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Effects of seasonal grazing, drought, fire, and carbon enrichment on soil microarthropods in a desert grassland

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ABSTRACT

This study was designed to test hypotheses about the combined effects of short-term, seasonal grazing with seasonal drought, fire, and carbon enrichment on soil microarthropod communities in a Chihuahuan Desert grassland. The study was conducted in eighteen 0.5 ha plots following three consecutive years of treatment: six plots intensively grazed in summer, six in winter, and six not grazed. There was no difference in perennial grass cover on the summer-grazed and winter-grazed plots. Intensive seasonal grazing had no effect on the abundance and community composition of soil microarthropods. Within each plot there were six subplots: summer rain-out, winter rain-out, burned, glucose amendment, rain-out control and burn-glucose control. Fire and carbon enrichment had no significant effect on soil microarthropod abundance or community composition. The average number of microarthropods ranged from $8915 \pm 1422 \text{ m}^{-2}$ in the ungrazed, unburned plots to $7175 \pm 1232 \text{ m}^{-2}$ in the winter-grazed, unburned plots. Microarthropod densities in the glucose-amended plots were $8917 \pm 4902 \text{ m}^{-2}$ in the winter-grazed plots and $10,731 \pm 863 \text{ m}^{-2}$ in the glucose-amended, summer-grazed subplots.

The prostigmatid mite, *Tydeus* sp., was the most abundant microarthropod taxon in all treatment plots.

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1. Introduction

The basins and piedmont slopes in the Chihuahuan Desert in North America that were classified as grasslands in the mid-1800s are now predominately shrublands. By the mid-1900s, most of the grasslands had been replaced by desert shrub communities (Buffington and Herbel, 1965; Gibbens et al., 2005). Formerly shrub-free grasslands now exist as shrub-grass mosaics, coppice dunes, or shrubland with little or no grass understory. These changes in vegetation are thought to have been the result of periodic drought and overgrazing by domestic livestock. Reduction in grass cover allowed water and wind erosion of soil and the dispersal of shrubs into the grassland. Now most remnant patches of grassland have varying densities of shrubs.

It has been suggested that desert grasslands remained shrub-free because of periodic fires (Bahre, 1991; Humphrey, 1958) but recent studies in the Chihuahuan Desert reported that fire had little effect on invasive shrubs and cover of some perennial grasses

(Drewa and Havstad, 2001; Killgore et al., 2009). Prescribed fire had no effect on rodents and termites but did reduce the numbers of burrowing spiders (*Geolycosa* spp.) (Killgore et al., 2009). Since fire had little effect on vegetation in Chihuahuan Desert grasslands, we hypothesized that fire would not affect the abundance and diversity of soil microarthropods.

Livestock grazing is the primary land use in arid lands and is known to have numerous effects on the structure and function of arid and semi-arid ecosystems. Livestock grazing changes soil characteristics such as bulk density, microtopography, and vegetation cover (Kay et al., 1999; Nash et al., 2004). Soil microarthropod abundance and diversity have been reported to be reduced in areas heavily impacted by livestock (Kay et al., 1999; Kinnear and Tongway, 2004). We hypothesized that intense livestock grazing would reduce the abundance and diversity of microarthropods and that seasonal grazing would exacerbate the effects of seasonal drought and fire.

Soil microarthropods are important components of the soil biota in arid ecosystems (Neher et al., 2009). In arid ecosystems, soil mites are the only component of the soil fauna that remain active in very dry soils (Whitford, 1989). Soil microarthropods affect rates of litter decomposition (Santos et al., 1981; Santos and Whitford, 1981) and are most important in the mineralization of nutrients,

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especially nitrogen that is immobilized in microbial biomass (Parker et al., 1984). The species composition and abundance of soil microarthropods in arid environments vary with vegetation type, topographic position, weather patterns, especially rainfall, and with land use, especially livestock grazing (Cepeda and Whitford, 1989; Cepeda-Pizarro and Whitford, 1989b; Kay et al., 1999). Experiments using simulated rainfall (irrigation) and simulated drought (rain-out shelters) have been used to examine the effects of weather variables on microarthropod communities in shrub-dominated ecosystems (MacKay et al., 1986; Whitford et al., 1988; Whitford and Sobhy, 1999). Evaluation of the effects of soil properties, vegetation, and run-on, run-off processes on topographic sequences on microarthropod communities has been studied by sampling at various topographic positions on a watershed or disturbance gradients (Cepeda and Whitford, 1989; Cepeda-Pizarro and Whitford, 1989a, b; Kay et al., 1999).

The abundance of micro and mesofauna in arid land soils has been found to be a function of soil organic carbon (Steinberger et al., 1984; Wallwork et al., 1985). Soil organic-carbon levels are very low in Chihuahuan Desert soils (Nash and Whitford, 1995). In desert grasslands, the primary sources of soil organic carbon are turnover of roots of the tussock grasses and dead roots of annual forbs and grasses. A readily metabolized carbon source added to soil will produce a rapid increase in microbial biomass (Coleman et al., 2004). Increased microbial biomass should result in increased abundance of soil food-web biota. We hypothesized that adding glucose to desert grassland soil would result in higher abundance and diversity of soil microarthropods and would mask any effects of grazing on soil biota. Seasonal grazing may also interact with rainfall to affect soil organic carbon and soil organisms. Here we report on an experimental study designed to test hypotheses on the effects of intensive seasonal grazing modifying the effects of drought, fire, and carbon enrichment on the abundance and taxonomic composition of the soil microarthropod fauna.

2. Methods

The experimental site (latitude 32°35', longitude 106°50') is part of the USDA Jornada Experimental Range, located approximately 40 km NNE of Las Cruces, New Mexico. The experiment consisted of 18 one-half hectare plots located in an ecotonal area between black grama (*Bouteloua eriopoda*) grassland with sparse shrub cover and a mesquite (*Prosopis glandulosa*) coppice dune area. Study plots were set up in two rows of nine plots with a combination of treatments randomly assigned to each plot. In nine plots, all of the *P. glandulosa* shrubs were manually removed and the stumps treated with herbicide. Six plots were assigned to summer grazing, six plots to winter grazing, and six plots to no grazing. Grazed plots were stocked with 16–30 yearling steers. Before cattle were placed in the experimental paddocks, the available forage was estimated using line-point transects. Stocking rates were adjusted to remove approximately 50% of the available forage in approximately 36–48 h. Plots were grazed beginning in January 1995 (winter-grazed) and were summer-grazed in August for three consecutive years.

Six subplots were located in shrub-free patches within each 0.5 hectare plot: summer rain-out, winter rain-out, no rain-out, burn, glucose amendment and untreated control. The subplots were 2 m × 1 m and were trenched to a depth of 1 m or to the cemented calcrete layer. The trenches were lined with a double layer of landscape plastic sheeting in order to keep the roots of the plants within the subplots from accessing water and nutrients from the soils outside the plots. Experimental drought was imposed by rain-out shelters constructed with a steel frame A roof. The low edges of

the roofs were 2 m above the soil. The roofs extended 1.5 m beyond the perimeter of the subplots. One subplot assigned to summer drought was covered with the rain-out shelter from mid-May through mid-September. Winter drought plots were covered with the rain-out shelters from mid-September to mid-May. The rain-out treatments were terminated three months prior to the soil microarthropod sampling.

Glucose was applied to the soil carbon-enrichment plots at a rate of 100 g m⁻² (Coleman et al., 2004). The glucose amendments were applied three years before the plots were sampled for microarthropods. The prescribed burns were applied four years prior to the initiation of this study. The perimeters of subplots designated for prescribed burns were protected by a 0.5 m high metal sheeting that prevented the fire from spreading. These plots were burned using drip torches to get a complete burn of the vegetation within the subplots.

All subplots were sampled in August one year after the last grazing by livestock.

Perennial grass basal cover was estimated by a 1 m² frame subdivided into 0.1 m² units. Perennial grass basal cover of experimental plots was estimated by centering the frame in each plot. Perennial grass cover of the grazed and ungrazed plots was estimated in 20, 1 m² quadrats stratified to provide equal coverage of each plot. Microarthropods were sampled by removing one 10-cm diameter core to a depth of 20 cm from each of the subplots. Soil cores were placed in plastic bags and transported to the laboratory in insulated coolers. Microarthropods were extracted in modified Tullgren funnels into water. Extracted microarthropods were counted under a compound microscope and identified to family and/or genus where possible using keys in Krantz (1978). All data were subjected to statistical analysis of variance using the SAS model (GLM). Duncan's multiple range tests were used to evaluate differences between separate means (SAS, 1988; Sokal and Rohlf, 1969).

3. Results

There were no significant differences in perennial grass cover of winter-grazed and summer-grazed plots but perennial grass cover was significantly higher in the ungrazed plots ($F_{2, 15} = 83.5$, $p < 0.001$) (Table 1). There were no differences in abundance of microarthropods among grazing treatments in the soils of the winter-grazed, summer-grazed, and ungrazed plots ($F_{2, 33} = 0.62$, $p > 0.94$) (Fig. 1). There was no perennial grass cover in 16 of the 18 summer rain-out plots and the average perennial grass cover was 0.1% in the two plots with perennial grass. There were no significant differences in perennial grass cover in the winter rain-out plots (Table 1). There were no differences in microarthropod abundance in soils of winter rain-out plots and no rain-out plots. Soil microarthropod abundance was significantly lower in summer rain-out plots (573 ± 244 m⁻², mean ± SD) than in winter rain-out (4158 ± 710 m⁻²) or no rain-out plots (2659 ± 1175 m⁻²).

There were significantly fewer microarthropods in the summer-grazed burned plots (6013 ± 4102 m⁻²) than in the winter-grazed

Table 1
Mean perennial grass cover (percent ± standard deviation) for the treatment plots.

Treatment	Summer	Winter	Ungrazed
Grazing	3.3 ± 0.9	3.9 ± 1.8	14.7 ± 2.2
Summer rain-out	0.0	0.0	0.02 ± 0.04
Winter rain-out	3.4 ± 0.9	3.8 ± 1.7	4.0 ± 1.3
Burn	2.1 ± 1.2	3.9 ± 1.2	4.6 ± 2.1
Burn/glucose control	3.6 ± 1.1	3.9 ± 1.7	13.5 ± 3.1
Glucose amendment	3.9 ± 1.8	4.5 ± 3.0	5.1 ± 2.1

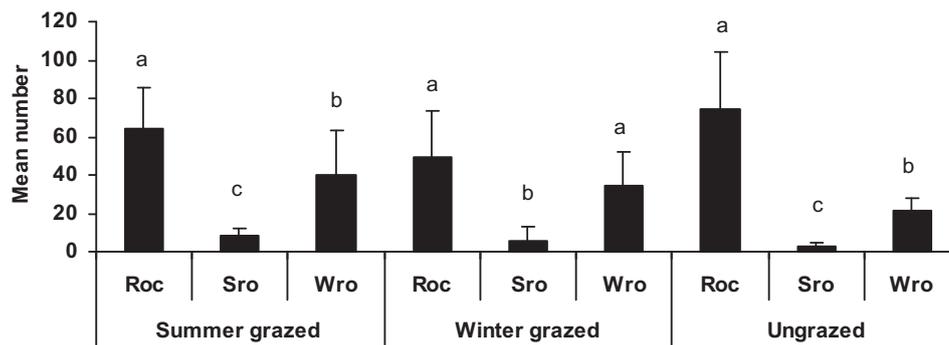


Fig. 1. The average number of microarthropods per core \pm standard deviation in the drought treatments. Roc = rain-out control, Sro = summer rain-out, Wro = winter rain-out.

burned plots ($6943 \pm 4739 \text{ m}^{-2}$) and ungrazed burned plots ($3988 \pm 1427 \text{ m}^{-2}$) (Fig. 2). There were no differences in microarthropod abundances between burned and unburned subplots among grazing treatments or among glucose-amended plots among grazing treatments ($F_{5, 30} = 1.7, p > 0.2$). The mean number of microarthropods ranged from $5613 \pm 1778 \text{ m}^{-2}$ in the ungrazed burned plots to $6057 \pm 2324 \text{ m}^{-2}$ in the winter-grazed, unburned subplots. The average number of microarthropods ranged from $8915 \pm 1422 \text{ m}^{-2}$ in the ungrazed, unburned plots to $7175 \pm 1232 \text{ m}^{-2}$ in the winter-grazed, unburned plots. There were no differences in microarthropod densities in the glucose-amended and unamended subplots among grazing treatments. The average densities ranged from $7155 \pm 1231 \text{ m}^{-2}$ in the unamended, winter-grazed subplots to $10,731 \pm 863 \text{ m}^{-2}$ in the glucose-amended, summer-grazed subplots.

The abundance of the dominant mite species, *Tydeus* sp., was significantly lower in the summer rain-out plots than in the winter rain-out plots and the rain-out control plots ($F_{8, 46} = 4.6, p < 0.001$) (Table 2). *Tydeus* sp. was the only mite taxon that occurred in all subplots. The other taxa that occurred in more than 50% of the sample plots occurred at low densities, and the absence of a taxon from some subplots resulted in large variances associated with these estimates of average density, thereby rendering statistical analysis meaningless. The abundance of these taxa was generally lower in the summer rain-out plots than in the winter rain-out and rain-out control plots (Table 2). There were fewer species of other taxa and fewer microarthropods of other taxa in summer rain-out subplots except in the summer-grazed plots. The number of species of other taxa and the numbers of microarthropods of other taxa were similar in all of the other rain-out treatments (Table 2).

There were no differences in abundance of the three most common mite taxa among treatments in the burned and glucose-amended plots, nor were there any differences in abundance of the individual taxa in the burned and glucose-amended plots ($F_{2, 15} = 0.74, p > 0.5$) (Table 3).

The remaining microarthropod taxa included in the samples occurred in very few subplots and in very low numbers. Most of these microarthropods were omnivorous prostigmatid mites (Table 4).

4. Discussion

While seasonal grazing reduced perennial grass cover, there appeared to be sufficient grass tussocks in the grazed plots to provide substrates for the rhizosphere microbial community necessary to support the microarthropod community. While grazing had an effect on microtopography and vegetation (Nash et al., 2004), the short duration of the grazing did not cause sufficient soil compaction to have an effect on the abundance and/or taxonomic composition of the soil microarthropod community. Long-term exposure to concentrated use by livestock has been reported to decrease the abundance and diversity of soil microarthropods (Kay et al., 1999; Kinnear and Tongway, 2004).

The reduction in microarthropod abundance and diversity in the summer rain-out plots was the direct effect of lack of water during the growing season of the C4 grasses. All of the grasses in the Chihuahuan Desert grasslands are C4 – warm-season grasses that depend on summer monsoons to grow and set seed. The summer rain-out system killed most of the perennial grasses prior to this study. The only soil carbon sources available to the microbial

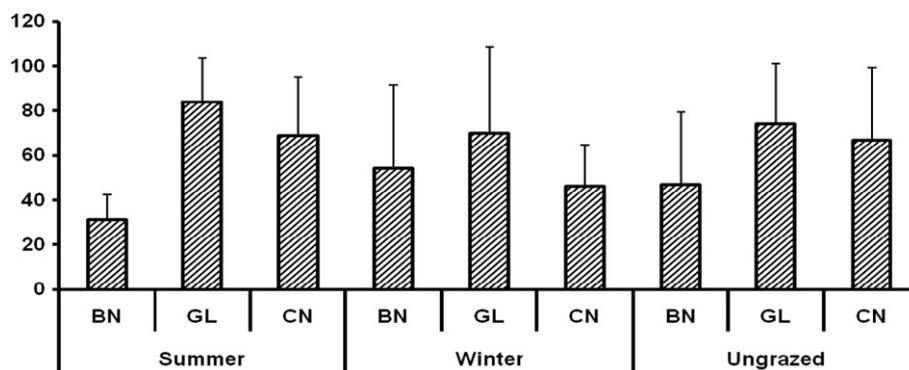


Fig. 2. Mean number of microarthropods per core \pm standard deviation in the burn- and glucose amendment subplots incorporated into the grazing experiment plots. BN = burn; GL = glucose amendment; CN = burn/glucose amendment control.

Table 2

Average number of mites (taxa that occurred in more than 50% of the sample plots) per sample core in the rain-out treatments. SRO = summer rain-out; WRO = winter rain-out; ROC = rain-out control. No. others = number of microarthropods of all other taxa, No. species = number of species of all other taxa of microarthropods.

Taxon	Summer-grazed			Winter-grazed			Ungrazed		
	SRO	WRO	ROC	SRO	WRO	ROC	SRO	WRO	ROC
Tarsonemidae	1.5	10.2	5.5	1.2	8.2	3.0	0.3	4.3	5.0
<i>Siteroptes</i> sp.	0.3	1.0	0.7	0.3	1.5	0	0.3	0.7	2.2
<i>Nanorchestes</i>	0.3	1.5	2.8	0.7	1.7	1.7	0.5	1.2	1.0
<i>Speleorchestes</i>	1.0	4.5	3.5	0.7	2.0	0.8	0	4.3	0.7
<i>Paratydeus</i> sp.	0.2	2.0	2.2	0.2	2.5	0.2	0	1.0	0.7
<i>Tydeus</i> sp.	1.7	12.3	11.5	1.0	16.3	6.7	0.5	10.5	5.0
No. others	13	15	25	2	26	25	4	24	26
No. species	9	7	7	2	10	7	4	11	9

community and the microarthropods in the summer rain-out plots were the roots of winter-spring annual plants. However, since the winter–spring annuals received no rain in the three months prior to the sampling, there were only a few dead stems in most of the summer rain-out subplots. Repeated summer drought in Chihuahuan Desert grassland resulted in a much greater reduction in soil microarthropod abundance and communities than simulated drought in creosotebush (*Larrea tridentata*) shrublands or in mesquite (*P. glandulosa*) shrublands (Whitford and Sobhy, 1999).

This difference is probably the result of litter accumulation under shrubs that provides a source of organic matter for the microbiota. Most of the shrubs in the creosotebush and mesquite shrublands had hemispherical morphologies that retained litter below the canopies (Whitford et al., 1996). In the desert grassland studied here, there was no litter on the surface of any of the subplots.

Simulated winter drought had no effect on the abundance or diversity of the soil microarthropod community. The perennial grass cover of the winter rain-out subplots was not different from the grazing plots in which the rain-out plots were situated. The winter rain-out subplots also had some summer annuals. These observations and the data on the microarthropod communities suggest that the effect of drought on soil microarthropods in arid ecosystems is primarily indirect by reducing the supply of food rather than the direct effect of loss of soil moisture. Several taxa of soil mites are known to be active even in extremely dry soils (Whitford, 1989) and these taxa were the most abundant soil mites in all of the treatments in this study. In Chihuahuan Desert grasslands, the food supply for microarthropods is a function of root turnover and root mass of perennial plants and the roots of annual plants that provide the organic matter base for the microbiota.

The desert grassland microarthropod community shared some, but not all, of the dominant mite taxa with the *L. tridentata* and *P. glandulosa* shrublands. In the shrub communities, nanorchestid and tarsonemid mites were the most abundant whereas the desert grassland soil mite community was dominated by tydeid and tarsonemid mites (Whitford and Sobhy, 1999). Most of the

Table 3

Average number of mites per core sample of mite taxa that occurred in more than 50% of the burned and glucose-amended plots. Sum = summer-grazed plots, Win = winter-grazed plots, Ung = ungrazed plots.

Taxon	Burned			Glucose			Control		
	Sum	Win	Ung	Sum	Win	Ung	Sum	Win	Ung
Tarsonemidae	9.7	16.2	18.5	34.3	15.5	37.2	19.8	14.8	24.0
<i>Speleorchestes</i> sp.	2.8	2.7	2.7	3.5	3.3	1.2	2.2	2.2	5.2
<i>Tydeus</i> sp.	13.0	23.2	23.3	35.7	19.8	30.5	38.7	20.8	27.5

Table 4

Total numbers of microarthropods and number of subplots in which each microarthropod taxon was recorded. Total number of subplots sampled = 108.

Taxon	Number individuals	Number of subplots
Prostigmata		
Omnivores		
Anystidae	33	13
Erythraeidae	39	10
<i>Eupodes</i> sp.	22	11
Linotetranaeidae	24	13
Nematalycidae	11	5
Scutacaridae	20	9
Stigmaeidae	22	11
Teneriffidae	15	9
Trombidiidae	24	11
Predators		
<i>Spinibdella</i> sp.	8	4
<i>Cunaxa</i> sp.	1	1
Cryptostigmata		
Oribatid Inferior	53	9
Mesostigmata		
	6	4
Psocoptera		
	7	5

less abundant taxa were recorded in both grasslands and shrublands.

Burning and glucose soil amendments affected the soil microarthropod community indirectly by decreasing the below-ground carbon in the burned plots and by increasing the below-ground carbon in the glucose-amended soil. Burning reduced both the live and dead root biomass. Population size and biomass of soil bacteria and fungi are a function of available carbon exudates from live roots and carbon compounds in dead roots (Coleman et al., 2004; Mun and Whitford, 1997). Glucose amendment provided available carbon to the soil microbes. Glucose amendments are known to increase microbial biomass and result in nitrogen immobilization (Coleman et al., 2004). Glucose soil amendment affected the fungivore–microbivore tarsonemids and tydeids but had little or no effect on the other prostigmatid mites. The abundance of nanorchestid mites in the shrub communities was attributed to populations of algae and cyanobacteria in the soils associated with shrubs (Whitford and Sobhy, 1999).

The results from the fire plots must be interpreted with caution as a single point in time measurement four years after the small plots had been burned. In arid regions, the effects of fire may extend over long time periods because of the cascading effects of climate variability (Tongway and Hodgkinson, 1992). The responses of the soil microarthropod communities to fire and intensive grazing cannot be interpreted as an equilibrium condition. Clearly, the microarthropod communities could be very different one or more years following this study.

In conclusion, this study provides evidence that short-term intensive grazing by livestock has no deleterious effect on soil microarthropods, despite reducing microtopography. Extreme summer drought reduces the abundance and diversity of soil microarthropods by killing perennial grasses, thereby reducing the organic matter base for the soil microbiota. Extreme winter drought does not affect perennial grasses and, therefore, has little effect on the soil microbiota of the grassland.

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