

## Fire in Chihuahuan Desert grassland: Short-term effects on vegetation, small mammal populations, and faunal pedoturbation

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### ABSTRACT

A prescribed burn resulted in significant decreases in canopy cover of the grasses: *Bouteloua eriopoda*, *Sporobolus flexuosus*, and *Aristida purpurea*. One year post-burn, basal cover of *B. eriopoda* remained significantly lower in burned patches than in unburned areas but there were no differences in basal cover of the other perennial grasses. Only one species of the 14 summer annual species occurred in both burned and unburned plots. There were six species of spring annuals in burned patches but no spring annuals in the unburned grassland ten months post-burn. Fire killed 100% of the snakeweed shrubs (*Gutierrezia sarothrae*), 77% of the *Ephedra torreyana* shrubs, and 36% of the *Yucca elata*. All mesquite shrubs that were top-killed by fire, resprouted one month post-burn. Fire had no effect on abundance and species richness of rodents. There were fewer wolf spider, *Geolycosa* spp. burrows in burned areas than in unburned grassland. The area and volume of soil in termite galleries and sheeting were significantly larger in the unburned grassland than in the burned areas.

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### 1. Introduction

Fire is a frequently used management tool for improving forage production in many arid and semi-arid regions (Bennett et al., 2003; Guevara et al., 1999; Perez et al., 2008; Snyman, 2004). The effects of prescribed burns vary among vegetation types, climate, and topography. In some arid regions, fire is an effective tool for reducing the abundance and biomass of undesirable shrubs that contribute to the woody weed problem in arid and semi-arid regions (Noble, 1997) while in other regions fire has little effect on shrubs (Drewa and Havstad, 2001).

The role of fire in desert grasslands remains controversial (Dick-Peddie, 1993; Drewa and Havstad, 2001; McPherson, 1995). Humphrey (1958) concluded that repeated fires were critical to the maintenance of treeless desert grasslands. More recently, Bahre (1991) stated that not only is fire essential for the maintenance of grasslands but fire size and frequency have been dramatically reduced since the establishment of the livestock industry in the southwestern United States. In a review of the history of desert grasslands in southern New Mexico there were no accounts of fire

in the region (Buffington and Herbel, 1965). In the Chihuahuan Desert grasslands of southern New Mexico, “fire appears to have been of little importance in desert grassland vegetation before settlement” (Dick-Peddie, 1993).

It has been suggested that fire regimes are more likely the result of vegetation patterns rather than serving as the agent responsible for them (Clark, 1990). Vegetation patterns of Chihuahuan Desert grasslands are a mosaic of dense bunch grasses interspersed with much sparser grass cover especially in the vicinity of banner-tailed kangaroo rat (*Dipodomys spectabilis*) mounds. In more than 40 years of noting the extent of lightning strike fires, the size of burned areas ranged from 2 to 5 ha in the grasslands of the Jornada Experimental Range in southern New Mexico (pers. obs. W.G.W.). This suggests that prescribed burns in Chihuahuan Desert grassland will produce patches of burned and unburned grassland that is a function of the mosaic pattern of high grass cover.

If fire is to be used as a management tool, it is important to examine the impacts of fire on important ecosystem properties and processes. Black grama grasslands are thought to be fire sensitive, because grass mortality can be high during dry years and regeneration through stolon and tiller expansion is slow (Cable, 1965; Jameson, 1962). In cases where drought and heavy grazing follow fire, black grama mortality can be extremely high and recovery could last 50 years (Bock and Bock, 1992; Wolters et al., 1996). A prescribed

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burn in Chihuahuan Desert grassland with a history of moderate to heavy grazing by livestock resulted in a burned–unburned mosaic with a marked reduction in live cover of black grama (*Bouteloua eriopoda*) in the burned areas (Drewa and Havstad, 2001). Our studies were conducted in a *B. eriopoda* grassland with high grass cover and sparse cover of the invasive shrub, mesquite (*Prosopis glandulosa*). This grassland had a history of light grazing because of the distance from water. Therefore we predicted a more complete burn than reported by Drewa and Havstad (2001). Several important sub-dominant plants e.g. *Yucca elata*, *Gutierrezia sarothrae*, and *Ephedra torreyana* are components of this grassland which allowed us to test the hypothesis that fire would have an adverse effect on important sub-dominant shrubs as well as on black grama.

Animals are important components of desert ecosystems because of their effects on ecosystem processes and properties (Chew, 1979). The effects of animals on soil properties and processes (pedoturbation) are considered to be the most important roles of animals on arid and semi-arid ecosystems (Whitford, 2000; Whitford and Kay, 1999). Small mammals affect ecosystem properties and processes by burrowing, foraging for insect larvae, and by excavating seed caches. These activities break surface crusts, create voids and depressions that trap seeds and organic material, and mix excavated soil with intact surface soil (Whitford and Kay, 1999). Several groups of desert grassland arthropods contribute to the transport of subsurface soil to the surface (pedogenesis) and to the concentration of nutrients in small soil patches (i.e. termites, ants, and burrowing spiders). We hypothesized that the change in vegetation structure resulting from fire would reduce the abundance of these animals and pedoturbation resulting from their activities. If fire affects small mammal population sizes and species composition, changes in the abundance of foraging pits have the potential of slowing the recovery of the grassland vegetation (Killgore, 2004). Because fire changes the physical structure of desert grasslands, we hypothesized that small mammal populations would be adversely impacted by fire and that small mammal produced pits would be reduced in burned grassland.

## 2. Materials and methods

### 2.1. Study site

We conducted this study on the Armendaris Ranch approximately 37 km east of Truth or Consequences, NM. The grasslands are dominated by black grama grass with scattered honey mesquite (*P. glandulosa*) shrubs. The soils of the study area are sandy-loam aridisols. The 32-ha study site was enclosed by a three strand, solar-powered electric fence to exclude bison (*Bison americana*) that used the large paddock. A 2-m wide road was cut through the grassland by a road grader around the approximately 20 ha area to be burned. The road served as a fire containment barrier. The plots were burned on 7 July 2001, during the season when dry lightning storms that precede the summer rains are prevalent. Despite the dry conditions, approximately 40% of the area did not burn; the sparse grass cover around banner-tailed kangaroo rat (*D. spectabilis*) burrow mounds in the area had insufficient fuel to carry the fire. The post-burn plot was a mosaic of burned and unburned areas that varied in size from a fraction of a hectare to approximately 1.5-ha. We selected burn areas greater than 1 ha for comparison with 1 ha plots located on the opposite side of the fire containment road that were not burned.

### 2.2. Vegetation

We measured basal cover, total cover, and species composition of perennial vegetation by the line intercept method (Canfield, 1941) on eight paired, 20-m lines in the unburned area outside the

containment road and in the burned area opposite the control line. Because the minor grass species did not occur on all transects, statistical analysis of the fire effects on these species was not considered. The effect of fire on the basal cover perennial grasses was examined by Analysis of Variance with Tukey's HSD to compare means.

Density of spring and summer annuals was estimated with a 100 cm × 50 cm quadrat frame divided into 50, 10 cm × 10 cm squares. Annuals were sampled in 15 quadrats in the burned patches and 15 quadrats in the unburned grassland. Spring annuals were sampled in May, one month prior to the burn, and in late March, nine months after the burn. Summer annuals were sampled four months after the prescribed burn.

We measured the pre- and post-burn canopy volume of honey mesquite by measuring the height and two canopy diameters of 30 shrubs on or adjacent to three, random, 100-m line transects. To better understand change in vegetative structure and recovery of Torrey Ephedra (*E. torreyana*) and soap tree yucca (*Y. elata*) to fire, we also established eight, 50-m × 3-m, random transects one year after the burn. We recorded number of live and dead target species that fell within each transect. We measured canopy diameter and height of new *Y. elata* resprouts, and estimated percent canopy pruned by black-tailed jackrabbit (*Lepus californicus*).

### 2.3. Small mammals

Pre-burn and post-burn small mammal surveys were conducted over three consecutive nights (52 weeks prior to the burn; 2 and 13 weeks post-burn) to estimate the relative abundance of small mammal species both on the burned plot and on adjacent control plots. Trapping was conducted with Sherman™ live traps baited with rolled oats and peanut butter. We established 4 pre-burn trap grids of 5 rows with 7 traps per row at 15 m spacing, one grid in each cardinal direction. Two post-burn grids were established in the middle of the largest (1.5 ha) burned patches and two were located in the unburned grassland approximately 100 m from the perimeter road. Each rodent captured on the first two nights of a census period was marked with a color code on the pelage using colored permanent dye Sharpie™ felt-tipped pens. Relative abundance was recorded as minimum number known to be alive on each plot.

A pre-burn and post-burn census of active and abandoned banner-tailed kangaroo rat mounds was conducted. The location of each mound and perimeter of burned plot were mapped using a Trimble Geo-explorer II™ Global Positioning System and processed in ArcView 3.2. We determined mound occupancy by banner-tailed kangaroo rats by the presence of indicators of active use: plant fragments kicked out of mound entrances, fresh soil around mound entrances, foot prints and tail drag marks, and fresh foraging trails radiating from mounds. These activity signs provide population-size estimates equal to those based on saturation trapping (Cross and Wasser, 2000).

### 2.4. Faunal pedoturbation

Faunal pedoturbations were measured on 3, 10 m radius circular plots located on the opposite side of the fire containment road (control plots) and 3, 10 m radius circular plots located in the largest burned patches. Soil disturbances were measured nine months prior to the prescribed burn for comparison with measurements one month and four months post-burn. Two diameters and depth were recorded for each rodent cache pits and larger pits produced by animals digging for grubs. The area of termite gallery-sheeting was estimated by average length and width. Two diameters were recorded for ant nest structures.

Burrows of ground-dwelling wolf spiders (*Geolycosa* spp.) were recorded. The effects of prescribed burn on these pedoturbations were examined by Analysis of Variance with Tukey's HSD to compare means.

### 2.5. Soil stability

Short-term losses in soil stability could follow disturbances such as fire. We collected soil aggregate stability measurements (Herrick et al., 2001) in burned and control plots (surface and subsurface) one year following the burn. Soil samples are rated on a scale based on a combination of ocular observations of slaking during the first 5 min following immersion in distilled water and the percent remaining on a 1.5-mm sieve after five dipping cycles at the end of the 5-min period.

## 3. Results

### 3.1. Vegetation

One month after the prescribed burn, the basal areas of the four main grasses (*B. eriopoda*, *A. purpurea*, *S. flexuosus*, and *Pleuraphis (Hilaria) mutica*) were significantly lower than pre-burn estimates (Fig. 1). One year following the burn, black grama basal cover remained significantly lower than pre-burn cover ( $F_{5,42} = 379$ ,  $p < 0001$ ); all other grasses exhibited significant recovery and there were no significant differences in basal cover of the other grasses in burned and unburned plots.

The fire also significantly reduced the cover one of the minor grass species, purple three awn, *Aristida purpurea*, and killed most of the small shrubs: *G. sarothrae*, *E. torreyana*. Three months after the burn, there were new stems and leaves of the honey mesquite around the base of each of the mesquite shrubs and new leaf crowns of soaptree yucca emerging from the base of some of the burned plants. Of 23 burned soaptree yucca plants measured, we observed 35 resprouts. Ten of the burned *Y. elata*, produced two resprouts and one produced three resprouts. Forty-three percent of the resprouts were pruned to some extent. Based on observations of fecal-pellet deposition and aggregation near pruned leaves, all

browsing of soaptree yucca resprouts was attributed to black-tailed jackrabbits (*L. californicus*). Rabbit browsing removed  $34.1 \pm 8.1\%$  of the leaf crown area of the browsed *Y. elata* in the burned areas. *L. californicus* browsed 24% of the yucca in the unburned grassland and  $26.2 \pm 8.1\%$  of the leaf crown area of the browsed plants. In the burned plots, the smaller resprouts were intensively browsed and a much larger percent of canopy was browsed. In the unburned area, only soaptree yucca plants with  $<0.05 \text{ m}^3$  volume were browsed, with higher percentage browsed associated with smaller plants. Of seventy-one *E. torreyana* encountered on transects in the burned patches, seventy-seven percent were killed by the fire.

Fire had a marked effect on the species composition of the annual plant communities that developed during the first monsoon season post-burn and nine months post-burn. Only one species of summer annual (*Hoffmanseggia glauca*) occurred in burned grassland patches and in the unburned grassland (Table 1). There were ten species of summer annuals recorded in the burned patches and five species recorded in the unburned grassland. One species (*Eriogonum abertianum*) that was relatively abundant in the pre-burn vegetation was eliminated by the fire but remained as the most abundant annual in the unburned grassland 6 weeks after the fire. There were differences in species composition in the pre-burn spring annual community and the post-burn annual community. Nine months after the prescribed burn there were six species of winter-spring annuals that established in the burned areas but no spring annuals were recorded in the unburned grassland.

Although the prescribed burn resulted in top kill of all honey mesquite shrubs in the burned areas, rapid re-growth from the root crowns and green shoots were recorded from every burned mesquite plant by October. By the end of the second post-burn growing season, *P. glandulosa* shrubs recovered to between 10% and 100% of the pre-burn canopy size.

### 3.2. Small mammals

The only rodent species that was present on all plots at all sampling periods was Ord's kangaroo rat, *Dipodomys ordii*. There were no differences in relative abundance of this species in burned

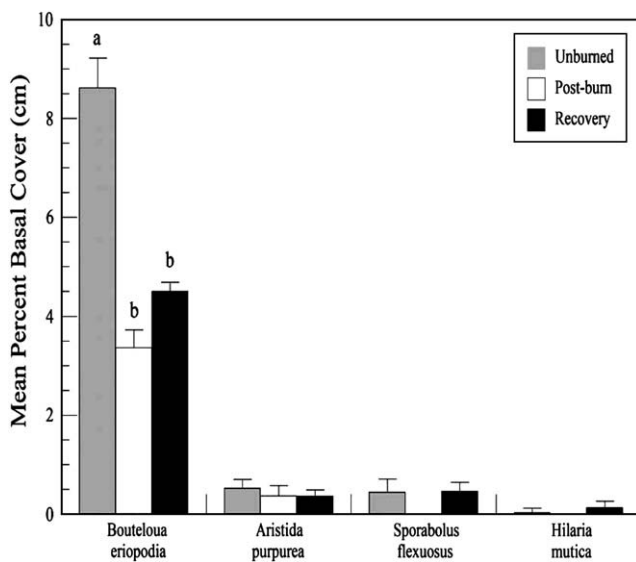


Fig. 1. Changes in basal cover of dominant grasses (Boer – *Bouteloua eriopoda*, Arpu – *Aristida purpurea*, Spfl – *Sporobolus flexuosus*, Himu – *Hilaria mutica*) four months after the burn (post-burn) and at the end of the growing season one year after the burn (recovery).

Table 1

Average number  $\text{m}^{-2}$  ( $\pm$  standard error) of summer and spring annuals on burned and unburned plots. There were no spring annuals on the unburned plots ten months post-burn.

Species	Summer		Spring	
	Burned	Unburned	Pre-burn	Burned
<i>Aphanostephus ramosissimus</i>	0	$0.4 \pm 0.1$	0	0
<i>Aristida adscensionis</i>	$0.6 \pm 0.3$	0	0	0
<i>Astragalus nuttalianus</i>	0	0	$2.6 \pm 1.7$	0
<i>Bouteloua aristoides</i>	$0.6 \pm 0.3$	0	0	0
<i>Cryptantha angustifolia</i>	0	0	$7.9 \pm 3.1$	$1.3 \pm 0.5$
<i>Cryptantha crassisepala</i>	0	0	$4.3 \pm 1.3$	0
<i>Descurainia pinnata</i>	0	0	$10.4 \pm 2.7$	0
<i>Dithyrea wislizenii</i>	0	0	$0.1 \pm 0.1$	$0.3 \pm 0.2$
<i>Eriogonum abertianum</i>	0	$9.2 \pm 1.7$	$8.4 \pm 2.7$	$0.7 \pm 0.3$
<i>Eriogonum rotundifolium</i>	$0.6 \pm 0.3$	0	0	0
<i>Euphorbia serrula</i>	$0.6 \pm 0.3$	0	0	0
<i>Gutierrezia sphaerocephalum</i>	0	$0.6 \pm 0.1$	0	0
<i>Hoffmanseggia glauca</i>	$0.6 \pm 0.3$	$1.0 \pm 0.5$	0	0
<i>Hymenopappus flavescens</i>	$1.1 \pm 0.3$	0	0	0
<i>Lepidium lasiocarpum</i>	0	0	$2.9 \pm 0.8$	$0.3 \pm 0.2$
<i>Lesquerella Gordonii</i>	0	0	0	$7.5 \pm 2.0$
<i>Machaeranthera linearis</i>	0	$4.1 \pm 0.6$	0	0
<i>Malacothrix Fendleri</i>	0	0	0	$1.3 \pm 0.5$
<i>Panicum hirticale</i>	$1.8 \pm 0.3$	0	0	0
<i>Portulaca oleracea</i>	$1.4 \pm 0.5$	0	0	0
<i>Tidestromia lanuginosa</i>	$0.4 \pm 0.2$	0	0	0
<i>Verbesina encelioides</i>	$3.2 \pm 1.0$	0	0	0

**Table 2**  
The relative abundance of rodents expressed as number captured per 100 trap nights  $\pm$  standard error on burned (burn) and unburned (control) plots following the fire in June 2001.

Species	One month				Four months	
	Pre-burn	Control	Burn	Control	Burn	Control
<i>Dipodomys ordii</i>	1.3 $\pm$ 0.4	0.4 $\pm$ 0.4	2.7 $\pm$ 0.9	0.9 $\pm$ 0.0	4.4 $\pm$ 1.4	2.2 $\pm$ 1.3
<i>Perognathus flavus</i>	0.0	0.0	2.2 $\pm$ 0.6	0.0	3.6 $\pm$ 1.3	1.8 $\pm$ 0.6
<i>Neotoma micropus</i>	0.0	0.0	0.0	0.0	0.4 $\pm$ 0.4	0.0
<i>Dipodomys spectabilis</i>	0.0	0.0	0.0	0.4 $\pm$ 0.4	0.0	0.4 $\pm$ 0.4
<i>Spermophilus spilosoma</i>	0.0	0.0	0.0	0.0	0.4 $\pm$ 0.4	0.4 $\pm$ 0.4
<i>Onychomys torridus</i>	1.8 $\pm$ 0.9	0.0	0.0	0.0	0.0	0.4 $\pm$ 0.4

areas and in the unburned grassland ( $F_{5,11} = 1.83$ ,  $p > 0.2$ ) (Table 2). The silky pocket mouse, *Perognathus flavus*, was captured only in the burned plots one month post-burn and more were captured in the burned areas than in unburned grassland four months post-burn. The low abundance of other rodent species provided data on the species composition of the grassland rodent community but not on the effects of a prescribed burn.

Fifty-one banner-tailed kangaroo rat mounds were mapped on the area to be burned. Prior to the burn, there were 6 occupied mounds on the area scheduled for a prescribed burn. In the burned area, only two of the mounds exhibited signs of use by *D. spectabilis* one month after the burn. The mounds that were occupied by banner-tailed kangaroo rats were in the middle of relatively large patches of unburned grass.

### 3.3. Faunal pedoturbation

Prescribed fire resulted in significantly fewer and smaller termite gallery-sheeting in the burned patches than in the unburned grassland four months post-burn ( $F_{3,8} = 3.86$ ,  $p < 0.05$ ) (Table 3). There were more rodent cache pits in the burned areas both one month post-burn ( $F_{5,12} = 28.3$ ,  $p < 0.001$ ). However there were no differences in the average area of the pits excavated by rodents ( $p = 0.07$ ). There were no differences in the number of nests or the area of soil in the mounds of ants that construct seasonal nest mounds in burned or unburned patches. The ant species that produce nest mounds or discs that persist for several years (seed harvester ants, *Pogonomyrmex rugosus* and generalists, *Aphaenogaster cockerelli*) were active in both burned and unburned grassland one month and four months post-burn. There were fewer wolf spider (*Geolycosa* spp.) burrows in the burned patches than in the unburned grassland four months post-burn ( $F_{3,8} = 4.7$ ,  $p < 0.03$ ) (Table 3).

### 3.4. Soil stability

Surface and subsurface aggregate stability were not significantly different between treatments (paired  $t$ -test,  $p = 0.46$ ,  $n = 11$ ; and Mann-Whitney rank sum test,  $p = 0.390$ ,  $n = 11$ ).

## 4. Discussion

This study adds to the body of evidence that fire has an adverse effect on black grama (*B. eriopoda*) grasslands by reducing the basal area of the surviving tussocks of the dominant species. The effects of the experimental fire on the dominant grasses were similar to those reported for cool fire in black grama grassland (Cable, 1965; Reynolds and Bohning, 1956; Valone et al., 2002). Protection from grazing and average summer monsoon rains allowed the black grama to recover to approximately 50% of the pre-burn basal cover by the end of the first post-burn growing season. The small changes in basal cover one year post-burn are consistent with the finding that recovery of black grama is slow when plants are subjected to

drought and grazing after fire (Drewa and Havstad, 2001; McPherson, 1995). Our prescribed burn preceded three months of rainfall that totaled approximately 130 mm, greater than 50% of the average annual total rainfall for this area. The recovery of *S. flexuosus*, *A. purpurea*, and *P. mutica* within one year post-burn demonstrates that these species are more resilient to fire than *B. eriopoda*. Fire has been shown to reduce the growth of three grass species during the first month after a burn but at the end of the growth cycle there were differences in the degree of recovery among the species (Pelaez et al., 2003). Species specific responses to fire may result in changes in species composition of the plant community following fire and must be considered in evaluation of the effectiveness of fire as a management tool.

While fire is known to be important for the productivity of spinifex grasslands (*Triodia* spp. and *Plectrachne* spp.) in Australia (Wright and Clarke, 2007), semi-arid, subtropical Australian grasslands dominated by *Themeda triandra* produced lower cover and biomass than unburned areas for 2.5 years post-burn. This reduced productivity occurred despite above average rainfall during the first season of recovery. The response of *T. triandra*

**Table 3**

Short-term effects of fire in a Chihuahuan Desert grassland on the area and abundance ( $\pm$  standard error) of soil disturbance by animals. Numbers followed by different letters are significantly different (Tukey's significant difference test,  $p < 0.05$ ).

Termite sheeting	Unburned area (cm <sup>2</sup> )	Unburned number	Burned area (cm <sup>2</sup> )	Burned number
Nine months pre-burn	2862 $\pm$ 394	31 $\pm$ 3.3a	3126 $\pm$ 325	17.7 $\pm$ 2.4b
One month post-burn	0	0	0	0
Four months post-burn	318,475 $\pm$ 29,425a	17,137 $\pm$ 1162a	64,975 $\pm$ 19,300b	2500 $\pm$ 550b
<i>Rodent cache pits</i>				
Nine months pre-burn	2649 $\pm$ 254	20.0 $\pm$ 2.0	2333 $\pm$ 434	15.3 $\pm$ 1.4
One month post-burn	812 $\pm$ 118	9 $\pm$ 1.7a	121 $\pm$ 156	1.7 $\pm$ 1.6b
Four months post-burn	4928 $\pm$ 857	21.7 $\pm$ 0.9	3205 $\pm$ 276	19.7 $\pm$ 0.9
<i>Seasonal ants</i>				
Nine months pre-burn	524 $\pm$ 176	3.6 $\pm$ 1.1	248 $\pm$ 183	1.2 $\pm$ 0.5
One month post-burn	0	0	0	0
Four months post-burn	800 $\pm$ 533	2.3 $\pm$ 1.3	1529 $\pm$ 1019	2.3 $\pm$ 1.3
<i>Geolycosa spp. holes</i>				
Nine months pre-burn		33.3 $\pm$ 9.5		18.3 $\pm$ 0.5
One month post-burn		0		0
Four months post-burn		26.3 $\pm$ 6.8a		4.3 $\pm$ 0.5b



grasslands to fire was similar to that of *B. eriopoda* grasslands in the Chihuahuan Desert. These studies provide additional evidence that generalizations about the responses of grasslands in arid and semi-arid regions are inappropriate. The responses of arid region grasslands to fire must be evaluated on the basis of responses of the dominant grass species.

Mortality of honey mesquite in response to fire has been reported as low (Box, 1967). Areas with higher grass cover and fuel loads have resulted in large reductions in canopy (up to 95%). However maximum fire-induced mortality of mesquite reported in earlier research remained around 10% (Ansley and Lucia, 1994). In black grama grasslands, due to the low fuel loads and cooler fire temperatures, there was no honey mesquite mortality, and top-killed canopy re-growth was a function of pre-burn canopy volume. The absence of mortality of honey mesquite in our study is probably due to the low fire intensity and season of the burn. Drewa (2003) reported higher percent resprouts of mesquite after dormant season burns than during growing season burns. In our study the relatively low fuel load resulted in patchy burns and insufficient heat penetration into the soils to adversely affect a deep rooted shrub like honey mesquite.

The fraction of *Y. elata* that produced new leaf crowns from the below-ground meristems on the rhizomes of the burned plants was higher than reported in the literature (Humphrey, 1958; Thomas and Goodson, 1992). The high proportion of two or more leaf crowns emerging from the base of a single charred yucca caudex may represent the effect of the monsoon rains following the fire. The high proportion of *Y. elata* leaf crowns in the burned patches that were heavily browsed by rabbits is probably because these leaf crowns were exposed to bare soil around the charred yucca stumps. In the unburned grassland, many of the small *Y. elata* leaf crowns were obscured by large grass tussocks. This level of herbivory on *Y. elata* re-growth leaf crowns appears not to have long term consequences for this plant. The leaves that were browsed by jackrabbits were on the outside of the leaf crown. *Y. elata* can recover from loss of more than 90% of the leaves by pack-rat (*Neotoma* spp.) herbivory and exhibits compensatory growth when the central cluster of leaves is removed by cattle (Kerley et al., 1992).

Of the vegetation censused in this study, prescribed fire had the largest negative impact on Torrey Ephedra. *E. torreyana* is an important winter browse for resident pronghorn antelope (*Antilocapra americana*), black-tailed jackrabbits (*L. californicus*) and desert cottontails (*Sylvilagus audubonii*). Other species of *Ephedra* have been reported to sprout vigorously following fire in the Great Basin desert (West and Hassan, 1985) but only 9% of the top-killed *E. torreyana* resprouted after a June wildfire in the Sonoran Desert of Arizona (McLaughlin and Bowers, 1982) a response similar to that recorded in this study. Another small shrub that was adversely impacted by fire was snakeweed, *G. sarothrae*. All of the *G. sarothrae* shrubs in our study were killed by the fire. However, *G. sarothrae* seedling establishment was reported to be higher in areas where burning left most of land barren (McDaniel et al., 2000) which cancels the potential of prescribed burning as a tool for reducing the abundance of this undesirable shrub. In Arizona grasslands a small shrub, *Haplopappus tenuisectus*, suffered nearly total fire-induced mortality and had not recovered 8 years following the burn (Bock and Bock, 1997). The differences in responses of shrubs to fire are probably a function of the physiology and life history characteristics of the shrubs.

The difference in species composition of the summer annuals on recently burned patches and in unburned grassland suggests that the fire affected soil moisture distribution and nutrient availability despite the lack of difference in soil aggregate stability. Soil moisture and nitrogen availability may be affected by the loss of grass canopy that intercepts rainfall and by a pulse of organic carbon in

the form of dead grass roots that affects the nitrogen mineralization-immobilization ratio (Tongway and Whitford, 2002). Soil moisture and available nitrogen have been shown to affect the species composition and community dominance of summer annuals in creosotebush, *Larrea tridentata*, shrubland (Gutierrez and Whitford, 1987a,b). The timing of the prescribed burn prior to the onset of the summer monsoon rains is also a factor in the responses of summer and spring annuals to the burn. The season during which a fire occurs in arid grassland has been reported to result in differences in herbaceous plants (McPherson, 1995).

Despite the lower density of annuals on recently burned patches, these annuals were larger than the annuals in unburned grassland suggesting that fire is beneficial for herbaceous annuals in Chihuahuan Desert grasslands. High productivity of herbaceous annuals were reported in recently burned spinifex grasslands (Wright and Clarke, 2007). Fire may affect soil nutrients and soil moisture which may lead to higher biomass and diversity of herbaceous annuals in recently burned areas.

This study supports the contention that fire regimes in some arid regions are the result of vegetation patterns rather than serving as the agent producing the patterns (Clark, 1990). The burned-unburned mosaic in this study occurred in a good condition black grama grassland that was judged to have a sufficient fuel load to carry fire. A burned-unburned mosaic resulted from a prescribed burn in another black grama grassland in the Chihuahuan Desert (Drewa and Havstad, 2001). The resulting mosaic in this was mostly attributable to the activities of banner-tailed kangaroo rats (*D. spectabilis*) that remove tillers from grasses and store the grass fragments in their mounds. Activities of *D. spectabilis* produce relatively bare areas around active and recently abandoned mounds (Cross and Wasser, 2000; Kay and Whitford, 1978).

The mosaic of burned and unburned patches contributed to the lack of a fire effect on the rodent community and consequently on the excavations produced by small mammals. In a similar Chihuahuan Desert grassland community, Fitzgerald et al. (2001) and Valone et al. (2002) reported no significant differences in abundance and species richness of rodents on fire treated plots and unburned plots. In this study, the only heteromyid rodent captured more frequently in burned patches than in the unburned grassland was the silky pocket mouse, *P. flavus*. *P. flavus* has been reported to prefer low grass cover habitats (Jones et al., 2003) which may explain the absence of this species from the trappable population in the unburned grassland one month post-burn. The high numbers of captures of *P. flavus* (average weight < 10 g) on the burned patches provide an explanation for the differences in abundance of cache pits with no difference in the area of soil excavated in the burned and unburned plots. Most of the cache pits in the burned patches were smaller than those in the unburned patches suggesting that those pits were dug by the smaller *P. flavus* while the cache pits in the unburned grassland were probably produced by the larger *D. ordii*.

The difference termite gallery-sheeting production after the burn can be explained by the loss in grass cover. Termite galleries are formed primarily around the base of old or dead grass stems and on woody litter (MacKay et al., 1985). Gallery and sheeting is formed by termites on dead plant material and was found predominantly on the blades and shoots of black grama. Canopy cover of black grama was 27% lower following the burn and there was an absence of grass litter in the burned patches which resulted in the decrease in termite gallery and sheeting in the burned plots.

Greater abundance of burrowing spiders in unburned grassland than in burned patches four months post-burn is probably the result of changed habitat structure affecting the food resources of the spiders (Conley, 1985). *Geolycosa* spp. are sit and wait predators on small arthropods. The absence of *Geolycosa* spp. in both burned and unburned grassland one month after the prescribed burn

suggests that the surviving adult spiders did not open their burrows until the onset of monsoon rains in late July. The burrow entrances recorded four months after the burn were small, <1 cm diameter, which is characteristic of juvenile spiders. The reduced grass cover in the burned patches probably reduced prey availability for spiders establishing burrows.

The absence of differences in abundance and area of nest mounds of seasonal ants four months after the burn is attributable to the production of *Y. elata* leaf crowns from the subterranean tubers and the production of *P. glandulosa* shoots from the root crowns of the top-killed shrubs and recovery of perennial grasses. The seasonal ants, *Forelius pruinosus* and *Dorymyrmex insana* collect honey-dew from homopterans on grasses, *Y. elata*, and *P. glandulosa* (Forbes et al., 2005). The vegetation production in the four months after the burn appears to have been sufficient to support sufficient homopteran populations for these ants in the burned patches.

This study adds to the evidence that prescribed fire is not a useful management tool for reducing undesirable shrubs in black grama grassland. The species specific responses of grasses, shrubs, and annual plants to a prescribed burn indicate that generalizations about the use and effects of fire are not applicable to other arid and semi-arid ecosystems. The effects of prescribed fire in *B. eriopoda* grassland on rodents, ants, wolf spiders and termites result from vegetation changes that affect food resources of the taxa or the physical structure of the habitat. Short-term changes in the abundance, spatial distribution, and activities of these animals affect the quantity and spatial distribution of soil water and nutrients (Whitford, 2000; Whitford and Kay, 1999). These short-term effects can change the rate and trajectory of recovery of the grassland ecosystem after a fire.

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