397.

MacArthur, R. H., and J. W. MacArthur. 1961. On bird species diversity. Ecology 42:594-598.

Mares, M. A., M. D. Watson, and T. E. Lacher, Jr. 1976. Home range perturbations in *Tamias striatus*. Oecologia 25:1-12.

Martin, S. C. 1983. Responses of semi-desert grasses and shrubs to fall burning. Journal of Range Management 36:604-610.

McCoy, E. D., and E. W. Kaiser. 1990. Changes in foraging activity of the southern harvester ant *Pogonomyrmex badius* (Latreille) in response to fire. American Midland Naturalist 123: 112-123.

M'Closkey, R. T. 1975. Habitat dimensions of white-footed mice, *Peromyscus leucopus*. American Midland Naturalist 93:158-167.

M'Closkey, R. T., and D. T. Lajoie. 1975. Determinants of local distribution and abundance in white-footed mice. Ecology 56:467-472.

McPherson, G. R. 1995. The role of fire in the desert grasslands. Pages 130-151 *in* M. P. McClaran and T. R. Van Devender, editors. The desert grassland. University of Arizona Press, Tucson, Arizona, USA.

Means, D. B., and H. W. Campbell. 1981. Effects of prescribed fire on amphibians and reptiles. Pages 89-97 *in* G. W. Wood, editor. Prescribed fire and wildlife in Southern forests. The Belle W. Baruch Forest Science Institute of Clemson University, Georgetown, South Carolina, USA.

Mohr, C. O. 1947. Table of equivalent populations of North American mammals. American Midland Naturalist 37:223-249.

Munger, J. C. 1984. Home ranges of horned lizards (*Phrynosoma*): circumscribed and exclusive? Oecologia 62:351-360.

Munger, J. C. 1986. Rate of death due to predation for two species of horned lizard, *Phrynosoma cornutum* and *P. modestum*. Copeia 1986:820-824.

Mushinsky, H. R. 1985. Fire and the Florida sandhill herpetofaunal community: with special attention to responses of *Cnemidophorus sexlineatus*. Herpetologica 41:333-342.

Mushinsky, H. R. 1992. Natural history and abundance of southeastern five-lined skinks, *Eumeces inexpectatus*, on a periodically burned sandhill in Florida. Herpetologica 48:307-312.

Oksanen, L. Logic of experiments in ecology: is pseudoreplication a pseudoissue? Oikos 94:27-38.

Pase, C. P. 1971. Effect of a February fire on Lehmann lovegrass. Journal of Range Management 24:454-456.

Patterson, G. B. 1984. The effect of burning-off tussock grassland on the population density of common skinks. New Zealand Journal of Zoology 11:189-194.

Pianka, E. R. 1966. Convexity, desert lizards, and spatial heterogeneity. Ecology 47:1055-1059.

Pianka, E. R. 1989. Desert lizard diversity: additional comments and some data. American Naturalist 134:344-364.

Pianka, E. R. 1996. Long-term changes in lizard assemblages in the Great Victorian Desert: dynamic habitat mosaics in response to wildfire. Pages 191-215 *in* M. L. Cody and J. L. Smallwood, editors. Long-term studies of vertebrate communities. Academic Press, San Diego, California.

Pianka, E. R., and W. S. Parker. 1975. Ecology of horned lizards: a review with special reference to *Phrynosoma platyrhinos*. Copeia 1975:141-162.

Pollock, K. H., S. R. Winterstein, C. M. Bunck, and P. D. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

Price, A. H. 1990. *Phrynosoma cornutum* (Harlan) Texas horned lizard. Catalogue of American Amphibians and Reptiles 469: 1-7.

Rasmussen, G. A., C. J. Scifres, and D. L. Drawe. 1983. Huisache growth, browse quality, and use following burning. Journal of Range Management 36:337-342.

Rempel, R.S. and A. P. Carr. 1999. Patch Analyst extension for ArcView: version 2.2. http://flash.lakeheadu.ca/~rrempel/patch/index.html

Reynolds, T. D. 1979. Response of reptile populations to different land management practices on the Idaho National Engineering Laboratory site. Great Basin Naturalist 39:255-262.

Rissing, S. W. 1981. Prey preferences in the desert horned lizard: influence of prey foraging method and aggressive behavior. Ecology 62:1031-1040.

Ruthven, D. C., J. F. Gallagher, and D. R. Synatzske. 2000. Effect of fire and grazing on forbs in the western south Texas plains. The Southwestern Naturalist 45:89-94.

Ruthven, D. C., A. W. Braden, H. J. Knutson, J. F. Gallagher, and D. R. Synatzske. 2003. Woody vegetation response to various burning regimes in South Texas. Journal of Range Management 56:159-166.

Schmidt, P. J., W. C. Sherbrooke, and J. O. Schmidt. 1989. The detoxification of ant (*Pogonomyrmex*) venom by a blood factor in horned lizards (*Phrynosoma*). Copeia 1989: 603-607.

Scifres, C. J., and W. T. Hamilton. 1993. Prescribed burning for brushland management: the south Texas example. Texas A&M University Press, College Station, Texas, USA.

Shannon, C. E., and W. Weaver. 1949. The mathematical theory of communication. University of Illinois Press, Urbana.

Simon, C. A. 1975. The influence of food abundance on territory size in the iguanid lizard *Sceloporus jarrovi*. Ecology 56:993-998.

Simpson, G. H. 1949. Measurement of diversity. Nature 163:688.

Stamps, J. A., and S. Tanaka. 1981. The relationship between food and social behavior in juvenile lizards (*Anolis aeneus*). Copeia 1981:422-434.

Stiling, P. 1999. Ecology: theories and applications. Third edition. Prentice Hall, Upper Saddle River, New Jersey, USA.

Taylor, R. B., J. Rutledge, and J. G. Herrera. 1999. A field guide to common south Texas shrubs. Texas Parks and Wildlife Press, Distributed to the trade by University of Texas Press, Austin, Texas, USA.

Tews, J., U. Brose, V. Grimm, K. Tielborger, M. C. Wichmann, M. Schwager, and F. Jeltsch. 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. Journal of Biogeography 31:79-92.

Turner, F. B. and P. A. Medica. 1982. The distribution and abundance of the flat-tailed horned lizard (*Phrynosoma m'calli*). Copeia 1982:815-823.

Vogl, R. J. 1971. Smokey: a bearfaced lie. Ecology Today 1:14-17.

Waldschmidt, S. 1983. The effect of supplemental feeding on home range size and activity patterns in the lizard *Uta stansburiana*. Oecologia 57:1-5.

Whitford, W. G., and M. Bryant. 1979. Behavior of a predator and its prey: the horned lizard (*Phrynosoma cornutum*) and harvester ants (*Pogonomyrmex* spp.). Ecology 60: 686-694.

Whiting, M. J., J. R. Dixon, and R. C. Murray. 1993. Spatial distribution of a population of horned lizards (*Phrynosoma cornutum*: Phrynosomatidae) relative to habitat and prey. The Southwestern Naturalist 38:150-154.

Whittaker, R. H., L. E. Gilbert, and J. H. Connell. 1979. Analysis of two-phase pattern in a mesquite grassland, Texas. Journal of Ecology 67:935-952.

Wone, B., and B. Beauchamp. 2003. Movement, home range, and activity patterns of the horned lizard, *Phrynosoma mcallii*. Journal of Herpetology. 37:679-686.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164-168.

Wright, H. A., and A. W. Bailey. 1982. Fire ecology. John Wiley & Sons, New York.

Table 1. Home range sizes (ha) of Texas horned lizards by 2 estimators (95% Minimum Convex Polygon and 95% Fixed Kernel) for 15 May – 15 August on Chaparral Wildlife Management Area, summer 2002 and 2003.

Variable	Minimum Convex Polygon					Fixed Kernel				
	n	$\bar{x}$	SE	Min	Max	n	$\bar{x}$	SE	Min	Max
Female	34	2.50	0.78	0.06	23.26	34	3.04	0.78	0.09	21.59
Male	13	1.49	0.38	0.10	5.22	13	1.84	0.50	0.11	6.85
2002	25	3.52	1.01	0.47	23.26	25	4.22	0.99	0.43	21.59
2003	22	0.74	0.17	0.06	2.84	22	0.99	0.25	0.09	4.69
Overall	47	2.22	0.57	0.06	23.26	47	2.71	0.59	0.09	21.59

Table 2. Home range sizes (ha) of Texas horned lizards by 2 estimates (95% Minimum Convex Polygon and 95% Fixed Kernel) for

15 May – 15 August on Chaparral Wildlife Management Area, summer 2002 and 2003. Treatments are designated as UBUG

(unburned, ungrazed), UBG (unburned, grazed), SBG (summer burned, grazed), and WBG (winter burned, grazed).

Treatment	Minimum Convex Polygon					Fixed Kernel				
	n	$\bar{x}$	SE	Min	Max	n	$\bar{x}$	SE	Min	Max
SBG	11	0.73	0.28	0.06	2.76	11	0.89	0.36	0.09	3.65
UBG	8	1.71	0.60	0.19	5.22	8	2.17	0.74	0.23	6.85
UBUG	20	3.26	1.27	0.19	23.26	20	3.56	1.23	0.11	21.59
WBG	8	2.19	0.60	0.47	5.00	8	3.61	1.06	0.43	7.73

Table 3. Comparisons of survival rates (Heisey and Fuller 1985) of Texas horned lizards by treatment, year, and sex at Chaparral Wildlife Management Area, summers 2002 and 2003

Comparison		Method [a]	Survival Rate 1		Survival Rate 2			
Rate 1	Rate 2		s	σ	s	σ	Z	P-value
Unburned	Burned	Live	0.62	0.01	0.92	0.01	-2.19	0.029
Unburned	Burned	Dead	0.32	0.01	0.71	0.01	-2.67	0.008
Unburned-grazed	Burned-grazed	Live	0.62	0.04	0.92	0.00	-1.42	0.156
Unburned-grazed	Burned-grazed	Dead	0.19	0.01	0.71	0.01	-3.13	0.002
Summer-burned-grazed	Unburned	Live	0.86	0.02	0.62	0.01	1.42	0.156
Summer-burned-grazed	Unburned	Dead	0.55	0.03	0.32	0.01	1.28	0.201
Summer-burned-grazed	Unburned-grazed	Live	0.86	0.02	0.62	0.04	0.99	0.322
Summer-burned-grazed	Unburned-grazed	Dead	0.55	0.03	0.19	0.01	1.82	0.069
Grazed	Ungrazed	Live	0.88	0.01	0.57	0.02	2.07	0.038
Grazed	Ungrazed	Dead	0.65	0.01	0.26	0.01	2.68	0.007
Winter-burned	Summer-burned	Live	1.00	0.00	0.86	0.02	1.08	0.280
Winter-burned	Summer-burned	Dead	1.00	0.00	0.55	0.03	2.73	0.006
2002	2003	Live	0.67	0.01	0.84	0.01	-1.14	0.254
2002	2003	Dead	0.55	0.01	0.35	0.01	1.28	0.201
Male	Female	Live	0.87	0.02	0.70	0.01	1.09	0.276
Male	Female	Dead	0.65	0.03	0.40	0.01	1.41	0.159

[a] Two methods were used to calculate survival rates to account for the unknown fate of missing (censored) animals. In method 1, all censored animals were assumed live. In method 2, all censored animals were assumed dead.

Table 4. Analyses of selection of individual woody species across treatments[a] by Texas horned lizards at Chaparral Wildlife Management Area, summers 2002 and 2003

Species	Selection Effect (use vs availability)			Treatment*Selection Interaction		
	χ^2	df	P	χ^2	df	P
Acacia	5.60	1	0.018	48.83	3	<.0001
Aloysia gratissima	3.69	1	0.055	1.14	2*	0.566
Celtis pallida	0.00	1	0.970	8.02	3	0.046
Condalia hookeri	3.18	1	0.074	14.67	3	0.002
Colubrina texensis	3.38	1	0.066	2.99	3	0.393
Diospyros texana	20.95	1	<.0001	1.03	3	0.794
Karwinskia humboltiana	3.79	1	0.052	4.12	2*	0.127
Lantana	0.07	1	0.795	12.07	3	0.007
Opuntia engelmannii	18.55	1	<.0001	11.71	3	0.008
Opuntia leptocaulis	1.58	1	0.208	2.93	3	0.403
Other	11.62	1	0.001	42.07	3	<.0001
Prosopis glandulosa	9.07	1	0.003	1.53	3	0.675
Zanthoxylum fagara	0.00	1	0.955	12.33	2*	0.002

[a] Treatments were summer-burned-grazed, unburned-grazed, unburned-ungrazed, and winter-burned grazed.

Table 5. Richness and diversity of woody species available to and used by Texas horned lizards by treatment at Chaparral Wildlife Management Area, summers 2002 and 2003.

Measure	Summer-burned grazed		Unburned grazed		Unburned ungrazed		Winter-burned grazed	
	Available[a]	Used	Available	Used	Available	Used	Available	Used
Species richness	13	19	24	25	26	28	16	25
Shannon-Wiener (H)	2.045	2.387	2.524	2.598	2.358	2.221	2.223	2.672
Simpson (D)	0.856	0.885	0.886	0.899	0.853	0.827	0.858	0.896

[a] Availability based on 100-m transects (n = 9-18) sampled within each treatment.

Table 6. Pairwise comparisons by treatment of richness and diversity of woody species used by radioed Texas horned lizards at Chaparral Wildlife Management Area, summers 2002 and 2003. Treatment abbreviations are: SBG = summer-burned grazed; UBG = unburned grazed; UBUG = unburned ungrazed; and WBG = winter-burned grazed.

Treatment Comparison	Diversity Measure	Wilcoxon Rank Sum (W)	*P*-value
SBG vs WBG			
	Richness	41.5	0.005
	Shannon-Weiner, H	41.0	0.005
	Simpson, D	42.0	0.007
SBG vs UBUG			
	Richness	103.0	0.672
	Shannon-Weiner, H	111.0	0.349
	Simpson, D	113.0	0.287
SBG vs UBG			
	Richness	60.0	0.674
	Shannon-Weiner, H	56.5	1.000
	Simpson, D	56.5	1.000
WBG vs UBUG			
	Richness	145.0	0.002
	Shannon-Weiner, H	144.0	0.002
	Simpson, D	151.0	0.000
WBG vs UBG			
	Richness	36.0	0.020
	Shannon-Weiner, H	36.0	0.024
	Simpson, D	36.0	0.024
UBUG vs UBG			
	Richness	95.0	0.317
	Shannon-Weiner, H	99.0	0.205
	Simpson, D	100.5	0.169

Table 7. Landscape metrics for 50% MCP, 95% MCP, and entire mapped area in the unburned-ungrazed site at Chaparral Wildlife Management Area.

Landscape Metric	50% MCP (451)[a]	50% MCP (713)	95% MCP (451)	95% MCP (713)	Entire Area
Mean Patch Size (m^2)	8.64	3.58	6.85	4.59	6.66
Edge Density (m/ha)	11691.32	20100.89	12562.79	15994.18	12614.40
Mean Patch Edge (m/patch)	10.10	7.20	8.60	7.34	8.40
Mean Perimeter-Area Ratio (m/ha)	51719.02	47191.25	75418.07	36312.28	28775.16
Open Area-Shrub Cover Ratio	0.87	1.03	0.92	1.60	1.18

[a] Number represents lizard identification.

Fig. 1. Study sites by treatment (summer burned–grazed, winter burned–grazed, unburned–grazed, unburned–ungrazed) on the Chaparral Wildlife Management Area, Dimmit and LaSalle Counties, Texas, summer 2003-2003.

	Summer burned, grazed
	Winter burned, grazed
	Unburned, grazed
	Unburned, ungrazed

1000   0   1000 2000 3000 Meters
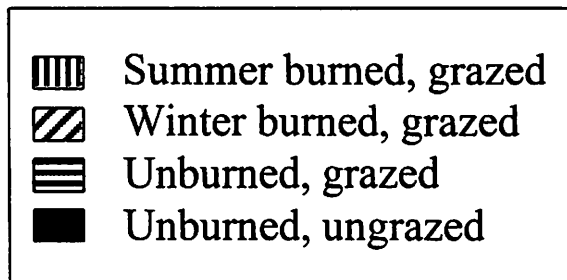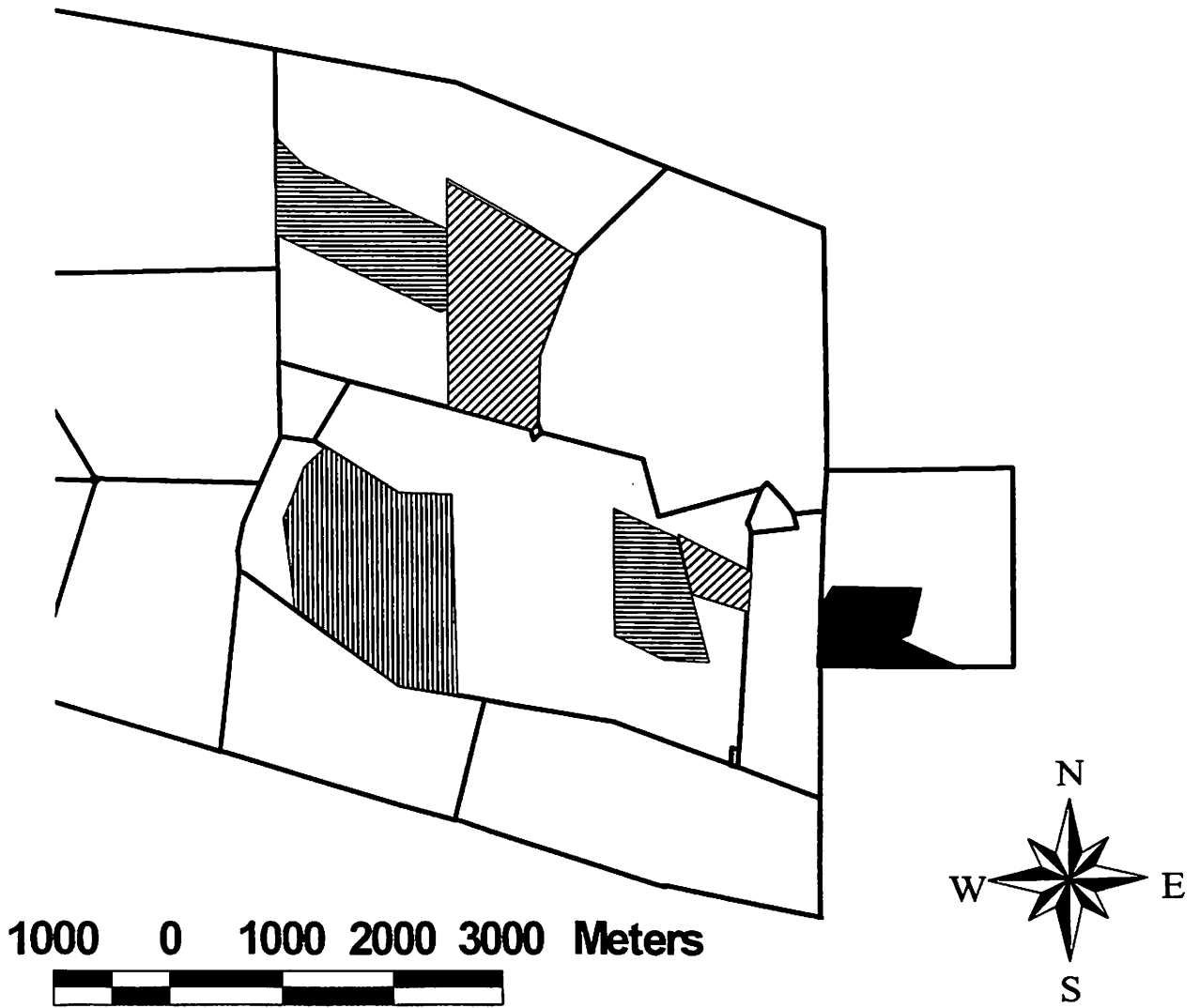
Fig. 2. Mean home range size of Texas horned lizards in summer burned – grazed, unburned – grazed, unburned – ungrazed, and winter burned – grazed study sites on the Chaparral Wildlife Management Area, Dimmit and LaSalle Counties, Texas, summer 2003-2003.
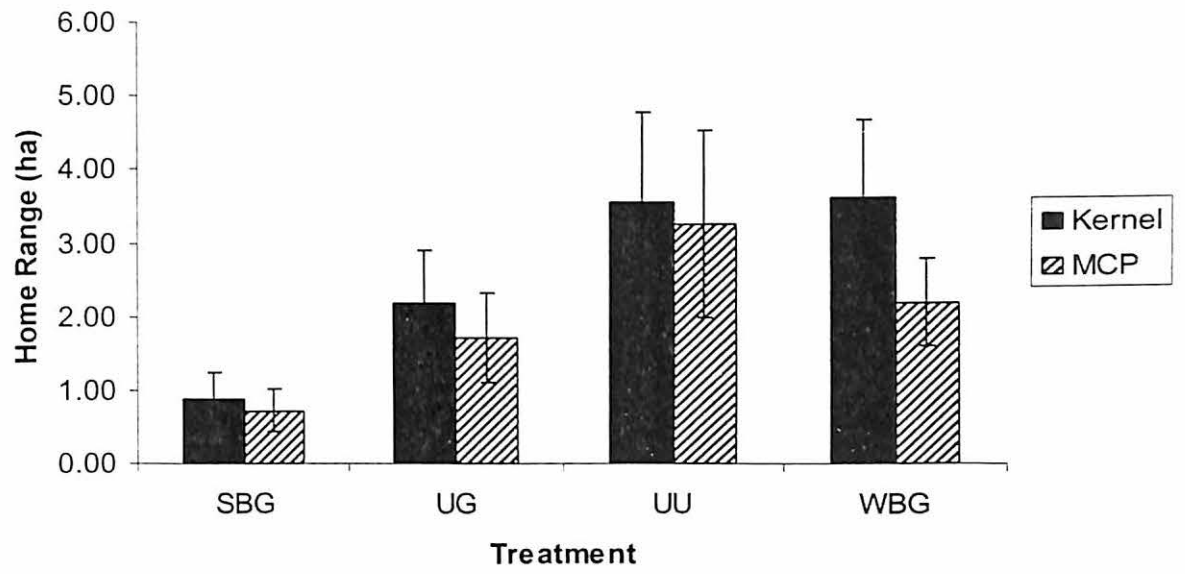
Fig. 3. Survival rates of Texas horned lizards in burned and unburned study sites on the Chaparral Wildlife Management Area, Dimmit and LaSalle Counties, Texas, summer 2003-2003. Censored animals assumed dead (top) and censored animals assumed live (bottom).
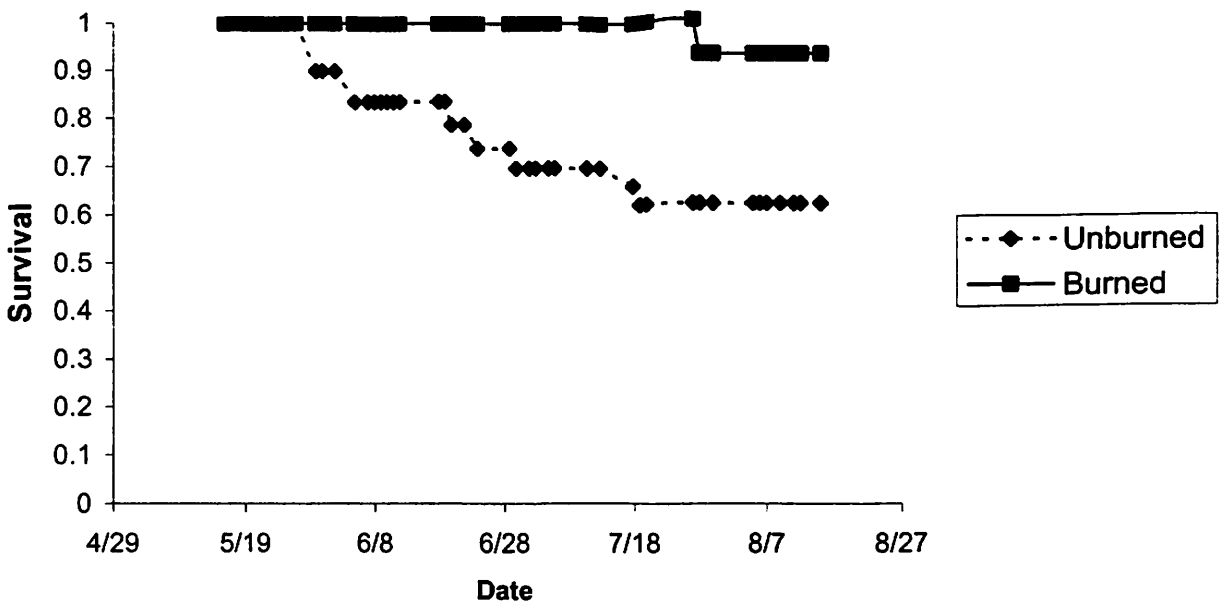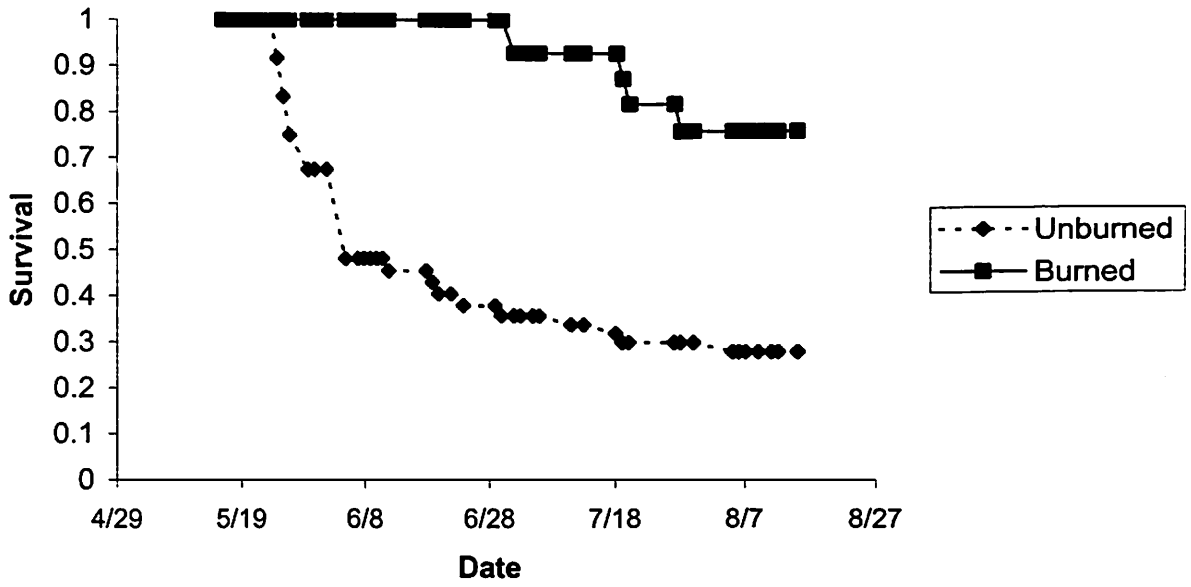
Fig. 4. Survival rates of Texas horned lizards in grazed and ungrazed study sites on the

Chaparral Wildlife Management Area, Dimmit and LaSalle Counties, Texas, summer

2003-2003. Censored animals assumed dead (top) and censored animals assumed live

(bottom).
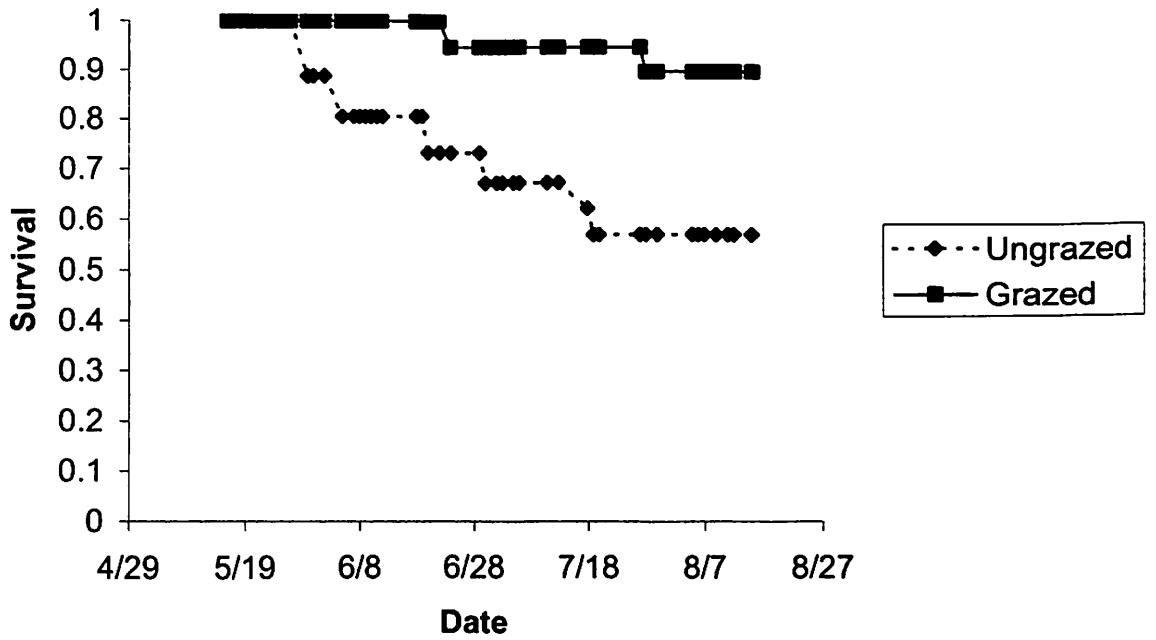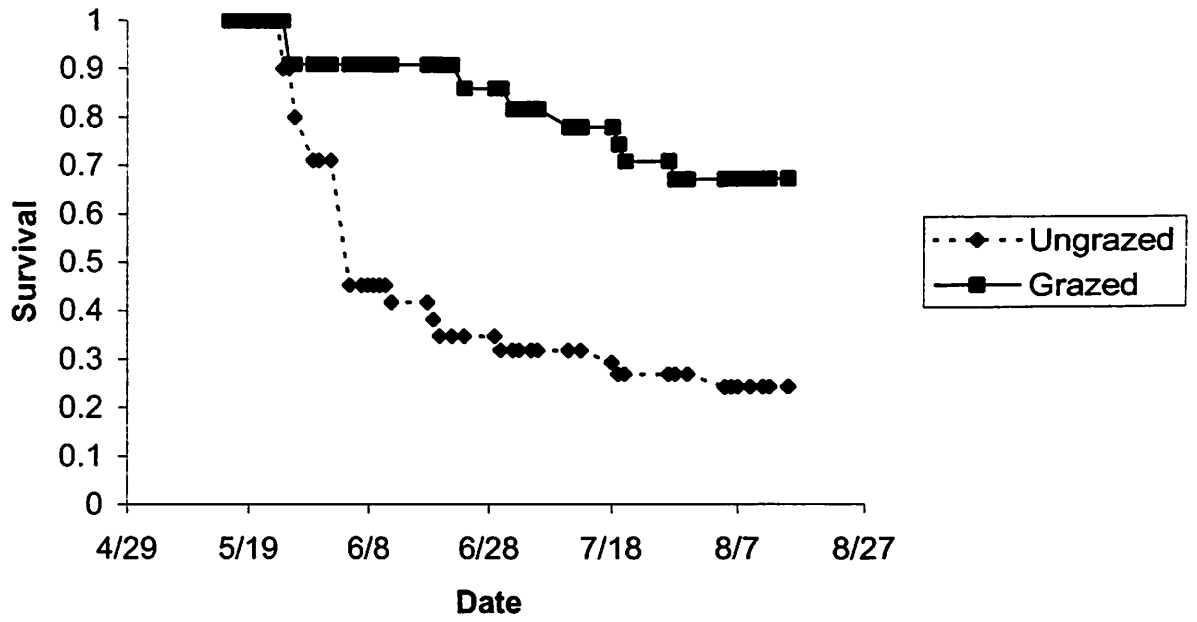
Fig. 5. Woody vegetation use versus availability by treatment for each of 12 species groups on Chaparral Wildlife Management Area, Dimmit and LaSalle Counties, Texas, summer 2003-2003.

Acacia
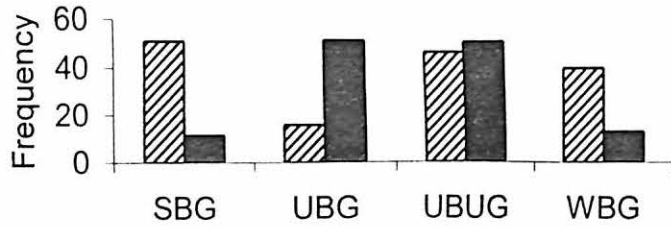
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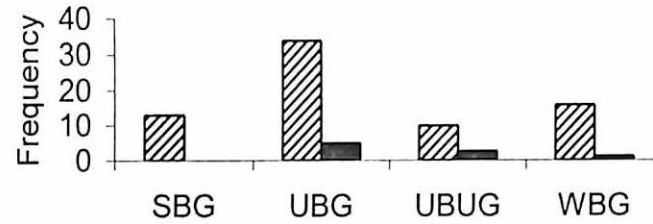
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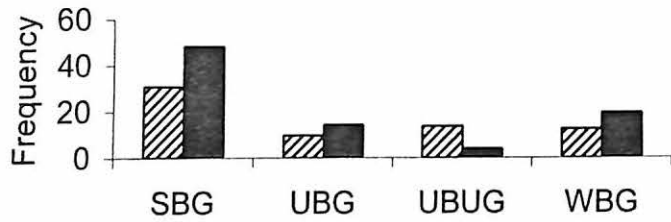
Condalia hookeri
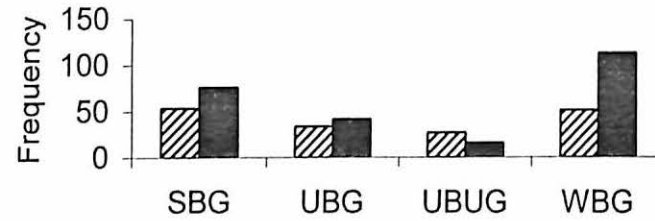
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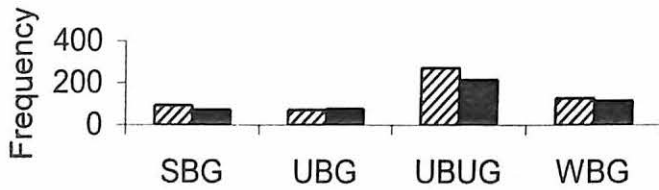
Diospyros texana
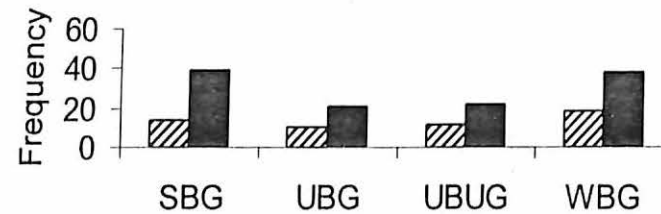
USE ▨ AVAIL ■

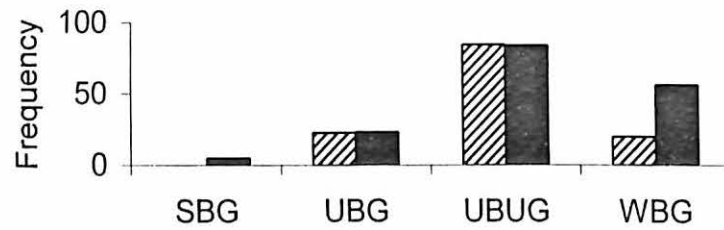# Karwinskia humboltiana
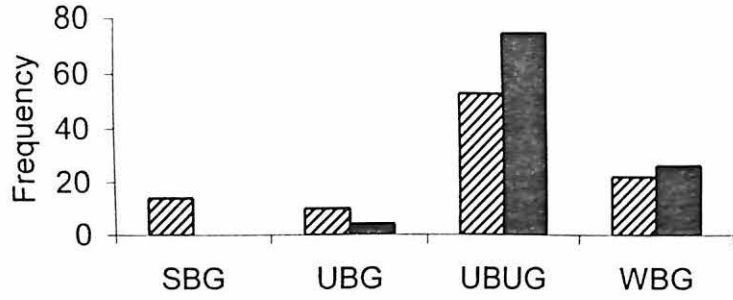
Lantana
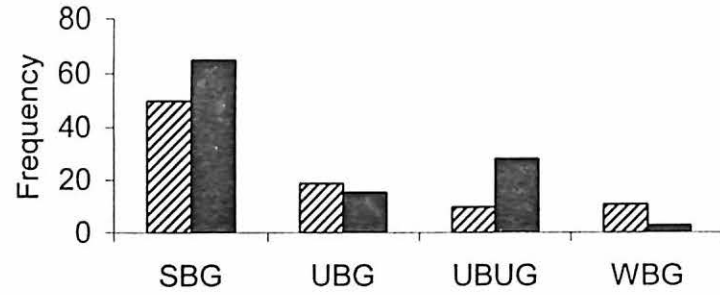
Opuntia engelmannii
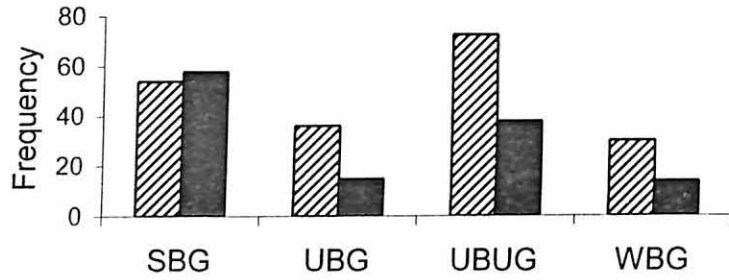
Opuntia leptocaulis
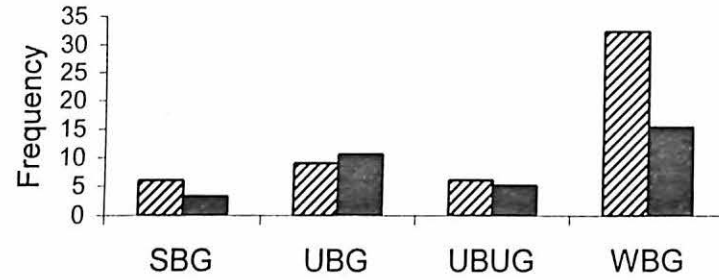
Prosopis glandulosa
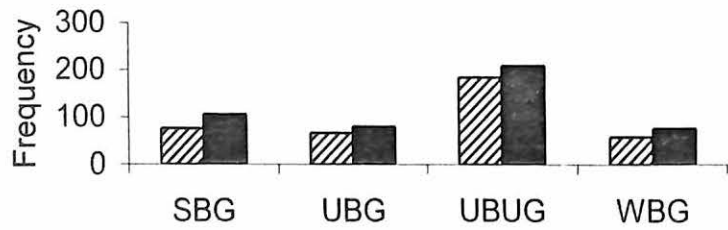
Zanthoxylum fagara

Fig. 6. Richness and diversity of woody vegetation used by treatment on a per lizard

basis at Chaparral Wildlife Management Area, Dimmit and LaSalle Counties, Texas.
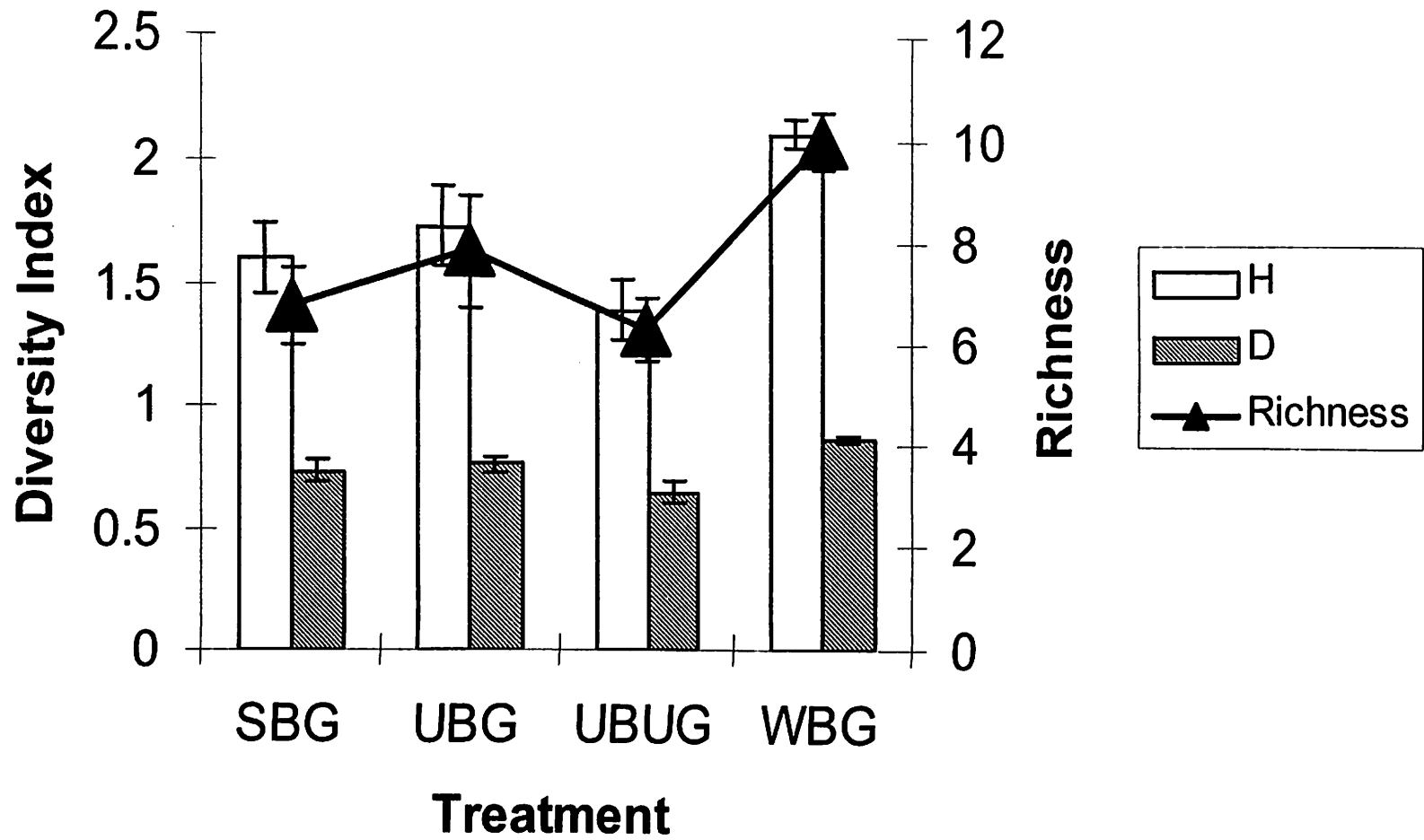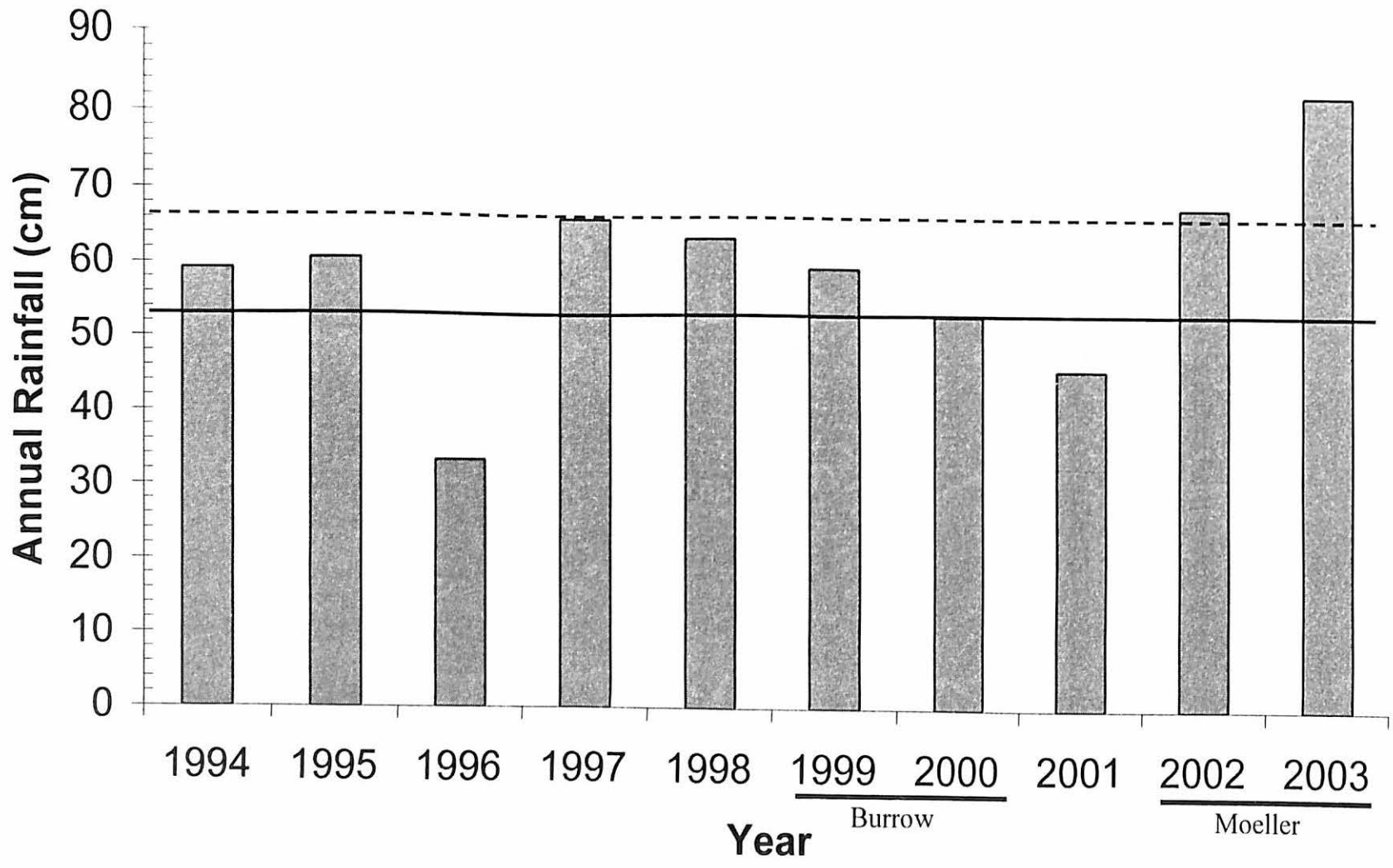
Fig. 7. Annual precipitation (cm) from 1994-2003 at Chaparral Wildlife Management Area (CWMA), Dimmit and LaSalle Counties, Texas. CWMA 20-year average is delineated with a dashed line. County average is delineated with a solid line. The years of study for Burrow (2000) and the present study (Moeller) are marked.

VITA ①

Candidate for the Degree of

Master of Science

Thesis: CONSERVATION ECOLOGY OF THE TEXAS HORNED LIZARD
(*PHRYNOSOMA CORNUTUM*): COMPARATIVE EFFECTS OF SUMMER
AND WINTER BURNING

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volunteer program and conducted Alabama beach mouse monitoring; Rob
and Bessie Welder Wildlife Foundation Fellow 2002 to present at
Oklahoma State University.

Professional Memberships: The Wildlife Society, Society for the Study of
Amphibians and Reptiles, Southwestern Association of Naturalists,
Oklahoma Academy of Sciences.