POTENTIAL IMPACTS OF RANGELAND MANIPULATIONS ON DESERT RODENT COMMUNITIES

by

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ABSTRACT

I compared vegetation features and rodent communities between manipulated and non-manipulated ranges in southeastern Arizona during summers 1996 and 1997. I also examined the effect of burying traps to determine if this procedure altered trap sensitivity. I used two-way analysis of variance or paired t-tests for all comparisons and identified relationships between rodent communities and vegetation features with linear regression. There was no difference in rodent species richness or population size between mechanically treated and control areas ($P \geq 0.10$). Two kangaroo rat (Dipodomys) species exhibited contrasting relationships with increasing grass cover. Rodent species richness and population size were greater on ungrazed compared to grazed areas ($P < 0.10$). Prescribed fire did not have an obvious impact on rodent species richness or population size, though kangaroo rats may have increased following the burn. Buried traps may have demonstrated a reduction in sensitivity because I caught fewer animals in those traps compared to non-buried traps ($P = 0.087$).

INTRODUCTION

The following chapters constitute partial fulfillment of the requirements for the degree of Master of Science in Wildlife and Fisheries Science in the Graduate College at The University of Arizona. The chapters consist of 4 manuscripts that are intended for submission to peer-reviewed journals. Chapter 1 is intended for submission to the Journal of Wildlife Management. Chapters 2 and 3 are intended for submission to Southwestern Naturalist. Chapter 4 was prepared for submission to the Journal of Mammalogy. The chapters represent my ideas, analyses, and writing abilities. I designed the studies, analyzed all the data, and prepared the manuscripts. Each chapter has two co-authors. For co-authorship I used the guidelines provided by Dickson and Conner (1978. Guidelines for authorship of scientific articles. Wildlife Society Bulletin 6:260-261). Authorship for each chapter is as follows: C. S. Fitzgerald, P. R. Krausman, and M. L. Morrison.

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RH: Rodents and Brush Control · Fitzgerald et al.

RODENT ABUNDANCE AND SPECIES RICHNESS ON BRUSH-CONTROLLED DESERT GRASSLANDS

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Abstract: The semi-desert grasslands of the southwestern United States have been increasingly invaded by woody plants such as mesquite (*Prosopis* spp.) during the last century due to fire suppression, overgrazing, climatic changes, herbivory by leporids, and seed dispersal by rodents. Controlling woody species and maintaining grasslands requires responsible grazing practices and prescribed fire. However, for fire to be effective, a sufficient fuel load must be present to carry a burn. Because of the depauperate fuel load throughout much of the region, efforts have been made to re-establish grasslands via mechanical techniques coupled with re-seeding of grasses. We assessed the potential impacts of mechanical shrub control on rodent communities in shrub-invaded desert grasslands. We used a paired design to compare areas where brush control treatments were previously applied with adjacent nontreated reference areas. We compared vegetation cover, rodent species richness, population estimates, and species abundance indices between treatment and reference areas. We also identified relationships between rodent community characteristics and vegetation cover. Rodent species richness and abundance did not differ between brush-treated and reference areas ($P > 0.10$). Overall rodent population size ($\hat{N}$) was negatively related to increasing bare ground ($r^2 = 0.173$, $P = 0.04$). Merriam's kangaroo rat (*Dipodomys merriami*) was negatively related to increasing grass cover ($r^2 = 0.322$, $P < 0.01$). Ord's kangaroo (*D. ordii*) rat was related negatively to increasing bare ground ($r^2 = 0.178$, $P = 0.04$) and positively to increasing grass cover ($r^2 = 0.312$, $P \leq 0.01$).

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Key words: Arizona, brush management, desert grassland, live-trapping, mark-recapture, mesquite, New Mexico, rodents, small mammals.

Desert rodents are the primary consumers and dispersers of annual and perennial plant seeds and may have a pronounced impact on plant communities within arid rangelands (Fagerstone and Ramey 1996). Heteromyids can affect the abundance of plant species whose seeds are their preferred foods (Fagerstone and Ramey 1996). However, desert rodent communities are, in turn, influenced by vegetation cover, density, and seed production (Rosenzweig and Winakur 1969, Brown et al. 1972, Reichman 1975, Whitford et al. 1978). Range management practices such as prescribed burning, herbicide application, and mechanical brush control alter the vegetation composition and structure (Scifres 1980) and may alter the suitability of habitat for some small mammal species by reducing cover and food availability (Kaufman and Fleharty 1974, Vallentine 1989). Several studies have examined the impacts of prescribed burning on desert rodent communities (Bock et al. 1976, Christian 1977, Bock and Bock 1978, Simons 1991). Less attention has been given to the effects of mechanical brush control on desert rodents.

During the last century, semi-desert grasslands of the southwestern United States have increased in woody vegetation such as mesquite (*Prosopis* spp.) because of livestock overgrazing, fire suppression, climatic change, herbivory by jackrabbits (Lepus spp.), and seed dispersal by rodents (Humphrey 1958; Wright 1972, 1974; Wright and Bailey 1982). Mesquite and other woody shrubs continue to persist, dominate, and increase in abundance (Gibbens et al. 1992, Holechek et al. 1994) to the detriment of native grasses, grassland ecosystems, and rangeland quality (Reynolds and Bohning 1956, Wright 1973). Without brush-control efforts, a reduction in forage for livestock and loss of habitat for grassland-dependent species, such as pronghorn (Antilocapra americana) (Gibbens et al. 1992), and less conspicuous species such as northern pygmy mice (Baiomys taylori) and cotton rats (Sigmodon spp.), may result. Ranchers and government agencies are attempting to restore grassland conditions via improved grazing systems, prescribed burning, and chemical and mechanical brush control (Martin 1975). Their primary goal is to re-establish native grasses and reintroduce fire to the system on a regular schedule once the fuel load of rangelands is sufficient to carry a burn.

There are several different methods of mechanical brush control used on arid rangelands: bulldozing, rootplowing, grubbing, chaining, roller chopping, mowing, railing, and disk plowing (Jordan 1981). We focused on rootplowing (Payne and Bryant 1994:286), grubbing (Jordan 1981), and roller chopping (Vallentine 1983), the practices most commonly used throughout the region for reduction of mesquite and other woody vegetation.

Our objectives were to compare vegetation features and rodent communities between brush-controlled and reference plots, to identify any differences, and to examine potential relationships between rodent communities and vegetation features. We predicted that species richness and abundance of rodents would differ between previously treated and nontreated ranges, and that those differences might be related to treatment-induced differences in vegetation.

STUDY AREA

We conducted the study at 7 sites throughout Cochise County, Arizona in 1996-1997. Mitchell (1976) described the climate as controlled by the summer monsoons occurring during July-September. Most precipitation occurs during summer with occasional rains in winter (Bourgeron et al. 1995). Rainfall was typical of the mean annual precipitation (Smith and Schmutz 1975; Brown 1982) and ranged from 22 - 38 cm among the study sites (NOAA 1996; Wendy Glenn, Malpai Borderlands Group, pers. commun.; Kevin Cobble, U.S. Fish & Wildlife Service, San Bernardino National Wildlife Refuge, pers. commun.). Annual mean temperatures were moderate, averaging 12-20 ° C, but summers were hot (Brown 1982, Bourgeron et al. 1995). Elevations ranged from 1,100 - 1,440 m. Dominant vegetation included grama grasses (Bouteloua spp.), tobosa grass (Hilaria mutica), three awns (Aristida spp.), velvet mesquite (Prosopis velutina), whitethorn acacia (Acacia constricta), creosotebush (Larrea tridentata), tarbush (Flourensia cernua), snakeweed (Gutierrezia sarothrae), and burroweed (Isocoma tenusectus).

METHODS AND ANALYSES

We used a paired treatment experimental design to compare previously manipulated (i.e., 1-6-year-old brush treatments) and reference areas. Each site consisted of an area that had been mechanically manipulated (hereafter treated) in some form and a nearby reference area that had not been manipulated. We established 1-3 90 x 90 m sampling grids within each treated and each control area. These grids were not established randomly because we were sampling areas that had been previously treated and needed to pair treated grids with grids in adjacent reference areas. Therefore, we established the first grid at each site approximately 35-50 m within the treatment edge and set each successive grid systematically at ≥ 500 m intervals (Santillo et al. 1989) along a line within treatment boundaries. The number of grids (1-3) set within a given treatment depended on the size of the treatment area. We located reference grids ≥ 200 m perpendicular to corresponding treatment grids to minimize recapture of animals moving between grids (Sullivan and Sullivan 1984, Hall and Willig 1994) and to maximize similarity between treatment and reference areas.

Within-site samples were not true replicates because they were drawn from the same treatment area (Hurlbert 1984). However, due to the inherent variation within treated areas and the independence of rodent populations among grids, it seemed biologically meaningful to examine these sub-samples as replicates. It is more important to have multiple treatment-control pairs than to have true replicates (Burnham et al. 1987:242). According to Hurlbert (1984), multiple plots within a single treatment are not identical. Therefore, by sub-sampling the treatment area, we achieved a better representation of the potential treatment impacts over a wider range of habitat conditions. This sampling design is referred to as clumped segregation by Hurlbert (1984) because it lacks treatment interspersion. Because we sampled an area that had already been treated, it was impossible to achieve treatment interspersion. However, we selected each pair of grids based on inherent (assumed pre-treatment) similarities between treatment and reference locations, such that the (pre-treatment) variability within a pair of grids was less than that among all grids. The problem of confounding (or pseudoreplication) due to comparisons of abundance between a disturbed and a reference location should be overcome by having several "replicated" disturbed and several reference locations (Underwood, 1994). Samples were replicated at other previously treated sites to increase the scope of inference for this study. We established 24 grids (12 pairs) within the 7 sites. All study sites had been treated within the last 6 years and included 3 rootplowed, 1 grubbed (tree dozed), 1 rootplowed and grubbed, and 2 roller chopped ranges and their respective reference areas (Table 1).

We set 1, 100 m line transect diagonally from a randomly selected corner of each trapping grid and measured percent cover of vegetation along the transects using the point-intercept method. We recorded the type of contacts made along a vertical line at 100 points (i.e., every 1.0 m) along each transect (Bock and Bock 1978). When vegetation was intercepted, we recorded the species. Dead but standing trees and shrubs were recorded separately from living woody plants. We categorized mesquite and acacia as trees; all other woody species were recorded as shrubs or suffrutescents (i.e., half shrubs). When canopies of ≥ 2 plants directly overlapped at a given point, we recorded multiple contacts. Non-vegetation contacts were recorded as pebble (< 5 cm) or cobble (≥ 5 cm), or as bare ground if no vegetative cover, litter, pebbles, nor cobbles were intercepted.

We sampled rodents with folding Sherman live traps (7.5 x 8.75 x 22.5 cm) in a 7 x 7 trap configuration ($\underline{n}$ = 49 traps) with 15 m between consecutive trap stations. At each trap station, we placed one trap and marked the station with a pin flag. As a precaution against trap mortalities, we partially buried each trap to provide some insulation from overnight cold and morning heat.

We trapped rodents on each grid for 3 consecutive nights. At each site, all treated and corresponding reference grids were trapped simultaneously. We set traps baited with mixed birdseed ≤ 3 hours before sunset and checked traps the following morning ≤ 3 hours after sunrise. For all captured individuals, we recorded the species, age, sex, reproductive condition, mass, and trap station. We marked all captures during a 3-night session using permanent ink and released all animals at capture locations. Permanent ink markers have been used for temporarily marking small mammals to prevent recounting individuals that may be recaptured during a 3-day trapping session (Hall and Willig 1994, Petryszyn and Russ 1996). We did not use unique identification marks. We sampled each 1996 site once during 10 - 26 July and each 1997 site once during 24 May - 1 June.

We compared vegetation characteristics for treatments with those of references using a randomized block analysis of variance (ANOVA) (Sokal and Rohlf 1995:352-356; Zar 1996:254-259). Blocking can reduce and control experimental error variance to achieve greater precision (Kuehl 1994:256). We analyzed these data with grid pairs as blocks to account for extraneous variation among pairs.

We generated estimates of combined rodent population size ($\hat{N}$) for each trapping grid with removal methods using program CAPTURE (Otis et al. 1978). Neither actual removal from the area nor kill-trapping is necessary to apply removal analysis methods to small mammal trapping data (White et al. 1982:101). All captured individuals were effectively removed from the unmarked population by marking them (Seber 1973:323). Therefore, we used only initial captures (i.e., new captures) to create capture history matrices and generate population estimates (White et al. 1982:101). We selected model M_{bh}-Pollock and used the variable probability removal estimator (Pollock and Otto 1983) for population estimation because it is robust to violations of the assumption of declining recapture rates during successive trapping occasions. We used natural log (ln) transformations to remove or reduce heteroscedasticity among population variances (Sokal and Rohlf 1995:413). To report these data in linear scale, we used the antilog to back-transform means (Sokal and Rohlf 1995:413).

Individual species exhibited low capture rates precluding species population estimates. Therefore, for species analyses, we were limited to comparing relative abundance indices (no. individuals / 100 trap nights) between treated and reference areas. We are aware of the weaknesses and limitations of abundance indices in long-term ecological monitoring studies (Rexstad 1994). However, for the purposes of this short-term study (i.e., comparisons between 3-day sampling periods), the number of animals trapped / 100 trap nights on a given grid seems a reasonable measure for comparisons. We assumed that equal proportions of treatment and reference populations were sampled during each sampling period, thereby making our indices appropriate (Lancia et al. 1994). We examined the relationship between combined species relative abundance indices and combined species population estimates for the 24 grids sampled and they were highly correlated (r = 0.955) suggesting that there is validity to comparisons of relative abundance.

We compared mammal community characteristics for treatments with those of references using a randomized block ANOVA. We analyzed these data with grid pairs as blocks to account for extraneous variation among treatment-reference pairs. Additionally, we examined the relationships between rodent abundance and vegetation features (i.e., percent cover) using simple linear regression. Minimum level of significance for all comparisons was $\alpha = 10\%$.

RESULTS

Both shrub ($F_{1,11} = 12.14$, $P = 0.005$) and tree ($F_{1,11} = 20.34$, $P < 0.001$) cover were significantly greater on reference grids compared to treated grids (Table 2). Dead but standing woody plant cover was significantly greater on treated grids compared to references ($F_{1,11} = 5.96$, $P = 0.03$). There was no significant difference in percent cover of grass, litter, suffrutescents, or bare ground between treatment and reference grids.

We captured 560 individuals representing 21 rodent species in 3,528 trap nights. Heteromyids made up the majority (80.2%) of individuals trapped with kangaroo rats (Dipodomys spp.; 59.3%) more prevalent than pocket mice (Perognathus and Chaetodipus spp.; 20.9%). Merriam's kangaroo rat (D. merriami) was the most abundant species comprising 46.4% of all individuals captured. Because of difficulties in distinguishing between sub-adult deer mice (Peromyscus maniculatus) and white-footed mice (P. leucopus), these species were pooled for analyses that considered species separately. To increase sample sizes we also grouped cogeneric species with similar life histories and low capture rates for ≥ 1 of the species involved. These grouped species included northern and southern grasshopper mice (Onychomys leucogaster and O. torridus, respectively), desert and rock pocket mice (Chaetodipus penicillatus and C. intermedius, respectively), western and fulvous harvest mice (Reithrodontomys megalotis and R. fulvescens, respectively), and yellow-nosed and Arizona cotton rats (Sigmodon ochrognathus and S. arizonae, respectively).

The greatest number of species trapped on a grid was 12 and the fewest species trapped was 3. Species richness was similar between reference and treatment grids ($F_{1,11} = 0.79$, $P = 0.39$) (Fig. 1, Table 3). Population estimates for all rodent species combined were not significantly different between treated and reference grids ($F_{1,11} = 1.24$, $P = 0.29$) (Fig. 2).

There was no significant difference in overall relative abundance of rodents ($F_{1,11} = 0.24$, $P = 0.63$) between treated and reference grids (Table 3). Relative abundance of Ord's kangaroo rat (D. ordii) was significantly greater on treated compared to reference areas ($F_{1,11} = 4.23$, $P = 0.06$). White-throated woodrats (Neotoma albigula) and Harris' antelope squirrels (Ammospermophilus harrisii) were significantly more abundant on references compared to treated areas ($F_{1,11} = 3.67$, $P = 0.08$), however, both were infrequent captures comprising < 4.0 % of all individuals detected.

Rodent populations exhibited a negative relationship with increasing bare ground ($r^2 = 0.173$, $P = 0.04$). We could not identify any additional relationships between rodent communities and cover variables measured. Merriam's kangaroo rats exhibited a negative relationship with increasing grass cover ($r^2 = 0.322$, $P < 0.01$) (Fig. 3). We found no other relationships between Merriam's kangaroo rats and vegetation characteristics. Ord's kangaroo rat exhibited a negative relationship with increasing bare ground ($r^2 = 0.178$, $P = 0.04$) and positive relationships with increasing forb cover ($r^2 = 0.197$, $P = 0.03$) and grass cover ($r^2 = 0.312$, $P \leq 0.01$) (Fig. 4). White-throated woodrats were related positively to percent cover of litter ($r^2 = 0.130$, $P = 0.08$), trees ($r^2 = 0.120$, $P < 0.01$), shrubs ($r^2 = 0.178$, $P = 0.04$), and suffrutescent species ($r^2 = 0.124$, $P = 0.09$) and negatively to increasing bare ground ($r^2 = 0.141$, $\underline{P} = 0.07$). No other relationships were identified between rodent species and cover.

DISCUSSION

The results from vegetation sampling were as expected. Cover of woody plants was greater on reference areas compared to treatments. This may illustrate the effectiveness of mechanical techniques in reducing woody plants for several years after initial treatment. Forbs were more prevalent on treated areas where they had less competition from woody plants and where grass was still sparse. According to Fulbright (1996), rootplowing is an intense disturbance that results in low successional species. The disturbance caused by the mechanical manipulations established an earlier seral stage community including many forbs.

Heteromyids made up the majority of individuals captured and Merriam's kangaroo rats were the most abundant species. Rodent species richness was not different between treatments and references. Nolte (1995) reported that species richness of small mammals did not differ between treated and nontreated areas 1 and 2 years following herbicide application to control mesquite and cacti (Opuntia spp.). In another study, species richness of small mammals increased following chaining of pinyon (Pinus spp.) - juniper (Juniperus spp.) (Sedgewick and Ryder 1987).

Changes in species richness should be examined following treatment application to identify any trends that may exist. We attempted to identify such trends between references and treatments under the assumption that rodent communities on treated areas were similar to those of references prior to treatment application. Wood (1969) found greater species richness (11 spp.) at the lowest successional stages compared to the climax (6 spp.). In our study, treated areas were at lower successional stages than were the references, however, we did not identify any clear patterns for rodent species richness on treatments compared to references.

Rodent population estimates for all species combined were similar for treatments and references. Due to low capture rates for individual species, we were limited to comparisons of relative abundance for analyses at the species level. Ord's kangaroo rats were more abundant on treatments compared to references. Ord's kangaroo rats inhabit a variety of vegetation communities throughout the southwest (Hoffmeister 1986:298). They have been found on alluvial fans of the mountains in southeastern Arizona, in mesquite, grasses, and cacti of central and southern Arizona, in fine sands, mesquite, and some grasses, and in alluvial fans among mesquite and yuccas (Yucca spp.) (Hoffmeister 1986:298). Ord's kangaroo rats have increased in density where junipers have been bulldozed. Hoffmeister (1986:298) attributes that increase to increased grass seed availability and open landscape. Both of these changes also accompanied the treatments in the present study. Six of the 7 sites sampled were re-seeded after treatment application and all treated areas exhibited reduced cover of woody plants compared to references. Abramsky (1978) reported an invasion or colonization of Ord's kangaroo rats following food (seed) supplementation.

White-throated woodrats and Harris' antelope squirrels were more abundant on references compared to treatments. Woodrats would be expected to decline following brush removal because of their den-building behavior. Brown et al. (1972) reported the dependence of woodrats on cholla (Opuntia bigelovii) density. In our study area, the woodrat's primary building material seemed to be mesquite. Woodrats rely on their dens for protection from predation (Brown et al. 1972) and for microclimate temperatures and humidities that enable them to withstand desert extremes (Lee 1963, Brown 1968 cited in Brown et al. 1972). The low-growing shrub-form mesquite on reference areas may have provided woodrats with both shelter and dietary requirements as cacti do in other parts of their range (Brown et al. 1972, Hoffmeister 1986:406). Mesquite thorns may serve the defensive purpose of cacti spines and green mesquite pods may provide the water obtained from succulent cacti in other areas. The relationship between Harris' antelope squirrels and reference areas is not as clear because only 3 (0.5%) individual squirrels were captured, all on reference areas. Our trapping methods were designed to estimate nocturnal rodent species and did not accurately estimate abundance of diurnal species.

Combined-species population size exhibited a negative relationship with increasing bare ground. This is not surprising given the increased risk of predation associated with open ground (Kotler et al. 1988, Longland and Price 1991). Species-vegetation relationships were only identified for 3 species. Merriam's kangaroo rat was negatively related to increasing grass cover. Reynolds (1950) observed Merriam's kangaroo rat's preference for areas of lower grass density and attributed this to predation avoidance. This species is thought to have a competitive advantage in open habitats due to its predation avoidance characteristics (Thompson 1982). The dense grass may interfere with the species' ability to detect and evade predators. According to Findley et al. (1975:183), Merriam's kangaroo rats are one of few mammals to inhabit degraded grasslands dominated by creosotebush and are common on desert pavements and in gravely bajadas. Within rangelands, this species may prefer areas of annual grasses and scattered woody plants (Reynolds 1958). Price (1978) demonstrated an increase in Merriam's kangaroo rats with the removal of large shrubs. We did not identify any direct relationship between Merriam's kangaroo rats and woody cover.

Ord's kangaroo rat was related negatively to increasing bare ground and positively to increasing forb and increasing grass cover. Schroder and Rosenzweig (1975) found Ord's kangaroo rats more abundant in grassier areas compared to Merriam's kangaroo rats, and suggested this preference as a possible mechanism of their coexistence. Whitford et al. (1978) also noted that increased grass and reduced shrub cover favored Ord's kangaroo rats, whereas the shrub-dominated area favored Merriam's kangaroo rats. However, Schroder (1987) did not observe a preference by Ord's kangaroo rats for large grass patches. Although we did catch both species on several grids, it does seem likely that they have reduced direct competition via differential use of vegetation features, particularly that of grass cover at moderate levels. Woodrats exhibited a positive relationship with percent cover of litter, trees, shrubs, and suffrutescent species and a negative relationship with bare ground as would be expected given their dependence on nests discussed previously.

Rodent abundance does not seem to differ between previously brush-controlled and nonmanipulated areas as predicted. Reliable relationships between particular rodent species and treatment-induced vegetation manipulations were not possible for the majority of species captured. It does seem that Merriam's kangaroo rats can maintain or re-establish populations at pre-disturbance levels following mechanical brush control. This species has been documented as preferring open areas with little canopy cover (Reynolds 1950,1958; Rosenzweig 1973; Petryszyn and Russ 1996) and, therefore would be expected to persist or recolonize despite the removal of woody vegetation. Ord's kangaroo rat may increase in abundance following shrub control especially if grass seed is added and sparse grasses are re-established.

The similar rodent abundance on treated and nontreated ranges might be related to their diets. Most of the rodents captured are granivorous or at least consume some seeds as a large portion of their diets. These animals would be expected to thrive in areas with greater grass cover and seed production, or in areas that were previously seeded and may serve as sinks (e.g., treatments). However, one would also expect rodents to persist in areas with substantial cover from predation where seeds may concentrate beneath shrubs (e.g., reference areas). Cover was certainly reduced by woody plant control, but did not seem to have a major effect on rodent numbers several years later. Grass cover was similar between treated and reference areas, but seed availability may have been greater on treatments due to re-seeding efforts. Additionally, soils were loosened and perhaps more suitable for burrowing and seed retrieval following rootplowing and rollerchopping. Small pocket mice probably have difficulty burrowing through compact soils (Rosenzweig and Winakur 1969). Rodent burrows were observed in the slopes of furrows created by rootplowing.

Stephen's kangaroo rat (Dipodomys stephensi) densities increased following manual removal of shrubs in California (Price et al. 1994). Intensive brush control treatments including herbicides, grubbing, and chaining did not harm the Texas kangaroo rat (D. elator) population, but may have increased or enhanced the habitat for that species (Stangl et al. 1992). Stangl et al. (1992) state that the effects on kangaroo rats of more destructive methods such as rootplowing are unknown. Our results suggest that rootplowing did not negatively impact Merriam's and Ord's kangaroo rats and may have improved their habitat conditions.

Powell (1968) found rodents significantly more abundant on previously rootplowed and rollerchopped areas than on nontreated areas in Texas and attributed that difference to more favorable cover (i.e., increased cover of grasses and herbaceous litter). Our results also suggest that rodents will persist or recolonize brush controlled ranges, but not at levels significantly greater than those for nontreated areas. Powell's (1968) captures were dominated by pygmy mice and cotton rats, species heavily dependent on dense cover. Most of the treated areas in our study did not have sufficient grass cover to support substantial populations of those grassland species. Our captures were predominately heteromyid rodents and dominated by Merriam's kangaroo rats, a species associated with open ground and scattered shrubs.

Pooling rootplowing, grubbing (tree dozing), and rollerchopping into 1 mechanical treatment increased the sample size but perhaps erroneously assumed that treated areas would be similar in testing for an overall effect. Ideally, several replicates for each mechanical treatment should be sampled along with complementary reference areas. This study was a first step and we feel that a manipulative experimental study would provide additional information regarding the actual impacts of mechanical brush treatments on resident rodent communities.

MANAGEMENT IMPLICATIONS

This effort should clarify some of the long-term impacts of brush control on rodent species and their habitats within the semi-desert grassland ecosystem. If the goal is to restore shrub-dominated communities to grassland condition, some consideration must be given to other organisms that currently inhabit or have previously inhabited those ranges. Small mammals have a major role at the consumer trophic level, as seed dispersers, and as the major prey base for many other species. Habitat alterations such as brush control are likely to have effects on nontargeted species such as rodents, and these effects may lead to unanticipated changes in plant and animal populations. Knowing the abundance of particular rodent species may be essential to success of arid rangeland restoration projects. Grassland dependent species such as pygmy mice and cotton rats may indicate healthy grassland conditions and may persist or increase with increasing grass cover. However, species that consume huge quantities of grass seed and are associated with large proportions of open ground, such as kangaroo rats, may prevent the realization of management objectives.

Rodent communities did not differ in size or richness between treated and reference plots, suggesting that rodent communities are not drastically affected by range manipulations. The fact that Merriam's kangaroo rat numbers were similar on treatments compared to references may preclude restoration of grassland states without additional seeding and shrub control efforts. However, the greater abundance of Ord's kangaroo rats on treated areas may represent a sign of improved grassland conditions.

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Table 1. Sites sampled for study of potential impacts of mechanical brush control on desert rodents in southeastern Cochise Co., Arizona during 1996 and 1997.

Site	Treatment	Area (ha)	Yr treated	Yr sampled
Malpai Ranch	rootplowed & reseeded	125	1994	1996
47 Ranch	rootplowed & reseeded	69	1995	1996
Kimble Ranch	mesquite grubbed	49	1992	1996
Ft. Huachuca	rootplowed & reseeded	4	1993	1997
Mallet Ranch	rootplowed, grubbed, & reseeded	100	1991	1997
Lee Stn. Ranch	roller chopped & reseeded	81	1996	1997
Lee Stn. Ranch	roller chopped & reseeded	65	1996	1997

Table 2. Percent cover for 12 previously brush-treated and 12 reference rodent trapping grids in southeastern Cochise Co., Arizona 10-26 July 1996 and 24 May - 1 June 1997. Means were tested using a randomized block ANOVA and were considered different at $\underline{P} \leq 0.10$.

Cover type	Treatment		Control		P-values
	Mean	SE	Mean	SE	
bare ground (%)	48.33	3.78	45.75	5.09	0.57
litter cover (%)	27.75	2.84	33.42	4.16	0.21
forb cover (%)	5.92	1.75	1.75	0.76	0.05
grass cover (%)	13.25	3.22	9.25	2.42	0.23
shrub cover (%)	1.58	0.86	13.00	3.88	< 0.01
half shrub cover (%)	9.50	2.42	12.25	3.30	0.45
tree cover (%)	0.17	0.17	16.58	3.65	< 0.01
dead woody plant cover (%)	4.67	0.96	2.33	0.36	0.03

Table 3. Mean rodent relative abundance, species richness, and community size on 12 previously treated and 12 reference desert grassland trapping grids, Cochise Co., Arizona 10-26 July 1996 and 24 May - 1 June 1997. All treatments consisted of mechanical shrub control. Means were tested using a randomized block ANOVA and were considered different at $\underline{P} \leq 0.10$.

Rodent Species and Community Characteristics	Treatment		Reference		P-values
	Mean	SE	Mean	SE	
Dipodomys merriami	8.05	1.89	6.69	1.08	0.50
Dipodomys ordii	2.78	1.54	1.08	0.74	0.06
Dipodomys spectabilis	0.11	0.11	0.11	0.11	1.00
Chaetodipus baileyi	0.40	0.40	0.00	0.00	0.34
Chaetodipus penicillatus + intermedius	2.15	0.86	2.89	0.66	0.31
Perognathus flavus	0.96	0.68	0.06	0.06	0.22
Perognathus hispidus	0.11	0.11	0.11	0.08	1.00
Peromyscus maniculatus + leucopus	1.08	0.40	1.53	0.47	0.52
Peromyscus eremicus	0.17	0.17	0.11	0.11	0.78
Onychomys torridus + leucogaster	0.51	0.12	0.68	0.31	0.61
Reithrodontomys megalotis + fulvescens	0.06	0.06	0.17	0.17	0.55
Sigmodon ochrognathus + arizonae	0.06	0.06	0.45	0.27	0.11
Neotoma albigula	0.23	0.13	0.91	0.40	0.08
Baiomys taylori	0.11	0.08	0.06	0.06	0.59

Table 2. Continued

Rodent Species and Community Characteristics	Treatment		Reference		P-values
	Mean	SE	Mean	SE	
Ammospermophilus harrisii	0.00	0.00	0.17	0.09	0.08
Spermophilus spilosoma	0.11	0.11	0.00	0.00	0.34
Relative abundance (all species)	16.44	1.49	15.36	1.86	0.63
No. captures	35.25	3.88	35.42	6.10	0.34
Species richness	4.92	0.71	5.50	0.60	0.39
Population size (all species)	27.11	1.15	32.47	1.10	0.29

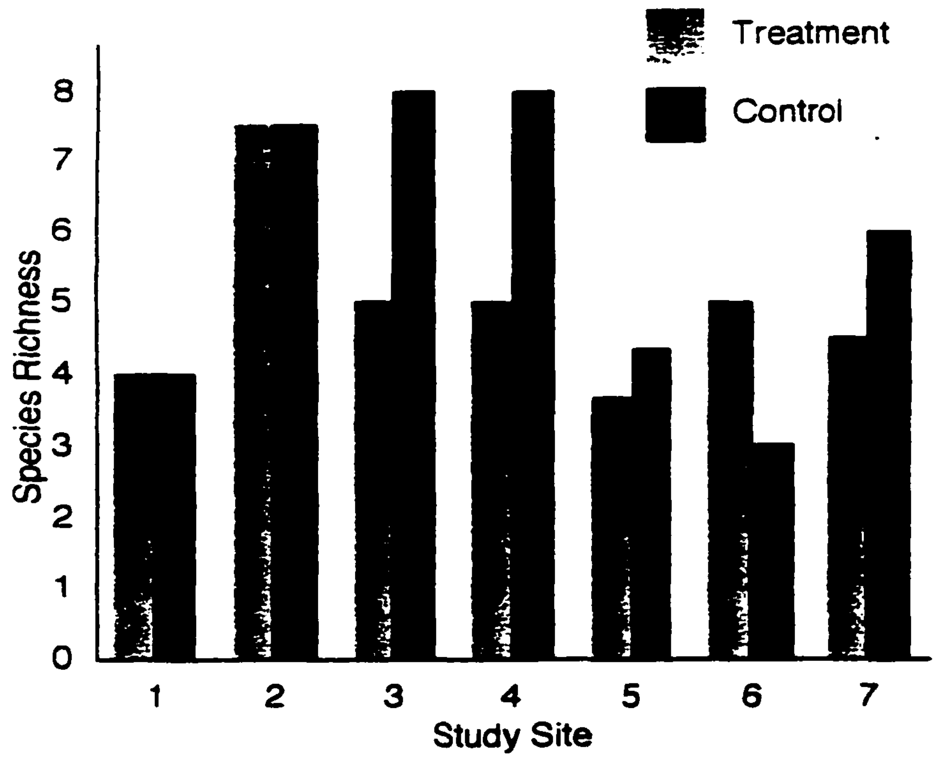

Fig. 1. Mean rodent species richness on 7 paired brush-controlled and non-treated study sites in southeastern Cochise Co., Arizona. Three sites were sampled during 10-26 July 1996 and 4 were sampled during 24 May - 1 June 1997. There was no significant difference in rodent species richness between treatment and reference areas ($\underline{F}_{1,6}$ = 1.05, $\underline{P}$ = 0.34).

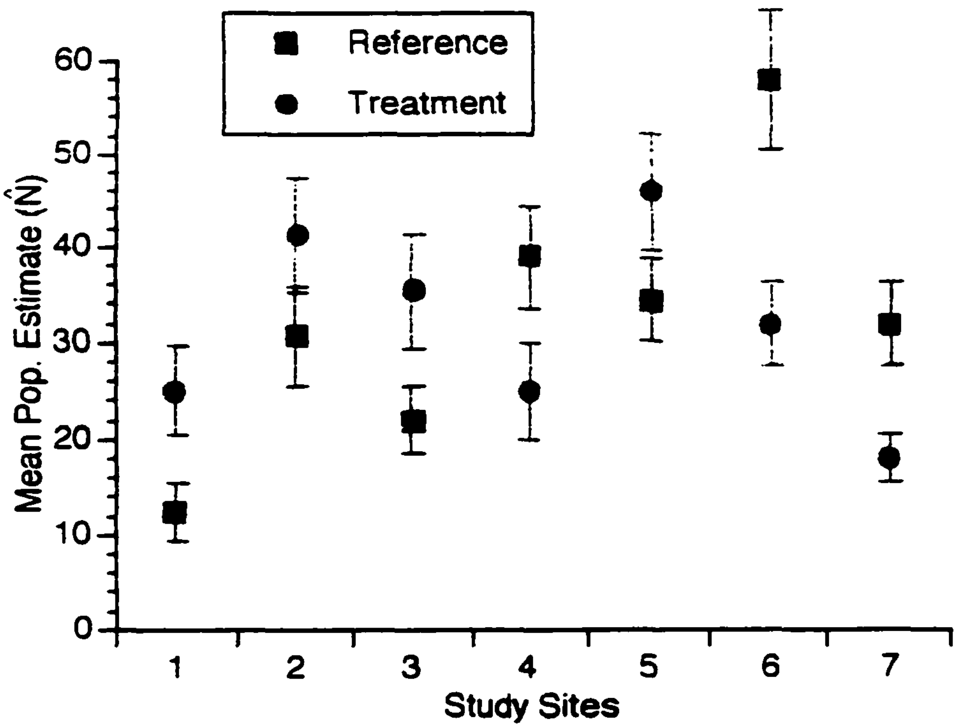

Fig. 2. Mean estimates of rodent community size ($\hat{N} \pm$ SE) on 7 paired brush-controlled and non-treated study sites in southeastern Cochise Co., Arizona. Three sites were sampled during 10-26 July 1996 and 4 were sampled during 24 May - 1 June 1997. Population estimates were not significatnly different between treatment and reference areas ($\underline{F}_{1,6}$ = 0.008, $\underline{P}$ = 0.93).

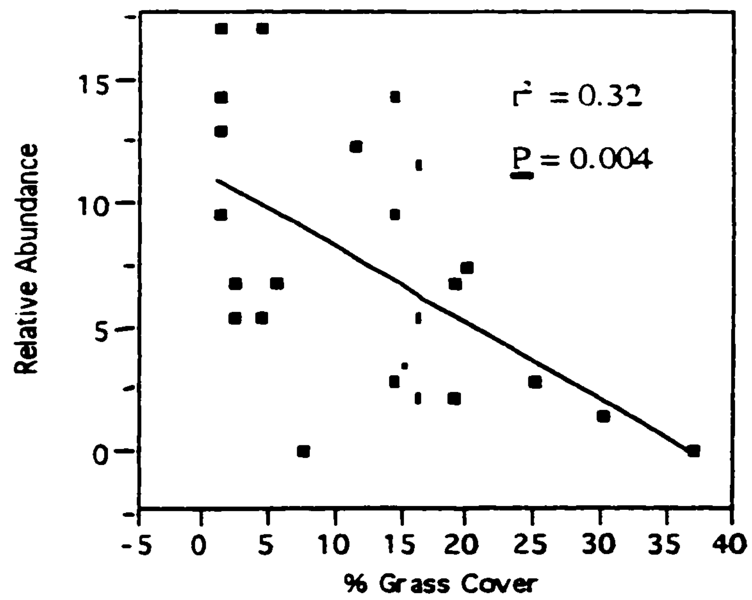

Fig. 3--Relationship between relative abundance of Dipodomys merriami and % grass cover on 12 paired brush-controlled and non-treated trapping grids in southeastern Cochise Co., Arizona. Minimum level of significance was set at α = 0.10.

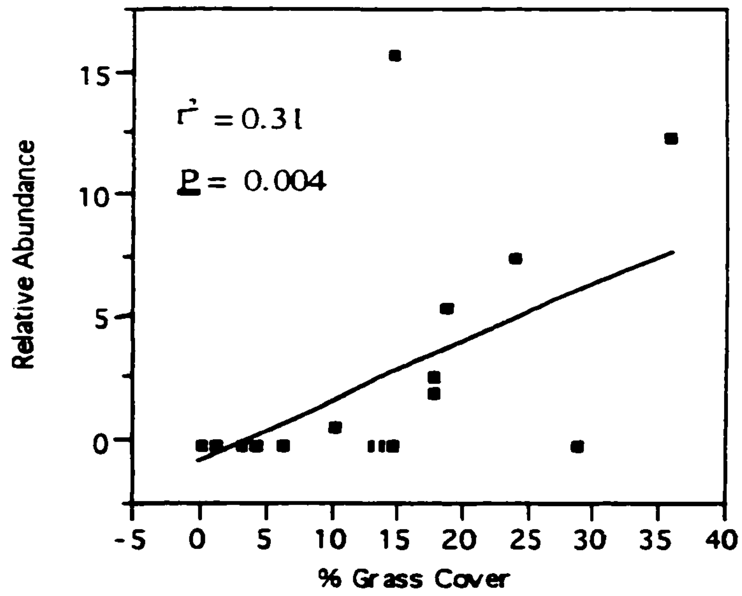

Fig. 4--Relationship between relative abundance of Dipodomys ordii and % grass cover on 12 paired brush-controlled and non-treated trapping grids in southeastern Cochise Co., Arizona. Minimum level of significance was set at $\alpha = 0.10$.

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RODENT COMMUNITIES ON GRAZED AND NON-GRAZED DESERT GRASSLANDS

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ABSTRACT--We examined the potential effects of livestock grazing on vegetation characteristics and rodent species richness, diversity, and abundance in a shrub-encroached desert grassland of southeastern Arizona. We did not identify any significant differences in vegetation cover between grazed and ungrazed plots. Rodent species richness and population sizes were significantly greater on ungrazed plots compared to grazed plots. There was no significant difference in rodent diversity between grazed and ungrazed plots. Rodent populations exhibited a positive relationship with cover and density of shrubs and with pooled shrub and tree density. Our data provide useful information regarding rangeland biodiversity and desert grassland restoration efforts.

The impacts of livestock grazing on ecological processes, biodiversity, and ecosystem functioning of desert rangelands have recently been examined (West, 1993; Brussard et al., 1994; Fleischner, 1994; Noss, 1994). Some work has been focused on the role of grazing in desert grassland conversion to shrubland (Smith and Schmutz, 1975; Brady et al., 1989; Holochek et al., 1994; Bock and Bock, 1997). Others have examined the impacts of grazing on desert rodents (Bock et al., 1984; Heske and Campbell, 1991; Rosenstock, 1996; Hayward et al., 1997). We were also interested in differences between rodent communities on grazed and ungrazed plots, but our intention was to gather baseline information for a woody plant control study. Although the brush control treatment was postponed, we thought our baseline data might be useful in understanding some of the impacts of grazing on rodent communities and that it might aid in understanding ecosystem responses to grazing. Furthermore, grazing has been proposed as a management tool in some areas and we wanted to explore its potential for managing desert shrub-grasslands.

During the past 100-150 years woody vegetation such as mesquite (Prosopis spp.), acacia (Acacia spp.), snakeweed (Gutierrezia sarothrae), and burroweed (Isocoma tenuisectus) have increased within the Chihuahuan Desert grasslands (Holochek et al., 1994). Some researchers suggest that these woody species are not invaders, but are residents that have increased within their ranges (Bahre, 1991). Ranchers in the borderlands region of southeastern Arizona and southwestern New Mexico are committed to restoring grassland conditions via improved grazing practices, prescribed burning, and woody plant control.

Grazing affects many aspects of grassland ecosystems, including species composition and diversity of plants, primary productivity, standing crop biomass, soil compaction, and soil moisture (Grant et al., 1982). Grazing-induced changes in plant community structure and composition may benefit some wildlife species while negatively affecting others (Kie and Loft, 1990). Even moderate grazing by large herbivores reduces cover drastically and thereby affects small mammal communities (Grant et al., 1982). Reynolds (1958) reported that Merriam's kangaroo rats (D. merriami) increase on grazed lands because livestock disperse mesquite which causes reductions in perennial grass cover. Grazing decreased the relative abundance of herbaceous vegetation in Great Basin communities (Hanley and Page, 1981). In that study, rodent species dependent upon perennial herbs for food and cover, were consistently less abundant (or absent) within grazed compared to ungrazed communities. Reynolds (1980) reports that neither grazing nor planting a sagebrush range significantly changed the total abundance of small mammals, however, each treatment significantly reduced species diversity of small mammals.

Our objectives were to determine the composition, percent cover, and density of vegetation within grazed and ungrazed semi-desert rangelands. We also wanted to determine the species composition, diversity, and relative abundance of rodents on grazed and ungrazed ranges. Additionally, we wanted to identify any potential relationships between rodent communities and vegetation characteristics. We predicted that overall rodent abundance, species-specific abundance, and diversity would be different between grazed and ungrazed ranges due to grazing-induced differences in vegetation cover and density.

STUDY AREA--We established our study site in Cochise County, Arizona east of Douglas at San Bernardino National Wildlife Refuge and the adjacent Malpai Ranch (109°16′ N 31°20′ W). According to Mitchell (1976), the climate is controlled by the summer monsoons occurring from July-September. Most precipitation occurs during summer with occasional rains in winter. Annual precipitation was approximately 30 cm (Wendy Glenn, Malpai Borderlands Group, pers. commun.) and typical of the region which ranges from 20 - 42 cm (Smith and Schmutz, 1975; Brown, 1982; NOAA, 1996). Annual mean temperatures are moderate, averaging 12-20 ° C, but summers are hot. Elevations range from 1,150 - 1,200 m. The site is dominated by tobosa grass (Hilaria mutica), white-thorn acacia (Acacia constricta), velvet mesquite (Prosopis velutina), tarbush (Flourensia cernua), and snakeweed (Gutierrezia sarothrae), but exhibits considerable variability from one end of the study area to the other. Both grazed and ungrazed plots range from areas with dense shrubs and suffrutescents (i.e., half-shrubs) virtually absent of grasses to areas of dense tobosa grass with scattered shrubs. The ranch was open to cattle grazing at about 18 ha per animal unit (AU). The refuge has been protected from grazing since its establishment in 1982 and served as the reference for the grazing treatment on the Malpai Ranch.

MATERIALS AND METHODS--We used a paired comparison sampling design to compare grazed and ungrazed plots. Although it is unlikely to select field plots with exactly the same conditions of soil, moisture, wind, slope, and other factors, it should be possible to set up two plots with similar environmental conditions (Zar, 1996:165). The reference plots must be a representative sample of locations of the same general area as that in which the impact occurred (Underwood, 1994). The problem of confounding (or pseudoreplication) due to comparisons of abundance between a disturbed and a control location should be overcome by having several "replicated" disturbed and several control locations (Underwood, 1994). The study site included an area that has been grazed (ranch) and a nearby area where livestock have been excluded for 14 years (refuge). We located three reference plots based on similarity (i.e., general soil type, slope, elevation, and aspect) and proximity to three corresponding grazed plots. We did not establish these plots randomly because we were confined to the area of the refuge closest to the grazed area and wanted to use as many plots as possible. Therefore, we located independent plots along a line approximately 100 m within the refuge boundary and corresponding plots approximately 100 m within the ranch. We were able to fit three grazed-ungrazed paired plots within the refuge and ranch and still maintain independence between adjacent plots.

We set one 100 m line transect diagonally from a randomly selected corner of each trapping grid and measured percent cover and density along the transects. We used the point-intercept method to estimate cover (i.e., the total ground area covered by various plant species, bare soil, rocks, or litter). We recorded the type of contacts made along a vertical line at 100 points (i.e., every 1.0 m) along each transect (Bock and Bock, 1978).

When vegetation was intercepted we recorded the species. Dead but standing trees and shrubs were recorded separately from living woody plants. We categorized mesquite and acacia as trees; all other woody species were recorded as shrubs or suffrutescents. When the canopies of ≥ 2 plants directly overlapped at a given point, we recorded multiple contacts. Non-vegetation contacts were recorded as pebble (< 5 cm) or cobble (≥ 5 cm), or as bare ground if no vegetative cover, litter, pebble, nor cobble was intercepted.

We estimated density of woody trees and shrubs such as mesquite, creosotebush, and tarbush using belt transects. We centered a 1.0 m belt on each 100 m transect and counted the number of individual plants of a species rooted within 0.5 m on either side of the transect..

We set up six trapping grids within the study site, three on grazed and three on ungrazed areas. We established one grid 100 m from the refuge boundary and located two successive grids at approximately 500 m intervals (Santillo et al., 1989) along a line within the treatment boundaries. We located control grids at a distance of ≥ 200 m (Hall and Willig, 1994) perpendicular to corresponding treatment grids. We set trapping grids in a 7 x 7 trap configuration (i.e., 49 traps) with 15 m between consecutive trap stations (Brown and Zeng, 1989; Bowers and Brown, 1992; Heske et al., 1994). We placed one 7.5 x 8.75 x 22.5 cm folding Sherman live trap at each trap station and marked the stations with pin flags. We trapped rodents on each grid for three consecutive nights. Two grids, a treated grid and its corresponding control grid, were trapped simultaneously. We set traps baited with mixed birdseed ≤ three hours before sunset and checked traps the following morning ≤ three hours after sunrise. We placed polyester filling in traps to prevent hypothermia. Additionally, we partially buried traps in the ground (leaving the doors exposed) as a further precaution against hypothermia overnight. This also reduced the risk of heat stress during daylight hours prior to the release of captured animals. We avoided trapping during the 6-10 days before and after a full moon because moonlight reduces nocturnal rodent activity (Carley et al., 1970; Kaufman and Kaufman, 1982; Kotler, 1984; Brown et al., 1988; Longland and Price, 1991). We recorded the species, age, sex, reproductive condition, weight, and trap station for all captured individuals. We marked animals using permanent ink and released all animals at capture locations. Permanent ink markers have been used for temporarily marking small mammals to prevent recounting individuals that may be recaptured during a 3-day trapping session (Hall and Willig, 1994; Petryszyn and Russ, 1996). We did not give individual animals unique marks. We used a different color ink for marking animals in each grid to determine if rodents were moving between grazed and ungrazed grids or among paired grids. This procedure assured the independence of sampling units.

According to Hurlbert (1984), multiple plots within a single treatment are not identical. Therefore, by sub-sampling the treatment area, we achieved a better representation of the potential treatment impacts over a wider range of habitat conditions. This sampling design is referred to as clumped segregation by Hurlbert (1984) because it lacks treatment interspersion. Because we sampled an area that had already been grazed, it was impossible to achieve treatment interspersion. However, we selected each pair of grids based on inherent (assumed pre-treatment) similarities between treatment and control locations, such that the (pre-treatment) variability within a pair of grids was less than that among all grids. Samples were not replicated at other ungrazed sites, thereby limiting the scope of inference from this study.

We generated estimates of rodent community size (i.e., all species combined) for each trapping grid using Program CAPTURE's removal methods (Otis et al., 1978). Neither actual removal from the area nor kill-trapping is necessary to apply removal analysis methods to small mammal trapping data (White et al., 1982:101). All captured individuals were effectively removed from the unmarked population by marking them (Seber, 1973:323). Therefore, we used only initial captures to generate population estimates

(White et al., 1982:101). We selected model M_{bh}-Pollock and used the variable probability removal estimator (Pollock and Otto, 1983) for population estimation because it is robust to violations of the assumption of declining recapture rates during successive trapping occasions. We used natural log (ln) transformations to remove or reduce heteroscedasticity among population variances (Sokal and Rohlf, 1995:413). To report these data in linear scale, we used the antilog to back-transform means (Sokal and Rohlf, 1995:413).

Capture success was low for individual species therefore we could not generate species population estimates. Instead, we calculated the relative abundance (no. individuals captured / 100 trap nights) for each species at each grid.

We conducted a paired comparison of grids with a paired t-test. Paired comparisons often result from dividing an individual unit such that half receives a treatment and the other half a control (Sokal and Rohlf, 1995:352-355). Therefore, we were assuming that each grazed and ungrazed pair was formerly a homogeneous unit and that any dichotomy was the effect of grazing half of that unit.

We measured species richness based on the number of individuals captured for each species. We measured diversity with the Shannon-Wiener index (H') (Pielou, 1966; Zar, 1996:39). We compared species diversity and richness values for differences between grazed and ungrazed grids with paired t- tests. We also used linear regression to identify relationships between rodent communities and vegetation features. For all tests, we set α = 0.10.

RESULTS--Statistically significant differences in vegetation characteristics were not identified (Table 1). However, large absolute differences were identified for some features and these may be biologically meaningful. Percent grass cover was 60% greater on ungrazed grids (33.3%) compared to grazed grids (20.7%). Shrub cover was nearly three times greater on ungrazed grids (9.7%) compared to grazed grids (3.3%). Cover of suffrutescents was nearly three times greater on grazed grids (9.3%) compared to ungrazed grids (3.3%). Tree cover was 35% greater on grazed grids (16.7%) compared to ungrazed grids (12.3%). Density of woody plants was more than twice as great on grazed plots (131.0 / ha) compared to ungrazed plots (61.7 / ha). Tree densities were virtually identical between grazed (13.3 / ha) and ungrazed plots (13.7 / ha). Shrub density was 80% greater on ungrazed plots (15.7 / ha) compared to grazed plots (8.7 / ha). Density of half shrubs was more than three times as great on grazed plots (105.7 / ha) compared to ungrazed plots (32.3 / ha).

We trapped 167 individual rodents representing 13 species during 882 trap nights (Table 2). In preliminary sampling within the study area, we trapped an additional 3 species: yellow-nosed cotton rat (Sigmodon ochrognathus), fulvous cotton rat (S. fulviventer), and banner-tailed kangaroo rat (Dipodomys spectabilis). Heteromyids comprised the majority (73.6%) of individuals captured. Bailey's pocket mouse (Chaetodipus baileyi) was the most abundant species (42.5%) followed by Merriam's kangaroo rat (Dipodomys merriami) (27%). Species richness was significantly greater on ungrazed grids compared to grazed grids (t = 3.00, 2 d.f., P = 0.096). Species diversity (H′) did not differ significantly between grazed and ungrazed grids (t = 2.45, 2 d.f., P = 0.13). There was no significant difference in overall relative abundance of rodents between grazed and ungrazed grids (t = 1.15, 2 d.f., P = 0.37). Rodent population estimates ($\hat{N}$) were significantly greater for ungrazed compared to grazed grids (t = 3.77, 2 d.f., P = 0.06). No significant differences in relative abundance were detected for particular species between grazed and ungrazed grids. Rodent populations were positively related to shrub cover (r^2 = 0.536, P = 0.098) and shrub density (r^2 = 0.552, P = 0.09) (Figs. 1 and 2, respectively). Rodents also exhibited a positive relationship with pooled shrub and tree density (r^2 = 0.691, P = 0.04) (Fig. 3).

DISCUSSION--The cover of trees was slightly greater on grazed compared to ungrazed areas, but tree densities were virtually equal. Velvet mesquite densities have increased at rates of 4.4 to 26.7 plants/ha/year on plots protected from cattle for 18 to 25 years (Glendening and Paulsen, 1955). Although the continued rapid increase in velvet mesquite on a protected range was not as rapid as on a grazed range, it still represented a threat to range improvement (Smith and Schmutz, 1975). We found grass cover 60% greater on ungrazed grids compared to grazed grids. Others have also noted reduced grass cover in grazed areas or increased grass cover within exclosures (Chew, 1982; Bock and Bock, 1993). Shrub cover was approximately three times greater and shrub density was nearly twice as great on the refuge compared to the ranch. Bock and Bock (1997) also found greater shrub densities within an ungrazed sanctuary and attributed the difference to the absence of browsing by livestock. Bock and Bock (1997) included burroweed, a suffrutescent or half-shrub, in their shrub sampling and found its density greater on ungrazed areas. We found cover and density of suffrutescent plants about three times greater on the ranch compared to the refuge. Broom snakeweed was the only suffrutescent species encountered. Snakeweed has increased on grazed and ungrazed areas following 19 years of cattle exclusion (Chew, 1982).

The dominance of heteromyid rodents within our study area was consistent with that of other researchers within the region (Brown and Zeng, 1989). We did not find rodent species diversity nor relative abundance to differ significantly between grazed and ungrazed grids, however there was a trend toward greater diversity on ungrazed grids. Species richness and rodent community estimates were significantly greater for ungrazed compared to grazed grids. Similarly, Rosenstock (1996) reported 50% greater species richness and 80% greater abundance for small mammals on ungrazed sites compared to grazed sites. Heske and Campbell (1991) also reported greater rodent abundance within livestock exclosures. However, Reynolds (1950) noted greater abundance of Merriam's kangaroo rats outside of grazing exclosures. Other studies have suggested that historically grazed plots may have fewer small mammals than ungrazed plots, but that kangaroo rats may be more abundant on grazed ranges (Reynolds, 1958; Hanley and Page, 1981; Bock et al., 1984; Fagerstone and Ramey, 1996; Hayward et al., 1997). Hayward et al. (1997) noted greater abundance of Merriam's kangaroo rats and silky pocket mice (Perognathus flavus) on grazed plots. We found approximately equal numbers of Merriam's kangaroo rats on grazed and ungrazed areas, but the species was not detected on two of the three grazed grids. We did not trap any silky pocket mice on our study plots. The differences in vegetation cover and density between our grazed and ungrazed grids may not have been as pronounced as those differences in other studies. This may be related to the stocking rate (18 ha / AU) and to the relatively recent removal of livestock from the refuge. Other studies suggest that small mammal populations may be negatively affected by heavy stocking rates (Bock et al., 1984; Kie and Loft, 1990).

Use of livestock as a management tool may be limited to regions where vegetation evolved with significant grazing pressure by large herbivores (Severson, 1990). The Great Plains is a good example of such a region; the range vegetation evolved with bison (Bos bison), whereas southwestern semi-desert grasslands evolved in the absence of large herbivores (Severson, 1990). Desert rodent populations seem to be negatively affected by grazing. Livestock may affect rodent populations directly by burrow trampling, soil compaction, or competition for food, or indirectly by altering the vegetation structure and species composition thereby influencing habitat selection (Heske and Campbell, 1991; Hayward et al., 1997).

We support Hayward et al. (1997) in that long-term experimental studies using replicated grazed and ungrazed plots are needed. However, our results suggest that rodent abundance, species richness, and perhaps diversity may be reduced on grazed ranges. This has management implications regarding the restoration of desert grassland ecosystems because rodents play a major role as primary consumers and dispersers of seeds and also comprise the major prey for several predators. Therefore, negative impacts on rodents may translate into secondary impacts on vegetation and species of greater management concern such as predators.

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Table 1--Vegetation cover on grazed and ungrazed trapping grids in southeastern Cochise Co., Arizona, 8-16 May 1996. Comparisons were made using paired t-tests with a minimum level of significance $\alpha = 0.10$.

% cover and density (# / ha)	Grazed		Ungrazed		t_2	P
	X	SE	X	SE		
% bare ground	26.67	9.96	29.67	24.58	0.31	0.79
% litter	32.33	5.36	28.00	5.57	1.86	0.20
% grass	20.67	7.54	33.33	14.71	1.61	0.25
% forb	3.00	1.00	4.00	0.58	0.87	0.48
% shrub	3.33	2.85	9.67	6.33	1.80	0.21
% half-shrub	9.33	5.21	3.33	2.03	1.31	0.32
% tree	16.67	0.33	12.33	2.60	1.86	0.20
% shrub+tree	20.00	2.52	22.00	4.93	0.67	0.57
% shrub+tree+ half-shrub	29.33	7.54	25.33	6.23	1.15	0.37
% dead woody	2.67	2.19	3.00	1.15	0.13	0.91
# woody/ha	131.00	60.38	61.67	25.86	1.96	0.19
# trees/ha	13.33	2.03	13.67	6.89	0.07	0.95
# shrubs/ha	8.67	7.17	15.67	9.53	2.29	0.15
# half-shrubs /ha	105.67	58.01	32.33	17.70	1.82	0.21
# shrubs+trees /ha	22.00	5.69	29.33	8.67	1.19	0.36

Table 2--Species and community characteristics for rodents captured on 3 grazed and 3 ungrazed grids in southeastern Cochise Co., Arizona, 8-16 May 1996. Comparisons were made using paired t-tests.

Species Relative Abundance and Community Attributes	Grazed		Ungrazed		P-value
	$\bar{X}$	SE	$\bar{X}$	SE	
Dipodomys merriami	4.08	4.08	6.12	3.49	0.19
Chaetodipus baileyi	9.75	0.60	6.35	3.29	0.46
Chaetodipus penicillatus	0.45	0.45	1.13	0.60	0.42
Baiomys taylori	0.00	0.00	0.68	0.39	0.23
Onychomys torridus	0.45	0.45	0.00	0.00	0.42
Onychomys leucogaster	0.00	0.00	0.45	0.23	0.18
Neotoma albigula	0.23	0.23	0.23	0.23	1.00
Reithrodontomys fulvescens	0.00	0.00	0.45	0.23	0.18
Reithrodontomys megalotis	0.45	0.23	0.23	0.23	0.42
Peromyscus maniculatus	0.91	0.60	2.72	0.39	0.21
Peromyscus leucopus	0.45	0.23	1.36	0.39	0.18
Peromyscus eremicus	0.68	0.68	0.23	0.23	0.42
Sigmodon ochrognathus	0.23	0.23	0.23	0.23	1.00
Population Size ($\hat{N}$)	34.39	1.14	46.72	1.08	0.06*
Species Richness	5.00	1.16	8.00	1.53	0.096*
Species Diversity (H')	1.05	0.11	1.57	0.29	0.13

* Significant difference ($P < 0.10$).

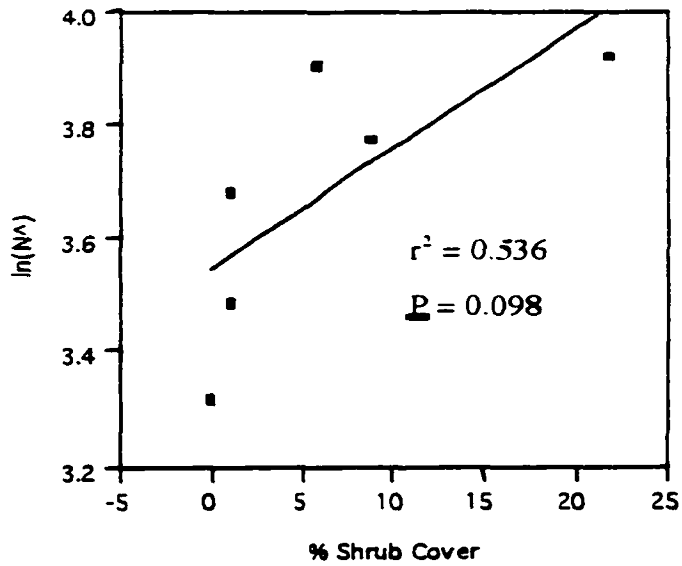

Fig. 1--The relationship between rodent community size and % cover of shrubs at 3 grazed and 3 ungrazed trapping grids in southeastern Cochise Co., Arizona 8-16 May 1996. Minimum level of significance was set at $\alpha = 0.10$.

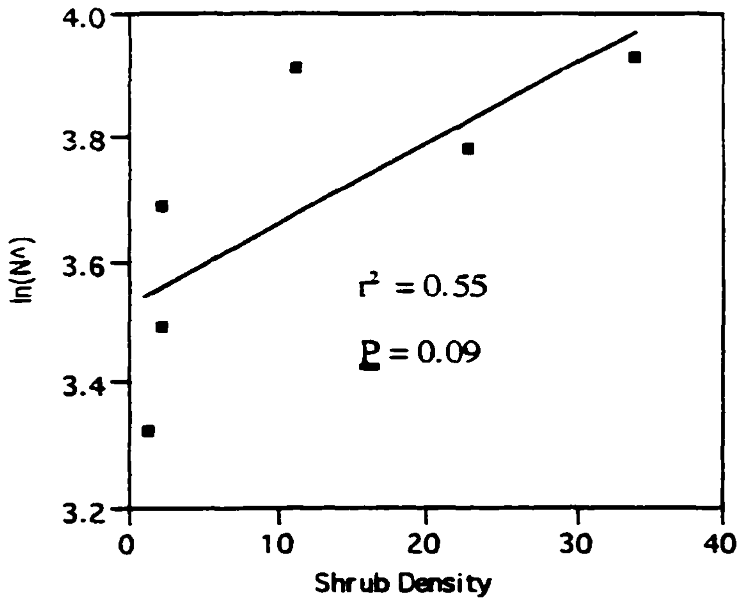

Fig. 2--The relationship between rodent community size and density of shrubs on 3 grazed and 3 ungrazed trapping grids in southeastern Cochise Co., Arizona, 8-16 May 1996. Minimum level of significance was set at $\alpha = 0.10$.

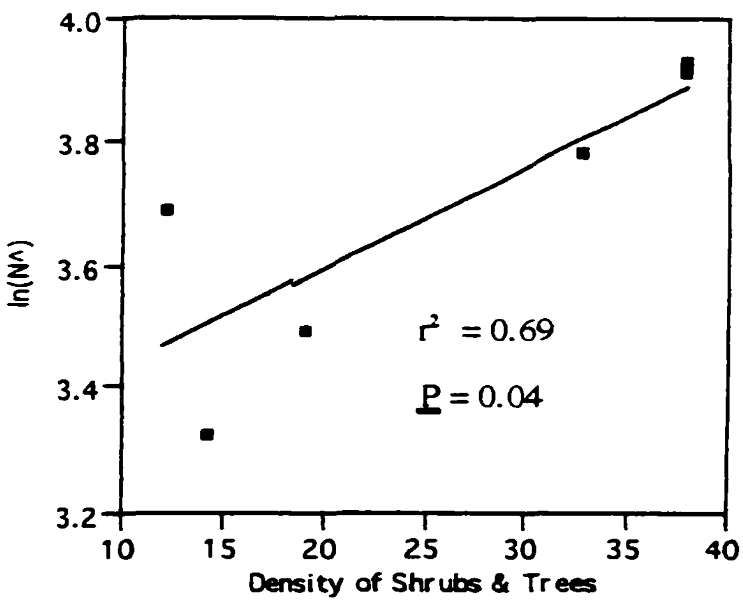

Fig. 3--The relationship between rodent community size and density of shrubs and trees on 3 grazed and 3 ungrazed trapping grids in southeastern Cochise Co., Arizona, 8-16 May 1996. Minimum level of significance was set at $\alpha = 0.10$.

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SHORT-TERM IMPACTS OF PRESCRIBED FIRE ON A RODENT COMMUNITY IN A DESERT GRASSLAND

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ABSTRACT--We compared vegetation features and rodent communities between a recently burned area and a nearby control area. Pre-treatment data were gathered 1 year prior to the fire treatment and therefore within-grid variation was expected between pre- and post-treatment samples due to temporal variation. Bare ground increased and grass and litter decreased following the burn. We did not detect differences in cover of trees and shrubs. Rodent communities did not differ in size or species richness between burned and unburned grids. There did seem to be a relative change in the proportions of bipedal and quadrupedal heteromyids on burned and control grids.

Fire was formerly a frequent natural event in semi-desert grasslands (Humphrey, 1958, 1974; Phillips, 1962; Bock et al., 1976) and its frequency prevented the invasion of woody plants (Thornber, 1910; Wooton, 1916; Leopold, 1924; Humphrey, 1958). Even occasional fires may prevent the establishment of woody climax species maintaining a grass subclimax (Humphrey, 1974). The occurrence and intensity of fires was lessened directly by fire control and indirectly by livestock overgrazing that reduced the fuel load of rangelands (Humphrey, 1963; Chew and Chew, 1965; Cable, 1973; Martin, 1975; Bock and Bock, 1992). Reduced competition from grasses due to overgrazing and the absence of fire, enabled woody trees, shrubs, and cacti to invade or increase within grasslands and often to out-compete native grasses for soil moisture (Cable, 1973; Wright, 1974). Increased abundance of woody plants and cacti meant less suitable rangeland for livestock and changes in wildlife habitat.

Mesquite and other woody shrubs continue to persist, dominate, and increase in abundance (Gibbens et al., 1992; Holechek et al., 1994) to the detriment of native grasses, grassland ecosystems, and rangeland quality (Reynolds and Bohning, 1956; Wright, 1973). Without control efforts, a reduction in forage for livestock and loss of habitat for grassland-dependent species, such as pronghorn (Antilocapra americana) (Gibbens et al. 1992) and less conspicuous species such as northern pygmy mice (Baiomys taylori) and cotton rats (Sigmodon spp.), may result. Livestock ranchers and government agencies are attempting to restore grassland conditions via improved grazing systems, prescribed burning, and chemical and mechanical brush control (Martin, 1975). Their primary goal is to re-establish native grasses and reintroduce fire to the system on a regular schedule once the fuel load of rangelands is sufficient to carry a burn.

Although many have examined the impacts of fire on small mammal communities (Baker, 1940; Lawrence, 1966; Kaufman et al., 1988; Simons, 1991; Haim and Izhaki, 1994), relatively little attention has been given to fire and desert-grassland rodents (Bock et al., 1976; Christian, 1977; Bock and Bock, 1978). Christian (1977) worked within a grassland in the Namib Desert in Africa. In the southwestern United States, efforts have focused on fire's effects on small mammals in an oak-savannah community (Bock et al., 1976; Bock and Bock, 1978). We had the opportunity to collect pre-burn and post-burn samples in a shrub-grassland while working on a brush control project.

Range management techniques, such as prescribed burning, alter the vegetation composition and structure (Scifres, 1980) and may alter the suitability of habitat for some small mammal species (Kaufman and Fleharty, 1974). Treatments such as prescribed burning, herbicide applications, and mechanical brush control reduce cover and food availability for rodents (Vallentine, 1989). Relationships of small mammal species to their habitats and vegetation physiognomy are critical to making informed decisions about future habitat modifications, wildlife management, and rangeland production (Fagerstone and Ramey, 1996).

Our objectives were to examine pre- and post-burn rodent communities and determine if prescribed burning had a measurable impact. We predicted that rodent abundance and species richness would differ between burned and unburned ranges due to treatment-induced differences in vegetation cover.

STUDY AREA--The study site was located in Cochise County, Arizona east of Douglas at San Bernardino National Wildlife Refuge and the neighboring Malpai Ranch. According to Mitchell (1976), the climate is controlled by the summer monsoons occurring from July-September. Most precipitation occurs during summer with occasional rains in winter. Annual precipitation was approximately 30 cm (Wendy Glenn, Malpai Borderlands Group, pers. commun.), typical of the region which ranges from 20.3 - 40.6 cm (Smith and Schmutz, 1975; Brown, 1982; NOAA ,1996). Annual mean temperatures are moderate, ranging from 12-20 ° C, but summers are hot (Bourgeron et al. 1995). The elevation is approximately 1,150 m. The Malpai Ranch is open to cattle grazing at approximately 18 ha per animal unit and both the ranch and the refuge are dominated by tobosa (Hilaria mutica), white-thorn acacia (Acacia constricta), mesquite, snakeweed (Gutierrezia sarothrae), and tarbush (Flourensia cernua).

METHODS AND ANALYSES--We conducted all sampling within plots that were set up for a comparison of grazed an ungrazed areas. The plots were not established randomly because of the limited area available for pairing grazed and ungrazed trapping grids along the refuge-ranch boundary. The area that was burned incorporated only one of three grids within the refuge. Therefore, we conducted all vegetation and rodent sampling within two 90 m^2 trapping grids, one on the refuge and one on the ranch. Within each grid we set one 100 m line transect diagonally from a randomly selected grid corner. We used the point-intercept method to estimate percent cover (i.e., the total ground area covered by various plant species, bare soil, rocks, or litter). We recorded the type of substrate intercepted by a vertical line at 100 points (every 1.0 m) along each transect (Bock and Bock, 1978). When vegetation was intercepted, we recorded the species. We recorded multiple interceptions when the canopies of ≥ two plants directly overlap at a given point.

We established one trapping grid within the area of the refuge scheduled for burning and one control grid approximately 200 m away (Hall and Willig, 1994) on the Malpai Ranch. Each grid had a 7 x 7 trap (49 traps) configuration with 15 m intervals between consecutive trap stations (Brown and Zeng, 1989; Bowers and Brown, 1992; Heske et al., 1994). We trapped small mammals on each grid simultaneously for three consecutive nights. We set traps baited with mixed birdseed ≤ 3 hours before sunset and checked traps the following morning ≤ 3 hours after sunrise.

We did not use polyester stuffing fiber for bedding because of problems identified in preliminary sampling. Kangaroo rats (Dipodomys spp.) frequently became tangled in the fiber cutting off the circulation in their hind limbs. However, we partially buried traps (leaving the doors exposed) as a precaution against overnight cold and morning heat. We were limited to trapping near full moon phases because of our trapping schedule for a separate brush-control study, but we are aware that moonlight may reduce nocturnal rodent activity (Kaufman and Kaufman, 1982; Kotler, 1984; Price et al., 1984; Bowers, 1988; Longland and Price, 1991).

We recorded the species, age class, sex, reproductive condition, body mass, and trap station for all captured individuals. We marked all captures using permanent ink markers and released all animals at capture locations. Permanent ink markers have been used for temporarily marking small mammals to prevent recounting individuals that may be recaptured during a 3-day trapping session (Hall and Willig, 1994; Petryszyn and Russ, 1996). We used different colored inks for marking animals trapped in each grid to determine if rodents were moving between treatment and control grids or among paired grids. This procedure assured the independence of sampling units. We used numbered monel ear tags (style 1005-1, National Band & Tag Co., Newport, KY) for all post-burn captures.

We conducted pre-treatment sampling on both grids during 11-13 May 1996. An area on the refuge of approximately 243 ha that included our treatment grid was burned on 30 May 1997. We returned and sampled both grids during 17-19 June 1997. Virtually all of the grass cover had been eliminated by the burn. Most of the acacia and mesquite was top-killed by the burn but sprouting occurred within a few months of the burn.

We generated estimates of rodent community (i.e., all rodent species combined) size ($\hat{N}$) for each trapping grid with removal methods using program CAPTURE (Otis et al., 1978). Neither actual removal from the area nor kill-trapping is necessary to apply removal analysis methods to small mammal trapping data (White et al., 1982:101). All captured individuals were effectively removed from the unmarked population by marking them (Seber, 1973:323; White et al., 1982:67). We selected model M_{bh}-Pollock using the variable probability removal estimator (Pollock and Otto, 1983) for population estimation because it is robust to violations of the assumption of declining recapture rates during successive trapping occasions. Because we used uniquely marked ear tags in 1997, we were also able to generate population estimates for those data with model M_h using the jackknife estimator (Otis et al., 1978, White et al., 1982:63-66). Capture rates for particular species were too low to generate species population estimates, so we calculated the relative abundance (no. individuals captured / 100 trap nights) for each species within each grid. We measured species richness (i.e., no. species) based on the number of individuals captured for each species at each study site.

RESULTS--There were small between-year differences in vegetation cover on the control grid that are most likely attributed to setting a slightly different line transect. There were also pre-treatment differences between the control and pre-burn grids (Table 1). In 1996 there was twice as much bare ground on the control grid (30%) compared to the pre-burn grid (14%). There was 46% grass cover on the pre-burn grid compared to 31% on the control grid. Tree cover was similar between grids with 17% on the control grid and 12% on the pre-burn grid. After the burn, bare ground increased from 14 to 51%, grass cover was reduced from 46 to 2%, and litter was reduced from 32 to 12%.

The pre-burn population estimates indicate a smaller rodent community on the control grid ($\hat{N} = 28 \pm 5.48$ [SE]) compared to the treatment grid ($\hat{N} = 40 \pm 6.0$) (Table 2). Removal population estimates were greater on both grids during the 1997 sampling period compared to 1996 and there was less difference between the treatment and control estimates. The 1997 rodent population estimate for the control grid ($\hat{N} = 58 \pm 8.83$) was more than twice the pre-burn estimate from 1996. The population estimate for the burned area ($\hat{N} = 50 \pm 8.12$) was 25% greater than the pre-burn estimate from 1996 (Table 2). The jackknife population estimates were similar between burned ($\hat{N} = 43 \pm 5.77$) and unburned ($\hat{N} = 46 \pm 5.74$) grids (Table 2). We caught 5 species on the burned grid and 4 species on the control grid. Based on species relative abundance, Merriam's kangaroo rats (D. merriami) were more abundant on the burned grid and deer mice (Peromyscus maniculatus) were more abundant on the control grid. Bailey's pocket mouse (Chaetodipus baileyi) was the most abundant species on each grid. There were 32% more Bailey's pocket mice trapped on the unburned grid compared to the burned grid.

DISCUSSION--Bare ground increased and grass cover and litter decreased after the burn as expected. We did not identify a reduction in tree or shrub cover, but foliage was reduced. The pre-burn samples suggest a slightly larger rodent community may have inhabited the treatment grid on the refuge. This may be related to the absence of livestock on the refuge and the associated differences in cover. Rodent abundance was greater in 1997 compared to 1996 on burned and unburned grids despite the fact that both sampling periods occurred near full moon phases. This between-year difference may simply represent year-to-year fluctuations in population size. Post-burn removal estimates were slightly greater on the unburned grid. This slight difference was also supported by the jackknife estimates. The jackknife estimates are probably more realistic because they account for recaptures not included in removal estimation. According to others, overall small mammal abundance may be reduced for ≥ 1 year after burning, but these reductions are not likely to persist beyond the second year (Cook, 1959; Bock and Bock, 1978; Groves and Steenhof, 1988). Our study was a first step in examining the impacts of fire on a desert rodent community and inferences may be limited because there was no replication. However, if our results are applicable to other areas, then they may suggest that the immediate impacts of fire on rodent communities does not include reduced abundance.

There did not seem to be a difference in species richness between years nor between burned and unburned grids, though the species composition seemed to differ between treatments. There were more Merriam's kangaroo rats on the burned area. This may reflect their association with more open areas. Wondolleck (1978) reports that Merriam's kangaroo rats forage in areas without basal cover. Bailey's pocket mice were slightly more abundant on the unburned grid supporting their documented preference for areas with greater cover (Reynolds and Haskell, 1949; Rosenzweig and Winakur, 1969). Price (1978) and Rosenzweig (1973) demonstrated reductions in pocket mice and increases in kangaroo rats when shrub cover was reduced. We caught 1 pygmy mouse in 1997 on the unburned grid where there was substantial grass cover. We had trapped 1 pygmy mouse on the treatment grid prior to the burn, however their absence from post-burn sampling was not surprising given their association with dense grass cover (Powell, 1968; Turner and Grant, 1987).

Small mammal species may serve as indicators of rangeland quality. Species typical of xeric environments, such as Merriam's kangaroo rat, may be useful indicators of desertification in semi-desert grasslands (Bock et al., 1984). This effort may clarify some of the impacts of prescribed fire on rodents and their habitats within the semi-desert grassland ecosystem. Although overall rodent abundance did not seem greatly affected by the burn, the species composition was altered. Bailey's pocket mice were still the predominant species, but their abundance on the burned area was reduced relative to that of the control area. Additionally, Merriam's kangaroo rats increased in abundance after the burn. If the goal is to restore grass-shrub ecosystems to grassland condition, some consideration must be given to the other organisms that currently or previously inhabited these ranges. Small mammals have a major role at the consumer trophic level, as seed dispersers, and as the major prey base for many other species. Rangeland alterations such as prescribed burning are likely to have effects on wildlife species such as rodents, and these effects may lead to unanticipated changes in plant and animal populations of greater management concern.

Our study was funded by the US Forest Service Rocky Mountain Experiment Station and the International Arid Lands Consortium. Technical support was provided by C. B. Edminster, G. J. Gottfried, and L. F. DeBano. Field assistance was provided by J. A. Bittner. Use and handling of rodents complied with the American Society of Mammalogists (1987) and the Animal Welfare Act enforced by the Institutional Animal Care and Use Committee, The University of Arizona, Tucson (Protocol # 96-090).

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Table 1. Pre- and post-treatment values for percent cover of vegetation on a burned plot and a control plot on San Bernardino National Wildlife Refuge and the Malpai Ranch, Cochise Co., Arizona. Pre-treatment sampling was conducted 4 May 1996. The refuge plot was burned 29 May 1997. Post-burn sampling was conducted 17 June 1997.

Cover type	Pre-burn		Post-burn	
	Treatment	Control	Treatment	Control
bare ground (%)	14	30	51	36
litter (%)	32	40	12	21
grass (%)	46	31	2	37
forb (%)	12	5	0	0
half-shrub (%)	0	0	2	0
shrub (%)	1	0	0	0
tree (%)	12	17	19	12

Table 2--Relative abundance and population estimates of rodent communities on a burned and an unburned grid in southeastern Cochise Co., Arizona. Pre-burn sampling occurred during 11-13 May 1996. The treatment grid burned on 29 May 1997 and post-burn sampling occurred during 17-19 June 1997.

Species and Community Characteristics	Pre-burn		Post-burn	
	Treatment	Control	Treatment	Control
Chaetodipus baileyi	12.93	8.84	12.24	16.33
Dipodomys merriami	1.36	0.00	4.76	0.68
Peromyscus maniculatus	2.04	2.04	0.68	4.08
Peromyscus leucopus	2.04	0.68	0.68	0.00
Reithrodontomys megalotis	0.68	0.68	0.00	0.00
Baiomys taylori	0.68	0.00	0.00	0.68
Sigmodon ochrognathus	0.00	0.68	0.00	0.00
Onychomys torridus	0.00	0.00	0.68	0.00
All Species	19.73	12.93	19.05	21.77
Removal Estimates (95% CI)	40 (33-58)	28 (22-45)	50 (39-72)	58 (46-81)
Jackknife Estimates (95% CI)			43 (36-59)	46 (39-62)
Species Richness	6	5	5	4

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USE OF BURIED VERSUS NON-BURIED TRAPS IN DESERT RODENT SAMPLING

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ABSTRACT.--Burying traps may be a cost-effective means of providing captured rodents some insulation from desert temperature extremes. However soil entering traps may effect their sensitivity and under-represent species of lower body mass. We attempted to identify the effectiveness of burying traps compared to not burying traps with respect to the capture success in each trap position. Captures were greater ($\underline{P} = 0.087$) for non-buried traps compared to buried traps. We did not observe a significant difference in the number of animals captured weighing ≤ 20 g nor ≤ 15 g between buried and non-buried traps. There did seem to be a trend toward fewer small species captured in buried traps.

Procedures for live-trapping small mammals in extreme climates should incorporate strategies for reducing trap fatalities if reliable estimates of abundance, survival, reproductive potential, and microhabitat use and selection are to be obtained. Additionally, trapping procedures should account for behavioral adaptations to existence in extreme conditions. In desert regions, the majority of small mammal species are nocturnal and are typically sampled at night with traps closed during daylight hours (Bowers and Brown, 1992; Brown and Zeng, 1989; Price, 1978; Rosenzweig and Winakur, 1969; Swann et al., 1997). Other desert rodent studies have involved daytime and nighttime trap checks (Andersen, 1994; Kerley, 1992; Simons, 1991). Daily temperature fluctuations may be large in desert environments and captured animals may be vulnerable to hypothermia overnight and heat stress during daylight prior to release. Simons (1991) checked traps at night only during rainstorms or cold periods to minimize trap fatalities. Others have used stuffing fiber or wool as bedding and insulation (Kerley 1992). Cotton absorbs moisture and will not provide insulation when wet. We found polyester fiber unsuitable for trap insulation because kangaroo rats frequently became entangled and had the circulation cut off in their hind limbs. Some researchers have used 2 nighttime trap checks to reduce overnight fatalities (Price and Waser, 1985; Schroder and Rosenzweig, 1975; Wondolleck, 1978). Brown (1989) checked traps 2 or 3 times during daytime and Andersen (1994) shaded traps with Sherman aluminum trap tents to prevent daytime captures from overheating. Nighttime trap checks and shading devices add labor and costs to trapping studies.

Only one of these studies (Swann et al., 1997) was known to use soil and litter to shade traps from sun exposure and provide some insulation from overnight cold, though the burying process was not documented (A. J. Kuenzi and M. L. Morrison, pers. comm.). Using soil to insulate traps eliminates the added expense of purchasing manufactured trap tents or materials to construct shades and may provide an effective means of protecting captured animals from overnight cold and morning heat prior to release. We used partially buried traps in a brush control study conducted in southeastern Arizona. We noticed that soil entered traps through openings and may have altered trap sensitivity thereby under-representing small-bodied species. To our knowledge, the effectiveness of buried traps has not been tested. In the present study, we examined the effectiveness of partially buried traps compared to non-buried traps. We predicted that fewer species of low body mass (≤ 20 g) would be captured in buried traps due to reduced sensitivity.

METHODS

We trapped 4 locations in the San Bernardino Valley of southeastern Cochise Co., Arizona. The sites varied in vegetation features and soil types and therefore varied in rodent species composition and abundance. All sites were within historic desert grassland or desert scrub communities, and varied from mesquite (Prosopis velutina)-tobosa (Hilaria mutica) grassland to creosotebush (Larrea tridentata)- acacia (Acacia constricta) associations.

We used a 7 x 7 trap configuration with 15 m spacing between consecutive trap stations. We placed two large folding Sherman live traps at each station (i.e., 98 traps per grid). One trap was partially buried with the door left exposed and the other trap was placed on the ground within 1 m of the buried trap. We baited traps with mixed bird seed because the majority of rodents in the area are predominately granivorous. We set traps within 2 hours of sunset and checked all traps within 2 hours of sunrise. For each animal captured we recorded the species, age class, sex, reproductive condition, body mass, trap station, and trap position (i.e., buried, non-buried). Each grid was trapped for three consecutive nights (i.e., 294 trap nights per grid).

We compared number of captures between buried and non-buried traps with a paired t-test. We also compared the number of animals captured weighing ≤ 20 g and ≤ 15 g in buried and non-buried traps with paired t-tests. Originally, we chose 20 g as an upper limit for small-bodied captures because it excludes the larger species (Dipodomys spp., Chaetodipus baileyi, Onychomys spp., Sigmodon spp., Neotoma albigula) and incorporates juveniles and smaller species (Peromyscus spp., Chaetodipus penicillatus, Baiomys taylori, Reithrodontomys spp). However, because there was a substantial number of captures weighing ≤ 15 g (including 3 species always < 15 g), we decided to also use that mass as an upper limit for comparisons. We set $\alpha = 0.10$ for all tests.

RESULTS

We made 217 captures (not individuals) in 1,176 trap nights, 87 in buried traps and 130 in non-buried traps. The mean number of captures per site was significantly greater for non-buried traps ($\bar{x} = 32.50 \pm 5.55$) compared to buried traps ($\bar{x} = 21.75 \pm 1.44$) ($t = 2.51$, 3 d.f., $P = 0.087$). We did not identify a significant difference ($P = 0.20$) in the number of animals weighing ≤ 20 g between buried traps ($\bar{x} = 6.00 \pm 1.41$) and non-buried traps ($\bar{x} = 10.75 \pm 3.75$). We also did not identify a significant difference ($P = 0.26$) in the number of animals captured weighing ≤ 15 g between buried ($\bar{x} = 3.50 \pm 1.85$) and non-buried traps ($\bar{x} = 6.75 \pm 4.17$).

DISCUSSION

Overall capture success was significantly greater for non-buried traps compared to buried traps. This difference may be attributed to reduced trap sensitivity or alternatively to soil disturbance at the trap stations. Perhaps the disturbed soil deters rodent entrance into buried traps by altering the microhabitat (e.g., disruption of scents, topography) or because it is perceived as predator digging activity. Although we did not identify significant differences in the number of captures ≤ 20 g nor captures ≤ 15 g between buried and non-buried traps, there did appear to be a trend toward greater capture rates for small-bodied rodents in non-buried traps. If this trend is realistic, the lower capture rates for small-bodied rodents in buried traps may confirm our suspicions about reduced sensitivity due to the burying process.

Burying traps may be a cost-effective means of reducing trap fatalities associated with temperature fluctuations in desert environments. However, properly burying traps adds labor when establishing grids or transects, setting traps, and processing captured animals. Additionally, we believe that folding live-traps may lose some sensitivity when partially buried. This loss of sensitivity may under-represent small-bodied species within the community and should be accounted for if accurate estimates are to be achieved.

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CONCLUSION

I examined the potential impacts of mechanical shrub control, livestock grazing, and prescribed fire on rodent communities within the desert grasslands. Rodent communities did not differ in richness or abundance between mechanically treated and control areas. Merriam's and Ord's kangaroo rats exhibited contrasting relationships with grass cover that may explain their sympatry. Grazing did seem to have an impact on rodent communities. Grazed areas had lower species richness and abundance of rodents compared to ungrazed areas. The immediate impacts of burning did not include a detectable difference in rodent abundance or species richness, but kangaroo rats may have favored the burned areas because of reductions in basal cover.

Estimation of rodent abundance and identification of species composition should be incorporated into rangeland management and grassland restoration efforts. Because rodents are the primary consumers and dispersers of seeds they can have important impacts on grass germination and seed dispersal. Kangaroo rats in particular may have marked effects on desert grassland communities through their scatterhoarding and seed caching and their reliance on grass seeds for the majority of their diet. Thus, rodents can have an impact on the community that may translate into further impacts on other management concerns such as range quality for livestock and game species. Further, negative impacts on rodent numbers due to vegetation manipulation efforts may lead to unanticipated declines in predators reliant on rodent prey.

Finally, sampling live animals in extreme climates must incorporate strategies or methodologies for reducing animal fatalities while still providing reliable information about abundance, survival, habitat use, and resource selection. In desert regions, daily temperature fluctuations can be large and precautions must be taken to account for this when sampling rodent communities. I partially buried traps to provide insulation from overnight cold and (pre-release) daytime heat, but was concerned about reduced sensitivity for traps that were buried. I did capture fewer animals in buried traps compared to non-buried traps. There may have been a trend toward fewer small-bodied species captured in buried traps due to reduced sensitivity, but a larger sample size was needed. These limited results suggest that I may have under-represented the abundance of rodents by burying traps, but that bias was consistent between treated and control areas and among all trapping grids. Therefore, I am confident that the burying process did not affect the results of these investigations and that I minimized animal fatalities that would have otherwise occurred.