

Livestock grazing effects on ant communities in the eastern Mojave Desert, USA

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Abstract

The effects of livestock grazing on composition and structure of ant communities were examined in the eastern Mojave Desert, USA for the purpose of evaluating ant communities as potential indicators of rangeland condition. Metrics for ant communities, vegetation, and other ground-cover elements were evaluated as a function of distance from livestock water tanks, which represents a gradient in level of livestock activity in desert settings. Data were collected at six isolated water tanks used by cattle during early summer, with seven plots (90 m × 90 m; 100 pitfall traps) per tank.

Thirty-eight species of ants were recorded, with an average of 14 ant species per plot. Ant species richness did not differ as a function of distance from the water tank. Also, overall species composition, as measured by a similarity index for species presence/absence for paired-comparisons of plots, did not show differences attributable to the gradient in grazing impact. In contrast, the relative abundance of several taxa and functional groups was significantly related to distance from the water tank. The predominant pattern was for the greatest abundance to occur at the water tank, with little difference in ant abundance among plots away from the water tank. This pattern was shown by the abundant ants species, *Conomyrma bicolor* and *Pheidole tucsonica*, and the groups *Conomyrma* spp., *Pheidole* spp., homopteran tenders, and plant foragers. However, two species, *Aphaenogaster megommata* and *Monomorium wheelerorum* showed the greatest relative abundance at a distance away from the water tank. A number of ant metrics were significantly related to ground-cover metrics ($R^2 > 0.5$). Organic debris was the variable most frequently related significantly to ant abundance metrics, always in a positive direction, followed by cover for perennial grasses, annual forbs, and shrubs, and bare patch size. Ant community metrics in the study region appear to have little potential to serve as indicators of rangeland condition because differences were evident primarily in severely degraded localized conditions rather than in intermediate widespread conditions.

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1. Introduction

Commercial livestock grazing has comprised a major land use in the arid southwestern US since the late 15th century, with the greatest intensity occurring during the early 20th century (Hess, 1992). Vegeta-

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tion composition and cover have changed over much of this region during this time as a result of grazing, other land uses, and climate variability (Hastings and Turner, 1965; Buffington and Herbel, 1965; Harniss and West, 1973; Bahre and Shelton, 1993). In some areas the combined effects of insufficient rainfall and overgrazing have produced vegetation changes that appear irreversible (Whitford, 1995). Changes in plant community structure have also been accompanied by changes in the structure of animal communities (Jones, 1981; Waser and Price, 1981; Jepson-Innes and Bock, 1989; Heske and Campbell, 1991; Whitford, 1997; Nash et al., 2001).

A need has been recognized to develop methods for the assessment and monitoring the environmental condition of the arid and semi-arid rangelands of North America (National Research Council, 1994). Thus, indicators are needed that are sensitive to disturbance (e.g., livestock grazing) and can be consistently applied across large areas. Soil, vegetation, and fauna are major elements in many ecological processes in rangeland systems, and reflect rangeland biodiversity (McGeoch, 1998). Faunal species composition and abundance can be especially instructive of widespread conditions because they are often determined by processes operating well beyond the site (e.g., patch dynamics), such that local soil and vegetation traits represent only a subset of the factors determining the faunal characteristics of a site (e.g., Knick and Rotenberry, 1995). Faunal indicators have been little explored in evaluating ecological condition in rangeland systems (National Research Council, 1994).

This study explores the potential for ant species assemblages to be used as one of a suite of indicators of Mojave Desert rangeland condition. Ant communities have a number of attributes that may make them particularly useful as indicators of ecosystem change. They are often extremely abundant, have relatively high species richness, include many specialist species, include some species at higher trophic levels, are responsive to changing environmental conditions, are easily sampled, and are relatively easily identified (Majer, 1983; Andersen, 1990). Ants are ubiquitous in subtropical, warm temperate, and cool temperate deserts of the world, and are important in such semi-arid and arid ecosystems because of their abundance and trophic diversity. Ants also exhibit a diversity of behaviors, and affect important ecosystem processes

and properties in a variety of ways (Hölldobler and Wilson, 1990). Vegetation changes potentially affect the temporal availability, quality, and quantity of food for many ant species. Soil nesting ants should respond to changes in soil physical characteristics (Johnson, 1998).

A number of studies have examined the effects of different habitat disturbances on ant communities, including changes after fire (Andersen and McKaige, 1987; Andersen, 1991), mining (Majer, 1984, 1985; Majer et al., 1984, Majer and Nichols, 1998), forest clear-cutting (Jennings et al., 1986; Whitford and Gentry, 1981), soil-vegetation changes resulting from over-grazing and drought (Wisdom and Whitford, 1981; James et al., 1999), clearing for agriculture (Lobry de Bruyn, 1993), and agricultural tillage practices, soil conditions, and insecticide use (Peck et al., 1998). Ant communities are likely to be affected by a widespread land-use practice such as livestock grazing. Effects of livestock grazing on vegetation have been well documented, and effects on soil physical characteristics have also been demonstrated (e.g., Marrs et al., 1989; National Research Council, 1994; Whitford, 2002; Ares et al., 2003). Many studies have been conducted on whole ant communities in western North America (Chew, 1977; Whitford, 1978; Bernstein and Gobbel, 1979; Andersen, 1997), but few have examined ant communities over gradients in grazing intensity. In the arid western US, ant communities have been examined relative to livestock grazing in the Chihuahuan, Sonoran, and Great Basin Deserts (Chew, 1977; Whitford et al., 1999; Nash et al., 1998, 2001), but not in the Mojave Desert.

The objectives of this study were:

1. To identify and evaluate metrics of ant community composition and structure that potentially may be useful as indicators of ecological condition in desert scrub in the Mojave Desert.
2. To determine the influence of livestock grazing on the composition and structure of ant communities in the Mojave Desert.
3. To determine the relationships between ant community composition and structure and land-cover parameters.

To address these objectives, we sought to quickly and accurately estimate relative abundance of each ant species in a site, and identify relationships between

metrics describing the ant species assemblage and livestock grazing impacts along a gradient of grazing intensity (Karr, 1993). In particular, we sought to make a “first cut” in identifying potential indicators by identifying metrics that can distinguish between extremes in ecological condition (Fore et al., 1996). A gradient in grazing intensity is provided in the eastern Mojave Desert by scattered livestock water tanks. These facilities in arid regions can show extreme disturbance (both from grazing and trampling) close to the tank and progressively less disturbance (primarily from grazing) as distance increases away from the tank. Such progressively decreasing impact with distance as a result of grazing has been shown for vegetation characteristics (Lange, 1969; Graetz and Ludwig, 1978; Andrew and Lange, 1986; Fusco et al., 1995), vegetation patterns (deSoyza et al., 1997; Nash et al., 1999), and soil microtopography (Nash et al., 2003). Ant communities may reflect differences in ecological condition resulting from grazing effects on these or other factors. Ant species composition and abundance are affected by the type and abundance of food sources (Andersen, 1990; Sanders and Gordon, 2003), which ultimately derives primarily from vegetation.

We tested hypotheses in three categories. (1) Metrics describing ant community composition and structure differ as a function of distance from the water tank. Ant metrics used to reflect community composition were species richness, species similarity index, and the relative abundance of individual species and genera; metrics used to reflect community structure were functional groups for feeding and foraging microhabitat. (2) Land-cover metrics reflecting vegetation cover, organic debris, and bare patch size differ as a function of distance from the water tank. Metrics were percent cover by vegetation class, dead standing vegetation, and organic debris, and average bare patch size. (3) Metrics for ant community composition and structure are related to the above land-cover metrics.

2. Materials and methods

2.1. Study sites and study design

The study was conducted at six sites centered on water tanks used by cattle in the eastern Mojave Desert

(Fig. 1). These sites were selected because they had been recently sampled as part of a separate study of the effects of livestock grazing on the desert tortoise (*Gopherus agassizii*) conducted by the US Geological Survey (P. Medica, personal communication). Criteria for site selection in the latter study included that sites were isolated from other tanks such that radiating transect lines did not intersect within 6 km from each tank, had relatively homogeneous soil and ecological communities within several kilometers, showed relatively little impact except from grazing activities, and were dispersed across the eastern Mojave Desert. All sites were on open plains or alluvial fans between 640 and 1160 m elevation, and were dominated by two widespread and common plants in the Mojave Desert, *Larrea tridentata* and *Ambrosia dumosa* (Hickman, 1993). Depth to hardpan was less than 1 m in 32 of the 42 plots (described below); for the 32 plots depth to hardpan was averaged 0.63 m deep, and the minimum hardpan depth was 0.15 m. Soil texture was mostly sandy, and was often mixed with stones and/or cobbles that made sampling a challenging task. Cattle grazing in association with the water tanks had occurred for many years through the year of sampling at four sites, and until 2–3 years previously at two sites (Tenmile and Loran sites).

Sampling for ants and vegetation was conducted at seven plots (90 m × 90 m) at each of the six sites (total of 42 plots). Plots were positioned along two transects radiating out from the water tank (Fig. 2). The two transects were selected from a set of five transects with random azimuths established in the USGS study, given the constraint that transects could not cross roadways or railroads within 6 km. In the present study, the first transect was selected at random from the five available and the second was selected as the one with greatest angle between it and the first one. Plots were established adjacent to the water tank, and along the two transects at 200, 800, and 1600 m from the tank. No water spillage or leakage occurred on the plot adjacent to the water tank.

2.2. Ant sampling

Ants were sampled by pitfall trapping, which is a rapid, repeatable, and quantitative method that provides a relatively unbiased sample of the ants in the community in relatively open habitat (Andersen,

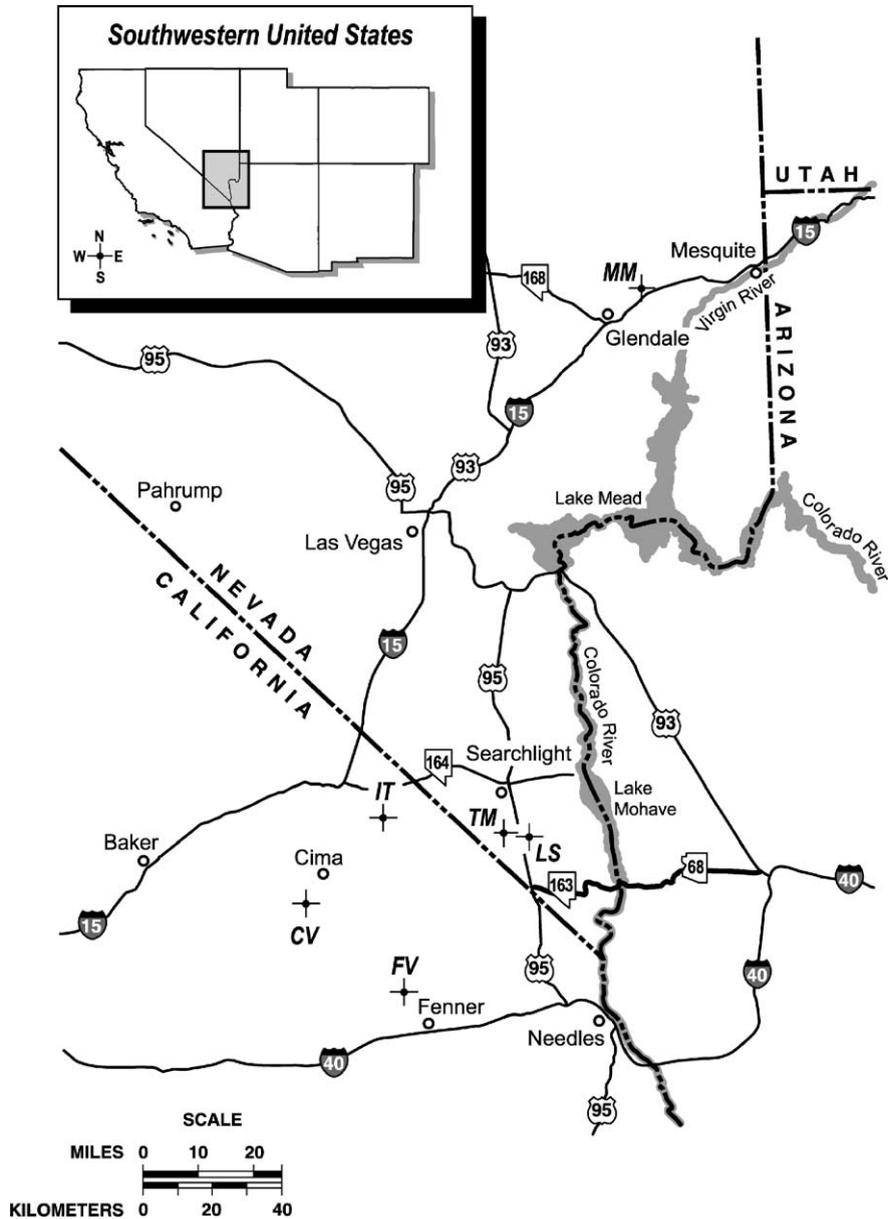


Fig. 1. Locations of the six sites in the Mojave Desert study area. CV, Cima Valley; FV, Fenner Valley; IT, Ivanpah Tank; LS, Loran Site; MM, Morman Mesa; TM, Tenmile.

1990). Pitfall traps (38 mm × 70 mm tall plastic vials) were placed in a 100-trap grid of 10 lines of traps with 10 m spacing between traps and lines, one grid per plot (Fig. 2). Thus, the total number of traps for the entire study was 4200 (six sites × seven plots/site × 100 traps/plot). Each vial was filled to a depth

of 30 mm with a mixture of 70% ethanol and 30% glycerol (Greenslade and Greenslade, 1971). Vials were carefully placed in holes in the soil, which were back-filled to make the soil surface level with the lip of the vial. Traps were left in place for 24 h, retrieved, labeled, and stored for shipment to the laboratory.

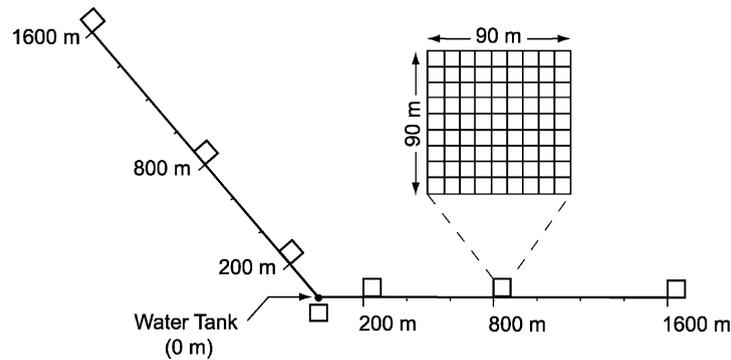


Fig. 2. Arrangement of plots along two transects, and pitfall traps grid comprising one plot, at one site. The angle between the two transects varied among sites (see text). Pitfall traps ($n = 100$) were set at the grid nodes.

The study was conducted in early summer (2 June 1994 and 22 July 1994), a time when ants are active above ground and seed production of annual plants is completed. There was a light rain on 18 and 19 June. Sampling was conducted at one site over several days before moving to another.

All ants in a vial were identified to species or to an operational taxonomic unit (Table 1). We used relative abundance per trap rather than absolute abundance because some traps near nests collected extreme numbers of individuals. The relative abundance of each ant species was determined by ranking the number of ants per species in a trap, where 1–5 ants = 1, 6–10 ants = 2, 11–20 ants = 3, 21–30 ants = 4, 31–40 ants = 5, 41–50 ants = 6, >51 ants = 7 (Whitford et al., 1999). Ranks were summed for each species for all traps in a plot; the sum was divided by the total number of recovered traps, yielding average relative abundance per trap. Each ant species was assigned to one of the following functional groups for feeding based on literature for the species or related species: generalist, seed harvester, homopteran tender, or liquid feeder (Table 1). Likewise, each species was categorized into a functional group for foraging microhabitat: plant canopy forager, soil surface forager, or both (Table 1).

To test for an effect of livestock grazing on ant species composition based on species presence/absence, we computed a species similarity index for each pairwise comparison of plots along each of the two transects at each site. This yielded six distance combinations for each of the two transects (0–200,

0–800, 0–1600, 200–800, 200–1600, 800–1600), for a total of 72 comparisons among the six sites. The species similarity index was computed as the number of species present in both plots in the comparison, divided by number of species that occur in the two plots collectively. We hypothesized that, because livestock grazing intensity is expected to diminish with distance from the water tank, the further apart two plots are along the transect, the lower would be their similarity index. We recognize that this hypothesis could also be attributed to distance alone, rather than a grazing effect, because species composition would normally be expected to be spatially correlated.

2.3. Vegetation and other sampling

Perennial plant cover and species richness were measured at each plot using a line intercept method on two 90-m transects (Canfield, 1941) adjacent to two of the pitfall trap grid lines, 50 m apart. Perennial plant species were grouped into four classes: shrubs, grasses, forbs and succulents (Table 2). Total perennial vegetation cover was computed as the sum for these four groups, ignoring a trivial amount of cover in which two classes overlapped. Cover for dead standing perennial vegetation, where it did not overlap live perennial cover, and cover for organic debris (e.g., leaves, cattle dung), which could overlap any of the vegetation categories were also measured in the same manner as other cover measurements. Bare patch size was also determined as the distance between live perennial plants.

Table 1

Frequency of occurrence of the 38 ant species at the six sites (42 total plots; 4104 total traps recovered from 4200 traps set)

Species	Functional group	Foraging medium	Number of sites	Number of plots	Number of traps	Number of individual
<i>Aphaenogaster cockerelli</i>	G	Soil	5	25	1237	9004
<i>Aphaenogaster megommata</i>	G	Soil	5	28	950	1815
<i>Conomyrma bicolor</i>	HT	Plant	4	14	90	1154
<i>Conomyrma insana</i>	HT	Plant	6	39	358	884
<i>Crematogaster depilis</i>	G	Plant/soil	6	34	241	528
<i>Crematogaster larreae</i>	G	Plant/soil	2	10	108	151
<i>Forelius foetidus</i>	HT	Plant	6	41	1871	6852
<i>Forelius pruinosus</i>	HT	Plant	4	20	455	1057
<i>Leptothorax obliquicanthus</i>	G	Plant	2	3	9	20
<i>Messor lariversi</i>	SH	Soil	1	1	1	1
<i>Messor pergandei</i>	SH	Soil	5	24	157	2004
<i>Monomorium wheelerorum</i>	G	Plant/soil	4	15	27	42
<i>Myrmecocystus depilis</i>	LF	Plant	5	15	92	153
<i>Myrmecocystus flaviceps</i>	LF	Plant	2	3	5	5
<i>Myrmecocystus mendax</i>	LF	Plant	4	8	14	16
<i>Myrmecocystus mexicanus</i>	LF	Plant	4	16	43	70
<i>Myrmecocystus navajo</i>	LF	Plant	2	2	5	8
<i>Myrmecocystus yuma</i>	LF	Plant	1	1	1	1
<i>Myrmecocystus species 02</i>	LF	Plant	2	2	8	11
<i>Pheidole desertorum</i>	SH	Soil	1	3	56	282
<i>Pheidole gilvescens</i>	SH	Soil	6	37	331	958
<i>Pheidole sitarches soritis</i>	SH	Soil	5	31	381	1304
<i>Pheidole tucsonica</i>	SH	Soil	5	32	391	2556
<i>Pheidole xerophila</i>	SH	Soil	1	1	1	1
<i>Pheidole species 02</i>	SH	Soil	5	22	101	342
<i>Pheidole species 20</i>	SH	Soil	3	6	6	6
<i>Pheidole species 31</i>	SH	Soil	1	1	1	2
<i>Pheidole species 35</i>	SH	Soil	2	5	8	19
<i>Pheidole species 36</i>	SH	Soil	5	22	80	139
<i>Pheidole species 39</i>	SH	Soil	1	1	11	15
<i>Pogonomyrmex californicus</i>	SH	Soil	6	27	351	689
<i>Pogonomyrmex desertorum</i>	SH	Soil	1	1	1	20
<i>Pogonomyrmex imberbicus</i>	SH	Soil	6	30	144	185
<i>Pogonomyrmex rugosus</i>	SH	Soil	6	28	762	6493
<i>Pogonomyrmex species 01</i>	SH	Soil	1	1	26	32
<i>Pogonomyrmex species 02</i>	SH	Soil	1	1	1	1
<i>Solenopsis molesta</i>	G	Plant/soil	3	5	5	5
<i>Solenopsis xyloni</i>	G	Plant/soil	6	38	454	2108

Functional groups are denoted as: G, generalist; HT, homopteran tender; LF, liquid feeder; SH, seed harvester. ANOVA was done for species with >32 individuals (see text).

Annual grasses and forbs were sampled by frequency in which plants were present or absent in 12 frames (20 cm × 20 cm) on each plot. Frames were placed in both the shrub subcanopy ($n = 6$ frames) and intershrub ($n = 6$ frames) spaces. To sample the subcanopy composition, the nearest shrub to the 15, 45, or 75 m positions on the two vegetation transects was identified and the frame was centered on the tape either under the canopy (if the shrub was large enough

to accommodate this placement and cover the frame) or over the shrub (if the shrub was too small to provide sufficient canopy to cover the frame). To sample the intershrub space, the intershrub space nearest the designated meter mark that was large enough to accommodate the frame was identified and the frame was centered on the tape midpoint in this intershrub area. All annual plants were dead at the time of sampling.

Table 2
Frequency and percent cover of perennial plants species at the six study sites and 42 total plots

Species	Growth form	Number of sites	Number of plots	%Cover
<i>Acacia greggii</i>	S	2	4	0.19
<i>Acantopappus sphaerocephalus</i>	S	4	15	0.33
<i>Ambrosia dumosa</i>	S	6	36	4.29
<i>Brickellia incana</i>	S	2	2	0.06
<i>Cassia armata</i>	S	2	3	0.06
<i>Croton californicus</i>	F	1	2	0.02
<i>Cucurbita palmata</i>	F	1	2	0.03
<i>Encelia frutescens</i>	S	2	2	0.08
<i>Ephedra nevadensis</i>	S	3	6	0.09
<i>Eriogonum fasciculatum</i>	S	2	5	0.01
<i>Eriogonum inflatum</i>	F	2	2	0.11
<i>Gutierrezia microcephala</i>	S	2	3	0.02
<i>Hymenoclea salsola</i>	S	4	11	0.45
<i>Krameria grayi</i>	S	3	15	0.29
<i>Krameria erecta</i>	S	5	16	0.53
<i>Larrea tridentata</i>	S	6	42	9.74
<i>Lycium andersonii</i>	S	1	2	0.02
<i>Lycium cooperi</i>	S	1	1	0.03
<i>Opuntia acanthocarpa</i>	SU	1	1	0.02
<i>Opuntia echinocarpa</i>	SU	4	11	0.07
<i>Opuntia ramosissima</i>	SU	5	14	0.14
<i>Oryzopsis hymenoides</i>	G	1	1	0.002
<i>Pleuraphis rigida</i>	G	3	14	0.70
<i>Psilostrophe cooperi</i>	S	1	2	0.03
<i>Salazaria mexicana</i>	S	4	8	0.05
<i>Sphaeralcea ambigua</i>	F	2	6	0.03
<i>Stephanomeria pauciflora</i>	F	1	2	0.003
<i>Thamnosma montana</i>	S	2	2	0.04
<i>Xylorhiza tortifolia</i>	S	1	1	0.01
<i>Yucca schidigera</i>	SU	3	7	0.12
Total				17.43

All plants were native. For growth form: F, forb; G, grass; S, shrub; and SU, succulent, “%Cover” applies to cover across all six sites and 42 plots.

3. Statistical analysis

Analysis of variance was used to test the hypotheses that ant metrics (species richness and relative abundance) and ground-cover metrics vary as a function of distance from the water tank. In these analyses we used the general linear model with least-square means option for multiple comparisons of means (Proc GLM; SAS, 1998). For annual vegetation, split plot analysis of variance was used to test the significance of the main effect, and the “*t*” test was used for multiple comparisons of means (Proc GLM; SAS, 1998). For

comparisons of ant species composition the similarity index, differences between distances were tested using ANOVA and the Bonferroni “*t*” test for multiple comparisons of means (Proc ANOVA; SAS, 1998). Multiple regression analyses (Proc Reg; stepwise selection; SAS, 1998) were used to examine relationships for ant metrics as a function of ground-cover metrics. Collinearity between independent variables was below 0.85. The significance level for all statistical tests was <0.05.

4. Results

4.1. Ants

Thirty-eight species of ants were recorded in the study, eight of which were recorded at all six sites: *Conomyrma insana*, *Crematogaster depilis*, *Forelius foetidus*, *Pheidole gilvescens*, *Pogonomyrmex californicus*, *Pogonomyrmex imberbicus*, *Pogonomyrmex rugosus*, and *Solenopsis xyloni* (Table 1). Fifteen of the 38 species occurred at >50% of the 42 plots among the six sites. Eight species were recorded on only one plot at a single site. Over 6000 individuals were collected in pitfall traps for three species: *Aphaenogaster cockerelli*, *F. foetidus*, and *P. rugosus*. The most frequently represented functional group for feeding, based on the number of species collected in the study, was seed harvester, followed by generalist, liquid feeder, and homopteran tender.

Ant species richness per plot differed significantly among the six sites (d.f. = 5, 33; $F = 16.87$; $P < 0.0001$). Ant species richness values per plot averaged 16.6 at Cima Valley, 17.6 at Fenner Valley, 10.5 at Ivanpah Tank, 13.1 at Loran Site, 9.2 at Mormon Mesa, and 18.1 at Tenmile. In contrast, ant species richness did not differ significantly as a function of distance from the water tank (d.f. = 3, 33; $F = 0.40$; $P = 0.77$). Mean species richness for each distance did not deviate much from 14 ant species (minimum = 13.6, maximum = 14.7).

The species similarity index, which was based on species presence absence, ranged from 37.5 to 92.9% among the 72 paired plots. The averages for the specific distance comparisons across all sites decreased as distance between the plots increased, from an average of 68.0% for the 0–200 m comparison (200 m

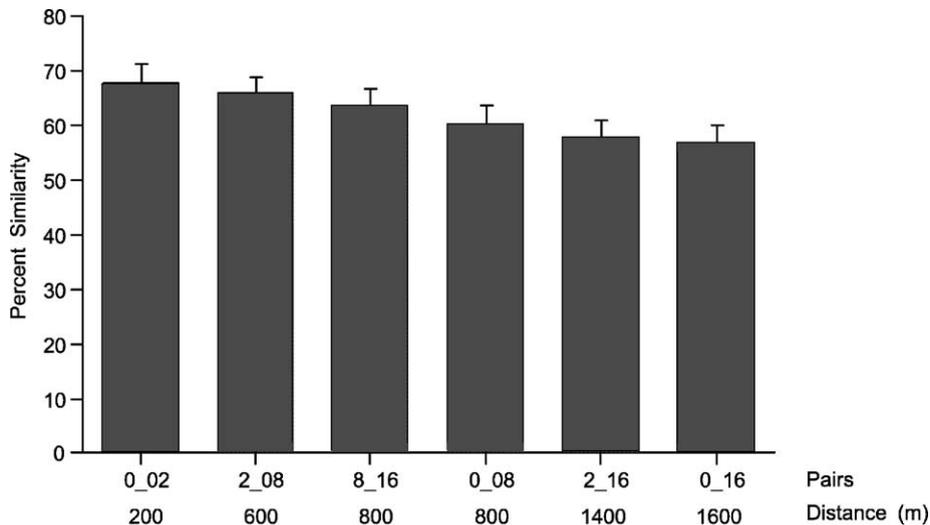


Fig. 3. Average species similarity index (percent) for pairwise comparisons of plots as a function of distance from the water tank. “Pairs” refers to the distance of two plots from the water tank along a single transect line: 0.02 (0–200 m), 0.08 (0–800 m), 0.16 (0–1600 m), 2.08 (200–800 m), 2.16 (200–1600 m), 8.16 (800–1600 m). “Distance” refers to linear distance between the plots compared. Sample size is 12 for each set of pairwise comparisons; total sample size is 72 pairwise comparisons across six sites. Vertical lines represent one standard error for each group ($n = 12$).

differential) to 57.0% for the 0–1600 m comparison (1600 m differential; Fig. 3). The overall analysis of variance was significant for distance (d.f. = 5, 61; $F = 2.62$; $P = 0.0328$); however, none of the paired distance combinations were significantly different from each other in post hoc comparisons (minimum significant difference using Bonferroni test was 11.9%).

The relative abundance of several taxa and functional groups was significantly related to distance from the water tank (Fig. 4). The predominant pattern was for the greatest ant abundance to occur at the water tank. Among individual ant species, *Conomyrma bicolor* and *Pheidole tucsonica* were significantly more abundant at the water tank (0 distance), whereas abundances at 200, 800, and 1600 m from the water tank did not differ significantly from each other. Several other ant species also showed peak abundance at the water tank, but this difference was not statistically significant: *F. foetidus*, *Mrymecocystus depilis*, *Pheidole gilvescens*, *P. sitarches soritis*, *Pheidole* 02, *P. californicus*, and *S. xyloni*. Among abundant genera, *Conomyrma* spp. and *Pheidole* spp. showed a significant pattern of high relative abundance at the water tank relative to all other distances from the tank. Among

functional groups, relative abundance for homopteran tenders and plant foragers was significantly greater at the water tank than all other distances. In contrast to this pattern, *Aphaenogaster megommata* and *Monomorium wheelerorum* showed the greatest relative abundance at a distance away from the water tank (200 m for *A. megommata*, 1600 m for *M. wheelerorum*).

4.2. Vegetation

4.2.1. Perennial plants

Thirty perennial plant species were recorded on the vegetation transects, all of which are native to the region (Table 2). Perennial plant diversity was dominated by shrubs (19 species), followed by forbs (five species), succulents (four species), and grasses (two species). Percent cover by all perennial vegetation averaged 17.4% among all sites and plots. Shrubs comprised most (16.2%) of this cover; grasses averaged only 0.7%, and forbs averaged only 0.2% (Table 2). Two shrubs, *L. tridentata* (creosote bush) and *A. dumosa* (burrobush) comprised the dominant cover (9.7 and 4.3%, respectively), and were nearly ubiquitous, occurring on 100 and 86% of the plots, respectively.

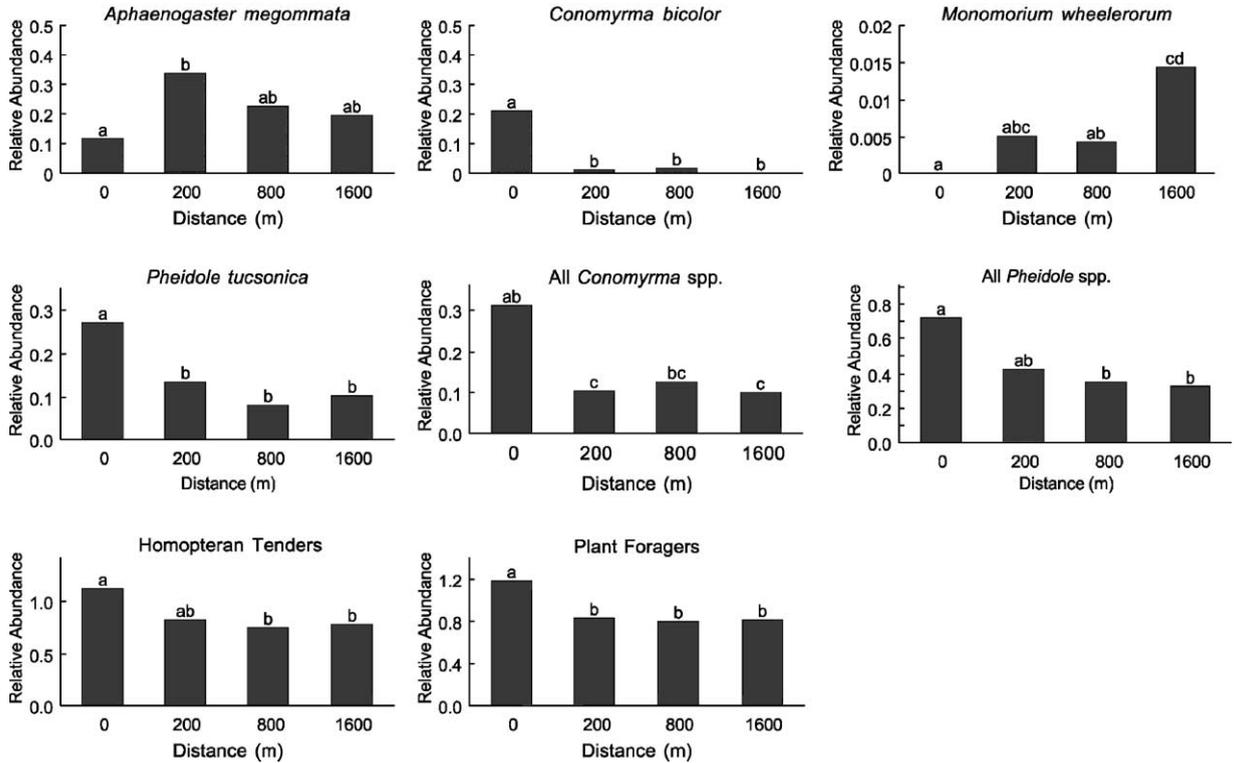


Fig. 4. Relative abundance of ant species and groups showing a significant relationship (ANOVA, $P < 0.05$) with distance from the water tank. Bars with the same letter indicate no significant difference in post-hoc comparisons ($P < 0.05$). Sample size is six plots for 0 distance, 12 for other distances.

Only three other plants occurred on more than a third of the plots, *Krameria erecta* (range ratany), *Krameria grayi* (white ratany), and *Acamtopappus sphaerocephalus* (rayless goldenhead), but cover was less than 0.5% for each species. Perennial grasses occurred only on three sites (Loran Site, Morman Mesa, and Tenmile) and were composed almost exclusively of *Pleuraphis rigida* (dixie grass). Plant species richness ranged from 6 to 19 species per site: Cema Valley (12 species), Fenner Valley (17 species), Ivanpah Tank (six species), Loran Site (nine species), Morman Mesa (17 species), Tenmile (19 species).

Percent cover for several vegetation categories differed as a function of distance from the water tank (Fig. 5). For shrubs and total perennial vegetation, cover was significantly lower at the water tank, and differences were not evident between 200, 800, and 1600 m from the tank. In a largely complementary fashion, average bare patch size was significantly

greater at the water tank, and differences between distances away from the tank were not significant. Bare patch size averaged 3.67 m among all plots ($n = 42$). For succulents and dead standing vegetation, cover was also the lowest at the water tank, but the differences were statistically different only between distances of 0 and 1600 m. Percent cover for perennial grasses and forbs, and organic debris, was not significantly related to distance from the water tank. Average perennial species richness increased with increasing distance from the water tank; however, there were no significant differences among averages ($F = 1.76$; d.f. = 3, 33; $P = 0.17$).

4.2.2. Annual plants

The frequency of annual grasses differed significantly as a function of distance from the water tank (Fig. 6). As with several of the perennial vegetation categories, the frequency of annual grasses was low-

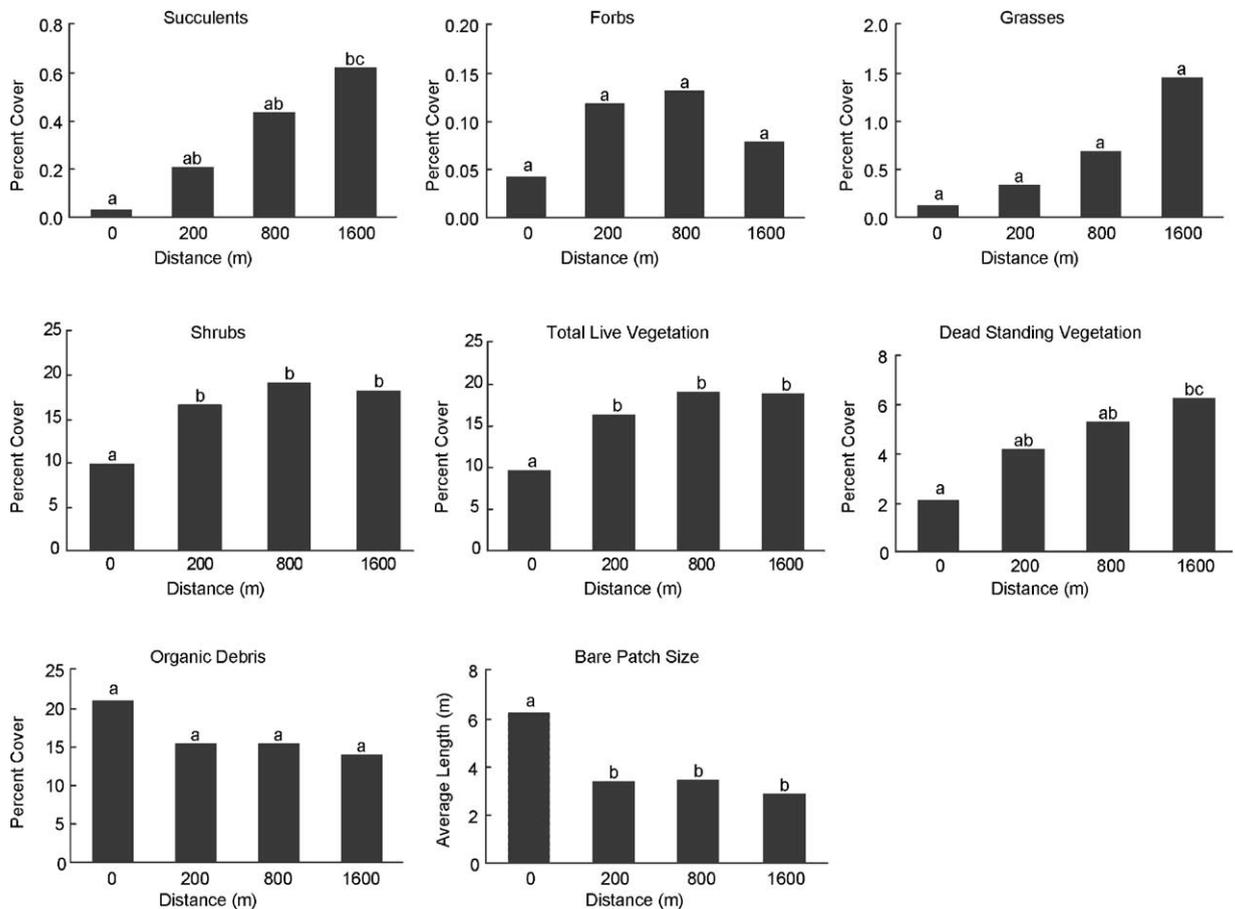


Fig. 5. Average percent cover of perennial vegetation, dead standing vegetation, and organic debris, and average length (m) of bare soil, as a function of distance from the water tank. Bars with the same letter indicate no significant differences in post-hoc comparisons ($P < 0.05$). Sample sizes are as Fig. 4.

est at the water tank. The relationship with distance, however, was curved, with no significant difference between the frequency at 0 and 1600 m from the tank. The frequency of annual grasses was significantly greater underneath the shrub canopy (average number of frames, 4.56) than in between shrubs (average number of frames, 3.94; $F = 4.93$; d.f. = 1, 56; $P = 0.0305$). For annual forbs, in contrast, frequency did not differ significantly as a function of distance from the water tank, nor between frames underneath the shrub canopy and frames in between shrubs. The frequency of annual forbs was low, averaging only 1.13 frames out of the possible 12 (Fig. 6).

4.2.3. Relationships between ants, vegetation, and other metrics

A number of ant metrics were significantly related to ground cover metrics ($R^2 > 0.5$; Table 3). Ant species richness was positively related to perennial shrub cover, and negatively to cover of annual forbs and dead standing vegetation. For the nine significant metrics for ant relative abundance, the independent variable that was most frequently found to be significant was organic debris cover. It was significant in all nine relationships, all in a positive direction. Cover of dead standing vegetation was significantly related to five abundance metrics (all negative). The number and direction of significant relationships for the other

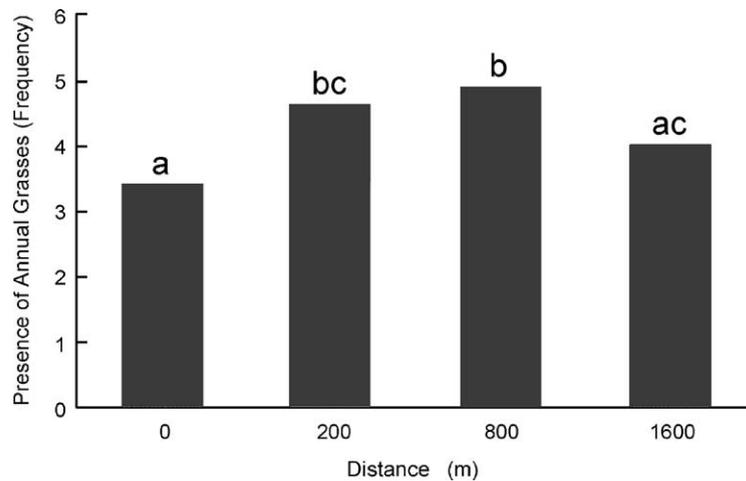


Fig. 6. Average number of frames per plot (12 frames total) where annual grasses were present. Sample sizes are the same as in Fig. 4.

Table 3

Coefficients and R^2 for significant ($P < 0.05$) relationships between ant metrics (dependent variables) and ground-cover metrics (independent variables) in multiple regression analyses ($n = 42$)

	S	G	AnF	DSV	OD	Bare	R^2
Species richness	0.274		-0.543	-0.206			0.61
<i>Conomyrma bicolor</i>					0.011		0.51
<i>Forelius foetidus</i>	0.016	0.086			0.030		0.64
<i>Myrmecocystus depilis</i>	-0.004		-0.004		0.005		0.55
<i>Pheidole tucsonica</i>				-0.011	0.009		0.52
All <i>Conomyrma</i> spp.					0.016		0.59
All <i>Forelius</i> spp.		0.098		-0.021	0.021	-0.048	0.72
Homopteran tenders		0.103		-0.02	0.037	-0.046	0.75
Seed harvesters			-0.075	-0.087	0.039		0.53
Plant foragers		0.156	-0.027	-0.030	0.043		0.78

Except for species richness, all ant metrics refer to relative abundance. The following metrics refer to percent cover: S, perennial shrub; G, perennial grass; DSV, dead standing vegetation; and OD, organic debris. The units for AnF (annual forb) are frequency (see text). Bare is average bare patch size (m). Perennial forbs and annual grasses were not significant in any analyses.

metrics were: grass cover (four, all positive), annual forb cover (three, all negative), shrub cover (two, one negative, one positive), and bare patch size (two, both negative). The relationship between ant species richness and perennial plant species richness in a separate regression of these two variables alone was not significant.

5. Discussion

Distance away from the water tank appeared to represent a strong gradient in ecological impact from live-

stock grazing activities. The influence of livestock activity on ground cover variables was conspicuous at the water tank, but was much less evident at distances away from the tank. Indeed, percent cover for shrubs (and total perennial vegetation) was significantly reduced at the water tank by a factor of about two in comparison to distances away from the tank, but cover levels did not differ among plots away from the tank. A complementary pattern was evident for bare patch size, which was approximately twice as large at the water tank than at distances away from the tank. This two-fold relationship for both shrub cover and bare patch size suggests that livestock activities resulted in

the elimination of about half the shrubs, rather than reduction in half the cover of individual shrubs. Cover for perennial succulents, dead standing vegetation, and annual grasses also showed significant differences as a function of distance from the water tank, with the lowest levels occurring at the water tank.

Although it is evident that the impacts observed at the water tank were severe, we do not know how the impacts at 1600 m from the water tank compare either to pre-grazing conditions or to conditions that would result from the suspension of grazing. Nevertheless, one study suggests that grazing has indeed affected the vegetation away from water tanks in the study region. Avery (1998) compared vegetation characteristics between a grazed area and one protected from grazing from 1981 to 1994 in the vicinity of our Ivanpah Tank site. The most striking difference observed was a 42% greater above-ground estimated biomass of the perennial grass, *Pleuraphis rigida*, in the protected plot. Percent cover of total perennial vegetation, however, did not differ significantly between two plots.

The impacts of livestock grazing on the ant community were manifested as changes in relative abundances of some species, genera, and functional groups, and did not appear to influence species presence/absence. That is, species richness was not significantly related to distance from the water tank, and the species similarity index did not show differences attributable to the gradient in grazing impact. Although the species similarity index between pairwise comparisons of plots decreased as distance between the plots increased, this pattern could be explained by either spatial autocorrelation, wherein close plots tend to have similar traits, or a grazing effect. These two potential effects cannot be distinguished from each other because the gradient in grazing intensity is represented as distance away from the water tank. Nevertheless, there is no evidence that the plot at the water tank, i.e., the point of greatest livestock activity, differed significantly from other plots. None of the values for the similarity index for pairwise comparisons involving this plot (i.e., 0–200 m, 0–800 m, and 0–1600 m) was significantly different from any other pairwise combination in post hoc comparisons.

The influence of livestock grazing on relative abundance was most evident for two species, *C. bicolor* (a hompteran tender and plant forager) and *P. tucsonica* (a seed harvester and soil forager). Both of

these species showed high relative abundance at the water tank, and little difference in relative abundance among distances away from the tank. Although several group metrics also showed this pattern (all *Conomyrma* spp., all *Pheidole* spp., homopteran tenders, and plant foragers), data for the two species above dominated these other metrics. In contrast to the pattern for these species and metrics, *M. wheelerorum* (a generalist and plant/soil forager) was most abundant at 1600 m, and was absent at the water tank.

A number of ant metrics were significantly related to ground-cover metrics. Organic debris was the variable most frequently related significantly to ant abundance metrics, always in a positive direction, followed by cover for perennial grasses (positive), annual forbs (negative), and shrubs (positive and negative), and bare patch size (negative). It is surprising, however, that the most conspicuous effect of livestock grazing on the ant community was an increase in abundance of two ant species at the water tank where much of the vegetation had been eliminated by livestock, and soil disturbance was conspicuously greatest. Interestingly, in the northern Chihuahuan Desert, a conspecific of one of these species, *Conomyrma insana*, was abundant and responded negatively to the impact of livestock activity (Nash et al., 1998). It is possible that abundances in the present study were greatest at the water tank in association with an abundance of cattle dung. Although the greater cover of organic debris (which included dung patties) observed at the water tank than elsewhere was not statistically significant, it is possible that total dung levels (i.e., both patties and smaller particles) were greatest at the water tank. Dung may serve as a source of food or moisture, or as a source of insects attracted to the dung. In the northern Chihuahuan Desert, *S. xyloni* colonized soil underneath cow dung, and was more abundant within areas of maximum grazing impact (Nash et al., 1998).

The findings that the relative abundance of several taxa and functional groups were affected by livestock grazing activities suggest that these metrics may be candidates for use as indicators of rangeland condition. That is, the metrics make the “first cut” in identifying potential indicators because they distinguish between extremes in condition (Fore et al., 1996). However, the effects of livestock activities on most of the significant ant metrics were manifested primarily at the water tank, where impacts were severe, with little

difference in metric values among the three distances away from the water tank. Thus, these metrics may have little potential to distinguish between livestock impacts at intermediate levels of grazing activity. This finding is similar to that for ant communities in rangelands of the Great Basin, USA, where a number of ant community metrics could distinguish between rangelands in poor versus fair or good condition, but not between rangelands in fair and good condition (Nash et al., 2001). Sensitivity of an indicator to intermediate conditions is important for detecting unsustainable management conditions.

The use of ants for assessing and monitoring land degradation and restoration processes has been strongly advocated by Australian researchers (e.g., Majer, 1983; Andersen, 1990; Andersen et al., 2002). Ant communities are practical to sample for such purposes, and substantial effects of various management practices on ant communities have been demonstrated. Particularly prominent are the effects of open-cast mine restoration and sulfate deposition (e.g., Majer and Nichols, 1998; Andersen et al., 2002). For livestock grazing impacts, however, the utility of ants as indicators appears to be equivocal at best. In studies employing several experimental designs in the Chihuahuan Desert, Great Basin rangelands, and Mojave Desert of North America, ant assemblages either did not show consistent patterns as a function of degradation due to livestock grazing or the effects were evident primarily in conditions of extreme degradation rather than intermediate degradation (Whitford et al., 1999; Nash et al., 2000, 2001; this study). In semi-arid rangelands of the Northern Territory of Australia, ant species richness and abundance showed no overall effect along the grazing gradients, a finding that was consistent with three other studies of grazing within Australia (Hoffmann, 2000). Moreover, in an experiment using intense pulses of unsustainable grazing in South Australia, abundances of most of the common species and functional groups did not respond significantly to grazing (Read and Andersen, 2000). In contrast, Hoffmann (2000) found that individual ant species and functional groups responded to grazing in a manner reflective of declining rangeland conditions, indicating that these metrics could potentially serve as indicators of pasture and rangeland conditions. In sum, these studies suggest that rangeland ant communities in general are non responsive

to moderate grazing disturbance, whereas they appear to be much less tolerant of mine restoration and chemical pollution (e.g., sulfate). In contrast, other invertebrate taxa such as spiders, grasshoppers, beetles can be much more responsive to grazing impacts than ants (Hoffmann, 2000).

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References

- Andersen, A.N., 1990. The use of ant communities to evaluate change in Australian terrestrial ecosystems: a review and a recipe. *Proc. Ecol. Soc. Aust.* 16, 347–357.
- Andersen, A.N., 1991. Responses of ground-foraging ant communities to three experimental fire regimes in a savanna forest of tropical Australia. *Biotropica* 23, 575–585.
- Andersen, A.N., 1997. Functional groups and patterns of organization in North American ant communities: a comparison with Australia. *J. Biogeography* 24, 433–460.
- Andersen, A.N., Hoffmann, B.D., Muller, W.J., Griffiths, A.D., 2002. Using ants as bioindicators in land management: simplifying assessment of ant community responses. *J. Appl. Ecol.* 39, 8–17.
- Andersen, A.N., McKaige, M.E., 1987. Ant communities at Rotamah Island, Victoria, with particular reference to disturbance and *Rhytidoponera tasmaniensis*. *Proc. Royal Soc. Victoria* 99, 141–146.
- Andrew, M.H., Lange, R.T., 1986. Development of a new piosphere in arid chenopod shrubland grazed by sheep 1. Changes to the soil surface. *Aust. J. Ecol.* 11, 395–409.
- Ares, J., Bertiller, M., Bisigato, A., 2003. Modeling and measurements of structural changes at a landscape scale in dryland areas. *Environ. Model. Assess.* 8, 1–13.

- Avery, H.W., 1998. Nutritional ecology of the desert tortoise (*Gopherus agassizi*) in relation to cattle grazing in the Mojave Desert, Doctorate Dissertation. University of California, Los Angeles.
- Bahre, C.J., Shelton, M.L., 1993. Historic vegetation change. *J. Biogeography* 20, 489–504.
- Bernstein, R.A., Gobbel, M., 1979. Partitioning of space in communities of ants. *J. Anim. Ecol.* 48, 931–942.
- Buffington, L.C., Herbel, C.H., 1965. Vegetational changes on a semidesert grassland range from 1858 to 1963. *Ecol. Monogr.* 35, 139–164.
- Canfield, R.H., 1941. Application of the line interception method in sampling range vegetation. *J. Forestry* 39, 388–394.
- Chew, R.M., 1977. Some ecological characteristics of the ants of a desert-shrub community in south eastern Arizona. *Am. Midland Nat.* 98, 33–49.
- deSoyza, A.G., Whitford, W.G., Herrick, J.E., 1997. Sensitivity testing of indicators of ecosystem health. *Ecosyst. Health* 3, 44–53.
- Fore, L.S., Karr, J.R., Wisseman, R.W., 1996. Assessing invertebrates responses to human activities: evaluating alternative approaches. *J. North Am. Benthol. Soc.* 15, 212–231.
- Fusco, M.J., Holechek, J., Tembo, A., Daniel, A., Cardenas, M., 1995. Grazing influence on watering point vegetation in the Chihuahuan desert. *J. Range Manage.* 48, 32–38.
- Graetz, R.D., Ludwig, J.K., 1978. A method for the analysis of piosphere data applicable to range assessment. *Aust. Rangeland J.* 2, 126–136.
- Greenslade, P.J.M., Greenslade, P., 1971. The use of baits and preservatives in pitfall traps. *J. Aust. Entomol. Soc.* 10, 253–260.
- Harniss, R.O., West, N.E., 1973. Vegetation patterns of the National Reactor Testing Station, Southeastern Idaho. *Northwest Sci.* 47, 30–43.
- Hastings, J.R., Turner, R.M., 1965. *The Changing Mile: An Ecological Study of Vegetation Change with Time in the Lower Mile of an Arid and Semiarid Region.* University of Arizona Press, Tucson.
- Heske, E.J., Campbell, M., 1991. Effects of an 11-year livestock enclosure on rodent and ant numbers in the Chihuahuan Desert, southeastern Arizona. *Southwestern Nat.* 36, 89–93.
- Hess, K., Jr., 1992. *Visions Upon the land: Man and Nature on the Western Range.* Island Press, Washington, DC, USA, 278 pp.
- Hickman, J.C. (Ed.), 1993. *The Jepson Manual: Higher Plants of California,* University of California Press, Berkeley.
- Hoffmann, B.D., 2000. Changes in ant species composition and community organisation along grazing gradients in semi-arid rangelands of the Northern Territory. *Rangeland J.* 22, 171–189.
- Hölldobler, B., Wilson, E.O., 1990. *The ants.* Belknap Press, Cambridge, Mass.
- James, C.D., Landsberg, J., Morton, S.R., 1999. Provision of watering points in the Australian arid zone: a review of effects on biota. *J. Arid Environ.* 41, 87–121.
- Jennings, D.T., Houseweart, M.W., Francouer, A., 1986. Ants (Hymenoptera: Formicidae) associated with strip-clear-cut and dense spruce-fir forests of Maine Canadian. *Entomology* 118, 43–50.
- Jepson-Innes, K., Bock, C.E., 1989. Responses of grasshoppers (Orthoptera: Acrididae) to livestock grazing in southeastern Arizona: Differences between seasons and subfamilies. *Oecologia* 78, 430–431.
- Jones, K.B., 1981. Effects of grazing on lizard abundance and diversity in western Arizona. *Southwestern Nat.* 26, 107–115.
- Johnson, R.A., 1998. Foundress survival and brood production in the desert seed-harvest ants *Pogonomyrmex rugosus* and *P. barbatus* (Hymenoptera, Formicidae). *Oecologia* 45, 255–266.
- Karr, J.R., 1993. Defining and assessing ecological integrity: beyond water quality. *Environ. Toxicol. Chem.* 12, 1521–1531.
- Knick, S.T., Rotenberry, J.T., 1995. Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. *Conserv. Biol.* 9, 1059–1071.
- Lange, R.T., 1969. The piosphere: sheep track and dung patterns. *J. Range Manage.* 48, 396–400.
- Lobry de Bruyn, L.A., 1993. Ant composition and activity in naturally-vegetated and farmland environments on contrasting soils at Kellerberrin, Western Australia. *Soil Biol. Biochem.* 25, 1043–1056.
- Majer, J.D., 1983. Ants: bio-indicators of mine site rehabilitation, land-use, and land conservation. *Environ. Manage.* 7, 375–383.
- Majer, J.D., 1984. Recolonisation by ants in rehabilitated open cut mines in Northern Australia. *Reclam. Reveg. Res.* 2, 279–298.
- Majer, J.D., 1985. Recolonisation by ants of rehabilitated mineral sand mines on NorthStadbroke Island Queensland with particular reference to seed removal. *Aust. J. Ecol.* 10, 31–48.
- Majer, J.D., Day, J.E., Kabay, E.D., Perriman, W.A., 1984. Recolonisation by ants in bauxite mines rehabilitated by a number of different methods. *J. Appl. Ecol.* 10, 31–48.
- Majer, J.D., Nichols, O.G., 1998. Long-term recolonization patterns of ants in Western Australian rehabilitated bauxite mines with refernece to their use as indicators of restoration success. *J. Appl. Ecol.* 35, 161–182.
- McGeoch, M.A., 1998. The selection, testing and application of terrestrial insects as bioindicators. *Biol. Rev.* 73, 181–201.
- Marrs, R.H., Rizand, A., Harrison, A.F., 1989. The effects of removing sheep grazing on soil chemistry, above-ground nutrients distribution, and selected aspects of soil fertility in long-term experiments at Moore House National Nature Reserve. *J. Appl. Ecol.* 26, 647–661.
- Nash, M.S., Whitford, W.G., Vanzee, J., Havstad, K., 1998. Monitoring changes in stressed ecosystem using spatial patterns of ant communities. *Environ. Mon. Assess.* 51, 201–210.
- Nash, M.S., Whitford, W.G., deSoyza, A.G., Van Zee, J.W., Havstad, K.M., 1999. Livestock activity and Chihuahuan Desert annual-plant communities: Boundary analysis of disturbance gradients. *Ecol. Appl.* 9, 814–823.
- Nash, M.S., Whitford, W.G., Van Zee, J.W., Havstad, K.M., 2000. Ant (Hymenoptera: Formicidae) responses to environmental to stressors in the northern Chihuahuan Desert. *Environ. Entomol.* 29, 200–206.
- Nash, M.S., Whitford, W.G., Bradford, D.F., Franson, S.E., Neale, A.C., Heggem, D.T., 2001. Ant Communities and Livestock Grazing in the Great Basin, USA. *J. Arid Environ.* 49 (4), 695–710.

- Nash, M.S., Jackson, E., Whitford, W.G., 2003. Soil Microtopography on Grazing Gradients in Chihuahuan Desert Grasslands. *J. Arid Environ.* 55, 181–192.
- National Research Council, 1994. Rangeland Health: New Methods to Classify, Inventory, and Monitor Rangelands. National Academy Press, Washington, DC.
- Peck, S.L., McQuaid, B., Campbell, C.L., 1998. Using ant species (Hymenoptera: Formicidae) as a biological indicator of agroecosystem condition. *Environ. Entomol.* 27 (5), 1101–1110.
- Read, J.L., Andersen, A.N., 2000. The value of ants as early warning bioindicators: responses to pulsed cattle grazing at an Australian arid zone locality. *J. Arid Environ.* 45, 231–251.
- SAS, 1998. Stat User's Guide. SAS Inst. Inc. Cary, NC, USA.
- Sanders, N.J., Gordon, D.M., 2003. Resources-dependent interactions and the organization of desert ant communities. *Ecology* 84 (4), 1024–1031.
- Waser, N.M., Price, M.V., 1981. Effects of grazing on diversity of annual plants in the Sonoran Desert. *Oecologia* 50, 407–411.
- Whitford, W.G., 1978. Structure and seasonal activity of Chihuahuan Desert ant communities. *Insects Socioux* 25, 79–88.
- Whitford, W.G., 1995. Desertification: implications and limitations of the ecosystem health metaphor. In: Rapport, D.J., Gaudet C.L., Calow P. (Eds.), *Evaluating and Monitoring the Health of Large-Scale Ecosystems*, vol. 128. NATO ASI Series, Springer-Verlag, Berlin, pp. 273–293.
- Whitford, W.G., 1997. Desertification and animal biodiversity in the desert grasslands of North America. *J. Arid Environ.* 37, 709–720.
- Whitford, W.G. 2002. *Ecology of desert systems*, Academic Press, London.
- Whitford, W.G., Gentry, J.B., 1981. Ant communities of southeastern longleaf pine plantations. *Environ. Entomol.* 10, 183–185.
- Whitford, W.G., Van Zee, J., Nash, M.S., Smith, W.E., Herrick, J.E., 1999. Ants as indicators of exposure to environmental stressors in North American desert grasslands. *Environ. Monitor. Assess.* 54, 143–171.
- Wisdom, W.A., Whitford, W.G., 1981. Effects of vegetation change on ant communities of arid rangelands. *Environ. Entomol.* 10, 893–897.