



## Ant communities and livestock grazing in the Great Basin, U.S.A.

Maliha S. Nash\*, Walter G. Whitford\*†, David F. Bradford\*, Susan E. Franson‡, Anne C. Neale\* & Daniel T. Heggem\*

\*US EPA, P.O. Box 93478, Las Vegas NV 89193-3478, U.S.A.

†USDA-ARS, Jornada Experimental Range, MSC 3JER, New Mexico State University, Las Cruces NM, 88003, U.S.A.

‡US EPA, 26 W. Martin Luther King, Cincinnati OH 45268, U.S.A.

(Received 15 September 2000, accepted 2 April 2001)

The objectives of this study were to determine if metrics for ant species assemblages can be used as indicators of rangeland condition, and to determine the influence of vegetation and ground cover variables, factors often influenced by livestock grazing, on ant communities. The study was conducted in two areas in the Great Basin: a sagebrush-steppe in southeastern Idaho ( $n = 30$  sites), and a salt-desert shrub in western Utah ( $n = 27$  sites). Sites were selected based on known rangeland condition (i.e. good, fair, poor) associated with livestock grazing. Ant communities differed considerably between the two study areas. Collectively, more ant species occurred at the Idaho sites (30) than at the Utah sites (21), relatively few species (eight) occurred in both areas, species richness was significantly greater at the Idaho sites (mean = 12.0 species) than the Utah sites (mean = 6.9 species), and *Formica* spp. were diverse (total of 15 species) at the Idaho sites but rare (one species) at the Utah sites. In Idaho, all species collectively, generalists, and *Formica* spp. were significantly less abundant on sites in poor condition than that on sites in good or fair condition, whereas in Utah, seed harvesters and *Pheidole* spp. were significantly more abundant on sites in poor condition than that on sites in good or fair condition. In Idaho, species richness was significantly lower on sites in poor condition. In Idaho, species richness and relative abundances of several ant groups were significantly related to bare patch size and parameters for cover or species richness of several vegetation groups. In contrast to the comparisons involving sites in poor condition, no differences in ant communities in either Idaho or Utah were evident between sites in good and fair condition. Thus, the ant communities responded only to large changes in rangeland condition and to large differences in climatic/edaphic conditions between the two areas. Hence, ant community metrics appear to have limited utility as indicators of rangeland condition in the Great Basin.

© 2001 Academic Press

**Keywords:** indicator; livestock grazing; ant; vegetation; bare patches; Great Basin; range condition; cool desert

### Introduction

The prevalent land use in the western United States is commercial livestock grazing. Land use, combined with climate variability, has resulted in changes in vegetation



composition and cover over much of this region (Hastings & Turner, 1965; Buffington & Herbel, 1965; Harniss & West, 1973; Bahre & 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 been accompanied by changes in the structure of animal communities, but these are less well documented (Jones, 1981; Jepson-Innes & Bock, 1989; Heske & Campbell, 1991; Whitford, 1997).

There is a clear need to develop assessment and monitoring systems for the arid and semi-arid rangelands of North America (NRC, 1994). To this end, indicators are needed that are sensitive to disturbance (i.e. livestock grazing), and can be consistently applied across large areas. The NRC (1994) suggested the development of indicators based on various soil and vegetation properties.

Faunal indicators were not included among the indicators suggested by NRC (1994), yet many ecological processes (e.g. herbivory, soil development, pollination, seed dispersal) and much of rangeland biodiversity (e.g. species and trophic diversity) involve faunal components (McGeoch, 1998). Such processes and diversity components may not be reflected by the measurement of soil and vegetation metrics alone because the suggested vegetation indicators (NRC, 1994) reflect individual species primarily, rather than plant community composition. Moreover, faunal species composition and abundance at a site 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 & Rotenberry, 1995).

The present study explores the potential for ant species assemblages to be used as one of a suite of indicators of rangeland condition. We follow an approach to developing an indicator of biological integrity (Karr, 1991, 1993; Fore *et al.*, 1996). A well known example of this approach is the Index of Biotic Integrity (IBI) developed for river and stream fish assemblages (Karr, 1991). Parameters reflect species richness and composition, trophic composition, fish abundance, and individual condition. A key requirement for the IBI approach is the ability to quickly and accurately estimate relative abundance of each species in a relatively diverse species assemblage at each site sampled. A second key requirement is to establish relationships between metrics describing the species assemblage and the degree of impact on a system by examining the metrics along a gradient of impact (Karr, 1993; Fore *et al.*, 1996).

Ant communities have a number of attributes that may make them particularly useful as indicators of ecosystem change. They are 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. 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 & 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 & McKaige, 1987; Andersen, 1991), mining (Majer, 1984, 1985; Majer *et al.*, 1984, Majer & Nichols, 1998), forest clear-cutting (Jennings *et al.*, 1986; Whitford & Gentry, 1981), soil-vegetation changes resulting from over-grazing and drought (Wisdom & 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. Livestock grazing affects soil physical characteristics as well as the

better documented effects on vegetation (NRC, 1994). However, a recent study in Chihuahuan Desert rangelands, found no consistent differences in ant species composition and abundance that could be attributed to changes in vegetation associated with livestock grazing (Whitford *et al.*, 1999).

In order to identify general patterns of response to disturbance resulting from land management-climate interactions on ant communities, it is necessary to examine ant communities over a broad geographic region and a diversity of vegetation/soil type. Most studies of ants have focused on a single species or small group of species. Studies of ant communities are less numerous and most of these have considered a specialized segment of the whole community as the unit of study (Andersen, 1997). Particularly, harvester ant communities in arid regions have received considerable attention because of their effects on soil seed stores and on vegetation (Davidson, 1977; Brown *et al.*, 1979b; Rissing, 1988; Johnson, 2001).

Studies of whole ant communities in western North America have largely been limited to warm south-western deserts (Chew, 1977; Whitford, 1978; Bernstein & Gobbel, 1979; Andersen, 1997); there are few studies of ant communities in cool temperate deserts (Herbers, 1989). Consequently, we designed a study to examine the impact of livestock grazing on the species composition and diversity of ant communities in two cool desert areas within the Great Basin of the western United States. The objectives of this study were: (1) to identify and evaluate metrics of ant species assemblages for use as indicators of rangeland condition and (2) to determine the influence of vegetation and ground cover variables on ant communities. The latter topic was investigated to provide insight about the mechanisms by which livestock grazing may influence ant communities because vegetation characteristics are generally strongly affected by livestock grazing. These objectives were addressed by comparing ant community metrics as a function of three levels of rangeland condition associated with livestock grazing in two widely separated areas of the Great Basin and as a function of several parameters for vegetation and bare ground cover. We postulated that ant species richness would decline as a function of rangeland condition, and that the relative abundances of some individual taxa or functional groups of species would respond consistently positively or negatively to rangeland condition.

## Materials and methods

### *Study sites*

Studies were conducted in two areas: (1) a portion of the upper Snake River Plain in Idaho that includes both the Idaho National Engineering Laboratory (INEL) and the U.S. Sheep Station at Dubois ( $n = 30$  sites), and (2) a portion of western Utah that includes the U.S. Forest Service's Desert Experimental Range ( $n = 27$  sites; see map in Bradford *et al.*, 1998). These areas were chosen because they represent the northern and southern extremes of the Great Basin Desert scrub formation (Brown *et al.*, 1979a; Turner, 1982). These areas differ greatly in plant species composition and productivity. Both areas are cool deserts with warm summers and cold winters (Blaisdell & Holmgren, 1984). The Idaho area receives a mean annual rainfall of 280 mm and the Utah area receives a mean annual rainfall of 160 mm. The vegetation at the Idaho study sites is sagebrush-steppe dominated by various species of *Artemisia* with an understory of perennial grasses and forbs (Blaisdell *et al.*, 1982; MacMahon, 1985). The vegetation of the Utah study sites is salt-desert shrub, dominated by chenopod shrubs and dwarf shrubs, particularly *Atriplex* spp. This vegetation occurs in situations that promote soil salinity, soil alkalinity, or both. Saline and alkaline soils occur at the bottom of drainages in closed basins or where marine shales outcrop (West, 1982). Sites within these areas have well documented histories of grazing by domestic livestock, including sites with

heavy grazing and sites with limited or no grazing for several decades (Blaisdell *et al.*, 1982; Blaisdell & Holmgren, 1984). Grazing restrictions were implemented at INEL in 1950, at the Sheep Station in 1923, and at the Desert Experimental Range in 1934.

Study sites (Bradford *et al.*, 1998) were selected to represent different levels of impact due to grazing by domestic livestock based on consultation with local range scientists and the U.S. Bureau of Land Management range conservationists. The sites were classified as good, fair, and poor condition, corresponding to the Natural Resource Conservation Service's classification of rangeland condition as excellent/good, fair, and poor, respectively (USDA Soil Conservation Service, 1976). Such *a priori* classification was feasible in the study areas because grazing histories are exceptionally well known, and grazing has been closely regulated at many sites (Bradford *et al.*, 1998). In Idaho, additional sites were selected and classified as 'altered' because they had been modified for grazing purposes by reducing shrub cover (by mechanical means and/or burning) and seeding with crested wheatgrass (*Agropyron cristatus*). To facilitate detection of general patterns of differences in ant communities as a function of range condition, rather than patterns associated with specific vegetation or edaphic conditions, we selected sites in three widespread range types (used synonymously with 'range site', USDA Soil Conservation Service, 1976) in Idaho and three in Utah, with approximately equal representation of condition classes and altered sites among the range types. The predominant species in Idaho for the three range types were: (1) Wyoming big sagebrush (*Artemisia tridentata*) and bluebunch wheatgrass (*Agropyron spicatum*), (2) Wyoming big sagebrush and needle-and-thread grass (*Stipa comata*), and (3) threepart hybrid sagebrush (*A. tripartita*) and bluebunch wheatgrass (Bradford *et al.*, 1998). The predominant species in Utah for the three range types were (1) winterfat (*Krascheninikovia lanata*), budsage (*A. spinescens*), and shadscale (*A. confertifolia*), (2) winterfat and Indian ricegrass (*Oryzopsis hymenoides*), and (3) black sage (*A. nova*), squirreltail (*Elymus elymoides*), and needle-and-thread grass (Bradford *et al.*, 1998). Most sites were located within land management units of 8 km<sup>2</sup> or larger, but four sites were within units of 0.6 km<sup>2</sup>, and eight units were within units (primarily livestock enclosures) 0.02–0.17 km<sup>2</sup> in size. Elevation averaged 1455–1670 m for Idaho sites and 1500–1930 m for the Utah sites. Soils at the Idaho sites were predominately loams (sandy loams, silt loams, and sandy clay loams). Soils of the Utah sites were predominately sandy loam or sand. The number of sites per range condition class was not equal because of the availability of sites within each range condition. Number of sites for Idaho were 30 (good = 12, fair = 9, poor = 4, altered = 5). Number of sites for Utah, were 27 (good = 9, fair = 10, poor = 8).

### *Sampling and analyses*

Dependent variables included (1) ant species richness, and (2) the relative abundances of five functional groups, three abundant genera, and all species collectively. Independent variables were geographic area, rangeland condition class, percent cover and species richness for four vegetation classes, and bare patch size.

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, 1990). Pitfall traps (38 × 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 site. Each vial was filled to a depth of 30 mm with a mixture of 70% ethanol and 30% glycerol (Greenslade & 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. Days with measurable rainfall were excluded. Dates of sampling were 14–31 May 1994 in Utah, and 8–24 June 1994 in Idaho.

All ants in a vial were identified to species or to an operational taxonomic unit (Table 1). The relative abundance of each ant species (Whitford *et al.*, 1999) 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. Ranks were summed for each species for all traps in a site; the sum was divided by the total number of recovered traps. Each ant species was assigned to one of the following functional groups based on literature for the species or related species: generalists, seed harvesters, homopteran tenders, liquid feeders, predators, or slave makers.

Vegetation cover and plant species richness were measured at each study site by two uniformly spaced, 90-m transects (Canfield, 1941) adjacent to pit-fall trap grid lines. Plant species were grouped into four classes: perennial shrubs (PS), perennial grasses (PG), perennial forbs (PF), and total vegetation cover (including annuals; VG). Mean length of bare patches, i.e. of soil surface lacking vegetation cover, was also measured on the two transects.

Ant categories used in analyses included the functional groups listed above (except the relatively rare slave makers), all ant species collectively, and abundant genera (*Formica* spp., *Pogonomyrmex* spp., *Pheidole* spp., and *Myrmecocystus* spp.). To relate abundance to range condition, abundance data were analysed using analysis of variance. Because of the unbalanced design, we used General Linear Model (GLM) with least-square means options for multiple comparison of means (SAS, 1998). Regression analysis was used to examine relationships for ants (species richness and relative abundance) as a function of the vegetation and bare patch parameters. Regression models were linear and curvilinear (quadratic, square root, inverse, and inverse quadratic). The significance level for all statistical tests was 0.05. The five altered sites in Idaho were excluded from the statistical analyses.

## Results

The ant faunas differed considerably between the Idaho sagebrush steppe sites and the Utah salt-desert sites. A total of 30 and 21 ant species were found at the Idaho and Utah sites, respectively (Table 1). There was little overlap in species composition between the two areas. Of the 43 species collected, only eight were found in both areas, and only *Pogonomyrmex salinus* and *Monomorium minimum* were common in both areas. Site species richness was significantly greater in Idaho (mean = 12.0 species) than in Utah (mean = 6.9 species;  $F = 90.82$ , df. = 1,50,  $p < 0.0001$ ; Fig. 1). Fifteen species of *Formica* occurred at the Idaho sites, whereas only one occurred at the Utah sites (Table 1).

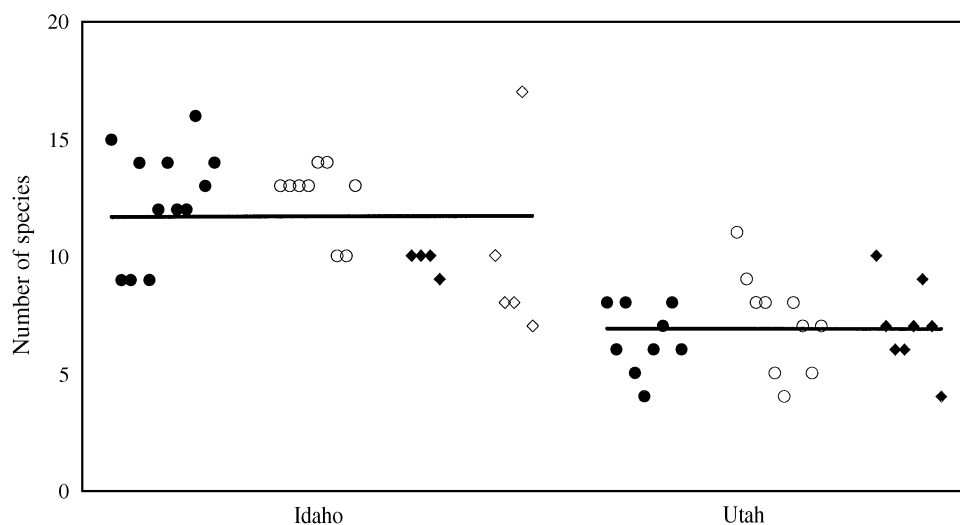
There was a significant relationship between ant species richness and range condition only for the Idaho sites ( $F = 3.37$ ; df. = 2,22;  $p = 0.05$ ; Fig. 1). In Idaho, species richness was higher for sites in good and fair condition than for sites in poor condition ( $t > 2$ ,  $p \leq 0.03$ ). Ant species richness was significantly related to vegetation cover and bare patch size at the Idaho sites, but not at the Utah sites (Table 2). The cover feature with the most significant effect on ant species richness in Idaho was mean bare patch size ( $R^2 = 0.42$ ). Other cover variables significantly related to ant species richness in Idaho were percent cover of total vegetation and perennial forbs ( $R^2 = 0.39$  and  $0.33$ , respectively; Table 2).

Ant species richness at the Idaho sites was significantly related to species richness of all four vegetation classes (Table 3). In individual regressions, species richness of perennial forbs and total vegetation each accounted for nearly half of the variation in ant species richness ( $R^2 = 0.47$  and  $0.45$ , respectively; Table 3). There were no significant relationships between species richness for ants and vegetation groups for Utah.

The relative abundances for some of the functional groups and abundant genera were significantly related to range condition. However, significant differences were evident

**Table 1.** Species names, associated functional groups, and percent of sites occupied by each ant species found at the Idaho (n = 30) and Utah (n = 27) sites

Genus	Species	Functional group	Percentage of sites	
			Idaho	Utah
<i>Camponotus</i>	<i>hyatti</i>	Generalist	3	—
<i>Camponotus</i>	<i>vicinus</i>	Generalist	87	4
<i>Conomyrma</i>	<i>insana</i>	Homopteran tender	7	59
<i>Formica</i>	<i>argentea</i>	Homopteran tender	33	—
<i>Formica</i>	<i>curiosa</i>	Homopteran tender	37	—
<i>Formica</i>	<i>haemorrhoidalis</i>	Homopteran tender	17	—
<i>Formica</i>	<i>laeviceps</i>	Homopteran tender	20	—
<i>Formica</i>	<i>neogagates</i>	Homopteran tender	83	—
<i>Formica</i>	<i>obscuripes</i>	Homopteran tender	53	15
<i>Formica</i>	<i>oreas</i>	Homopteran tender	37	—
<i>Formica</i>	<i>obtusopilosa</i>	Homopteran tender	23	—
<i>Formica</i>	<i>subnuda</i>	Homopteran tender	3	—
<i>Formica</i>	<i>subpolita</i>	Homopteran tender	27	—
<i>Formica</i>	S4	Homopteran tender	57	—
<i>Formica</i>	S10	Homopteran tender	10	—
<i>Formica</i>	S11	Homopteran tender	3	—
<i>Formica</i>	S12	Homopteran tender	3	—
<i>Formica</i>	S13	Homopteran tender	3	—
<i>Forelius</i>	<i>pruinosis</i>	Homopteran tender	13	85
<i>Lasius</i>	<i>crypticus</i>	Homopteran tender	93	—
<i>Leptothorax</i>	<i>furunculus</i>	Generalist	3	—
<i>Leptothorax</i>	<i>nevadensis</i>	Generalist	93	41
<i>Leptothorax</i>	<i>obliquicanthus</i>	Generalist	—	15
<i>Leptothorax</i>	<i>rugatulus</i>	Generalist	3	—
<i>Messor</i>	<i>smithi</i>	Seed harvester	—	4
<i>Monomorium</i>	<i>minimum</i>	Seed harvester	80	89
<i>Myrmica</i>	S1	Predator	43	—
<i>Myrmecocystus</i>	<i>flaviceps</i>	Liquid feeder	—	15
<i>Myrmecocystus</i>	<i>hammettensis</i>	Liquid feeder	—	78
<i>Myrmecocystus</i>	<i>mendax</i>	Liquid feeder	—	15
<i>Myrmecocystus</i>	<i>mexicanus</i>	Liquid feeder	—	33
<i>Myrmecocystus</i>	<i>navajo</i>	Liquid feeder	—	26
<i>Myrmecocystus</i>	<i>pyramicus</i>	Liquid feeder	—	7
<i>Pheidole</i>	<i>sitarches soritis</i>	Seed harvester	—	48
<i>Pheidole</i>	S28	Seed harvester	—	7
<i>Pheidole</i>	S29	Seed harvester	—	4
<i>Pogonomyrmex</i>	<i>magnacanthus</i>	Seed harvester	—	4
<i>Pogonomyrmex</i>	<i>occidentalis</i>	Seed harvester	—	30
<i>Pogonomyrmex</i>	<i>salinus</i>	Seed harvester	100	78
<i>Polyergus</i>	<i>breviceps</i>	Slave maker	13	—
<i>Solenopsis</i>	<i>molesta</i>	Generalist	83	33
<i>Tapinoma</i>	<i>sessile</i>	Generalist	87	—
<i>Tetramorium</i>	S1	Generalist	50	—



**Figure 1.** Ant species richness and site condition class in Idaho ( $n = 30$  sites) and Utah ( $n = 27$  sites). Site condition, Alt., is Altered (Idaho only). Horizontal lines indicate the overall mean (excluding the altered sites) in each study area. Good (●), fair (○), poor (◆), alt. (◇).

**Table 2.** Regressions of ant species richness ( $S$ ) and percent cover of vegetation and bare patch size for sites in Idaho (excludes altered sites;  $n = 25$ )

	$R^2$	$p > F$
$S = 11.07 + 0.37$ (PF)	0.33	0.003
$S = 7.72 + 0.16$ (VG)	0.39	0.0008
$S = 9.69 + 0.26/(BG)^2$	0.42	0.0005

Cover variables are percent cover of perennial forbs (PF), perennial shrubs (PS), perennial grass (PG), total vegetation (VG), and mean bare patch length (units = m; BG).

**Table 3.** Regressions of ant species richness ( $S$ ) and vegetation species richness for sites in Idaho (excludes altered sites;  $n = 25$ )

	$R^2$	$p > F$
$S = 9.89 + 0.24$ PF	0.47	0.0002
$S = 8.27 + 0.62$ PG	0.39	0.0009
$S = 9.25 + 0.41$ PS	0.16	0.047
$S = 8.80 + 0.14$ VG	0.45	0.0002

Vegetation variables are species richness for perennial forbs (PF), perennial grass (PG), perennial shrubs (PS), and all vegetation (VG).

in comparisons between good and fair condition versus poor condition (Table 4), but not between sites in good versus fair condition (Fig. 2). For ant groups showing significant relationships with range condition in Idaho (i.e. all species, generalists, and *Formica* spp. ), abundances were significantly higher at sites in good or fair condition than at sites in poor condition (Table 4; Fig. 2). In contrast to Idaho, abundances of the

**Table 4.** Values of *t* and its probability comparing relative abundances for functional groups and abundant genera on sites rated in good and fair versus poor condition in Idaho and Utah using GLM

Group	Idaho			Utah		
	Good vs. poor	<i>t</i>	<i>p</i>	Good vs. poor	<i>t</i>	<i>p</i>
All species	>	2.12	0.05*	<	1.64	0.12
Generalists	>	2.53	0.02*	<	0.44	0.66
Seed harvesters	<	0.39	0.70	<	2.12	0.04*
Homopteran tenders	>	1.89	0.07	<	1.21	0.24
Liquid feeders (all <i>Myrmecocystus</i> spp.)	—	—	—	>	1.79	0.09
Predators ( <i>Myrmica</i> sp.)	>	1.30	0.21	—	—	—
<i>Formica</i> spp.	>	0.04	0.97	<	1.11	0.28
<i>Pheidole</i> spp.	—	—	—	<	2.05	0.05*
<i>Pogonomyrmex</i> spp.	>	0.74	0.47	<	0.65	0.52
Group	Fair vs. poor	<i>t</i>	<i>p</i>	Fair vs. poor	<i>t</i>	<i>p</i>
All species	>	1.67	0.11	<	1.41	0.17
Generalists	>	2.01	0.06	=	0.004	1.00
Seed harvesters	<	1.04	0.31	<	2.28	0.03*
Homopteran tenders	>	1.70	0.10	<	0.30	0.77
Liquid feeders (all <i>Myrmecocystus</i> spp.)	—	—	—	>	0.77	0.45
Predators ( <i>Myrmica</i> sp.)	>	1.27	0.22	—	—	—
<i>Formica</i> spp.	>	2.06	0.05*	<	0.40	0.69
<i>Pheidole</i> spp.	—	—	—	>	1.47	0.16
<i>Pogonomyrmex</i> spp.	<	0.20	0.84	<	1.26	0.22

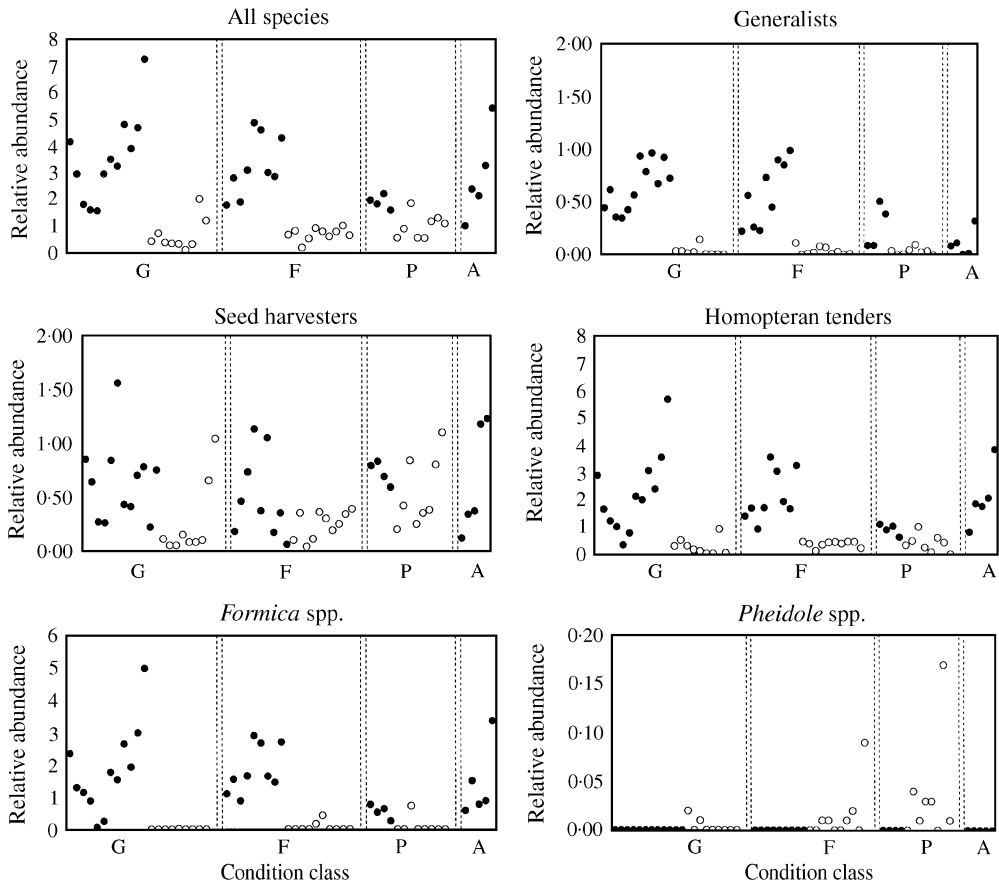
(Idaho, df. = 22; Utah df. = 24). Relative abundances that averaged greater on good/fair sites than poor sites are denoted by '>', the reverse by '<', and equal by '='. '\*' refers to significant at alpha = 0.05, and '-' refers to groups that were essentially absent from the area. Number of sites in Idaho per range condition was: good = 12, fair = 9, poor = 4. Number of sites in Utah per range condition were: good = 9, fair = 10, poor = 8.

two groups showing significant relationships with range condition in Utah (i.e. *Pheidole* spp. and seed harvesters) were significantly lower for sites in good or fair range condition than for sites in poor condition (Table 4; Fig. 2).

For the sites in altered condition in Idaho ( $n = 5$ ), relative abundances of two of the three ant groups showing significant relationships with range condition (i.e. all species and *Formica* spp.) were more similar to relative abundances at sites in good or fair condition than at sites in poor condition. In contrast, the relative abundances of generalists appeared to be more similar to abundances for sites in poor condition (Fig. 2).

Relationships between relative abundances and site condition were suggestive for several individual species (Fig. 3), but statistical tests were significant only for *Pheidole sitarches soritis*. This species was only observed in Utah and was significantly more abundant at sites in poor range condition than at sites in good condition ( $t = 2.48$ , df. = 24,  $p = 0.02$ ). In Idaho, three of the most abundant *Formica* spp. were absent or nearly absent from the poor range condition sites. In contrast, in Utah *Formica obtusopilosa* was more abundant on the poor range condition sites than on the good and fair condition sites. Relative abundance of *Pogonomyrmex salinus* showed no association

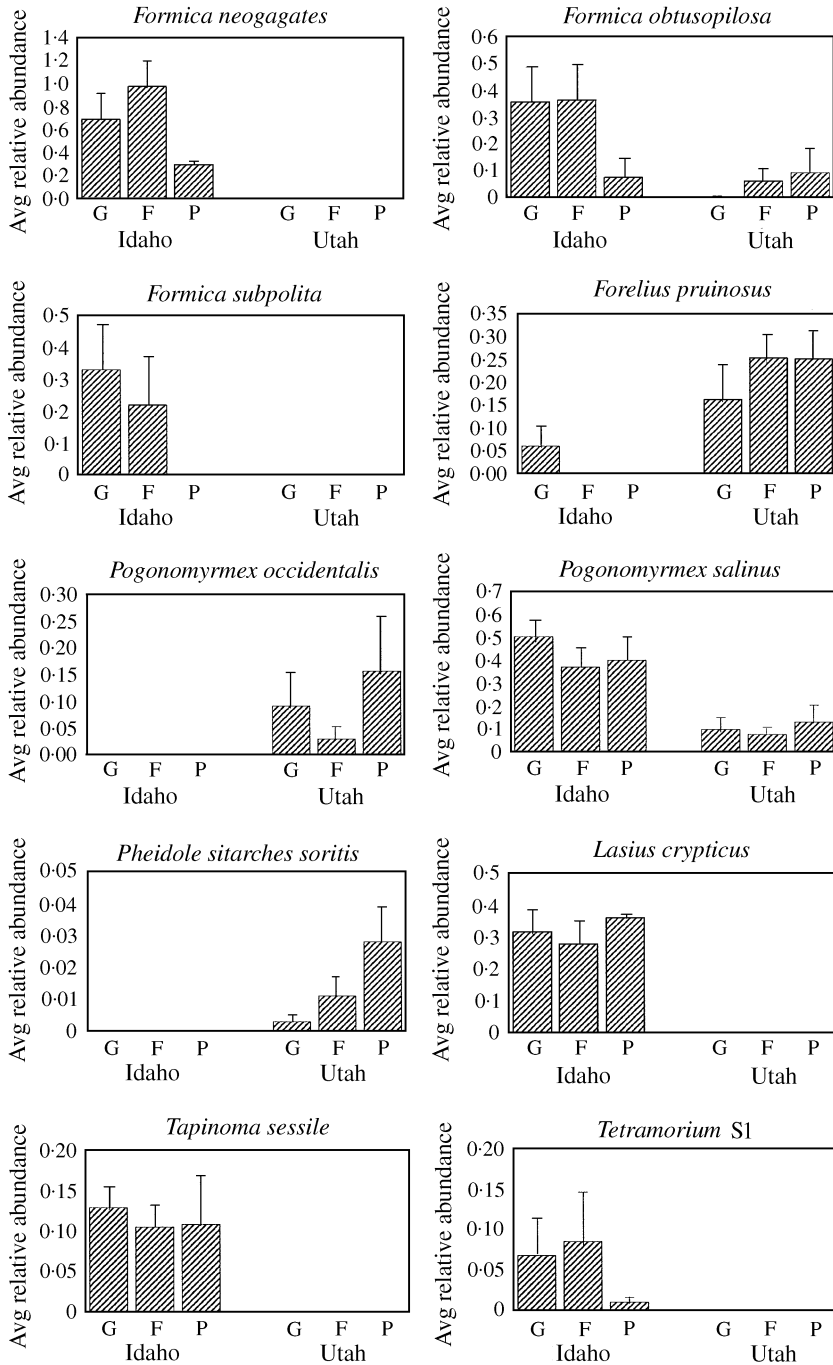




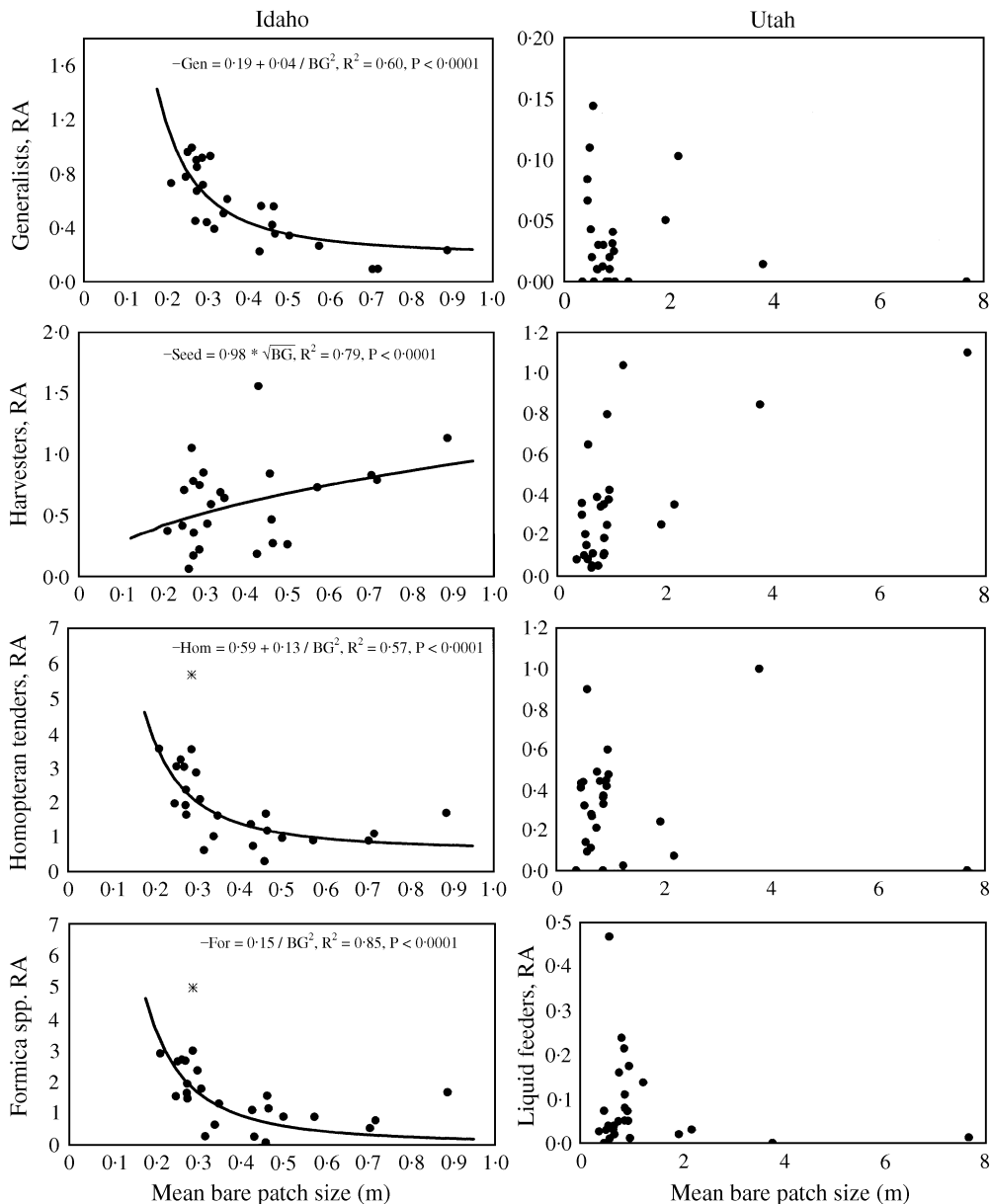
**Figure 2.** Relative abundances of six ant groups as a function of site condition class in Idaho (●) and Utah (○). Site condition classes are good (G), fair (F), poor (P), and altered (A). Sample size is 30 sites in Idaho (G = 12, F = 9, P = 4, A = 5) and 27 sites in Utah (G = 9, F = 10, P = 8).

with range condition in either Idaho or Utah, whereas abundance of *P. occidentalis* was generally higher on good and poor range condition sites than on fair range condition sites in Utah. *Forelius pruinus* in Idaho was recorded only on sites classified in good range condition. *Lasius crypticus* and *Tapinoma sessile* were only observed in Idaho and showed no pattern of abundance as a function of range condition. *Tetramorium* S1 was also present only in Idaho and its abundance was the lowest at sites in the poor range condition class.

Relative abundances of four ant groups (generalists, seed harvesters, homopteran tenders, and *Formica* spp.) were strongly related to bare patch size and vegetation cover for sites in Idaho (Fig. 4; Table 5), but not in Utah (Fig. 4). Of the cover variables, bare patch size had the most pronounced effect on the relative abundances of the four groups ( $R^2 = 0.57\text{--}0.85$ ; Fig. 4). Generalists, homopteran tenders, and *Formica* spp. decreased in number with increasing bare patch size, whereas the abundance of seed harvesters increased with increasing bare patch size (Fig. 4). Abundances of generalists, homopteran tenders, and *Formica* spp. were also significantly positively related to various parameters for vegetation cover ( $R^2 = 0.18\text{--}0.79$ ; Table 5), whereas the abundance of seed harvesters was negatively related to vegetation cover parameters ( $R^2 = 0.17\text{--}0.19$ ; Table 5).



**Figure 3.** Average relative abundances and 1 standard error (I) of selected abundant ant species as a function of site condition class in Idaho ( $n = 25$  sites) and Utah ( $n = 27$  sites). Site condition classes as in Fig. 2.



**Figure 4.** Relative abundance (RA) of several ant groups as a function of mean bare patch size in Idaho (excludes altered sites;  $n = 25$ ). Regression equations are given for the significant regressions. Regression models shown were the best fit of models tested (see text). Two regressions exclude one outlier, indicated by ‘\*’.

### Discussion

#### *Ants as indicators of rangeland condition*

The significant relationships between several ant community metrics and rangeland condition demonstrate that some ant metrics potentially could serve as indicators of rangeland degradation in the Great Basin. In Idaho, all species collectively, generalists,

**Table 5.** Regressions of relative abundances of ant functional groups and abundant genera as a function of percent cover of vegetation in Idaho (excludes altered sites; n = 25)

Model	R <sup>2</sup>	p > F
Gen = 0.29 + 0.20 (PF) <sup>0.5</sup>	0.44	0.0003
Gen = 0.26 + 0.02 (PS)	0.18	0.035
Gen = - 0.05 + 0.02 (VG)	0.46	0.0002
Seed = 0.96 - 0.02 (PS)	0.17	0.042
Seed = 1.11 - 0.02 (VG)	0.19	0.028
Hom = 1.04 + 0.03 (PG) <sup>2</sup>	0.43	0.0003
Form = 0.93 (PF) <sup>0.5**</sup>	0.79	< 0.0001
Form = - 0.108 + 0.06 (VG)**	0.32	0.004

\*\*n = 24; one observation was an outlier.

Regressions are given only for relationships with significant F values, and only the most significant regression model of the several tested for each dependent variable (see text) is presented. Ant groups are generalists (Gen), seed harvesters (Seed), homopteran tenders (Hom), and *Formica* spp. (Form). Cover variables are percent cover of perennial shrubs (PS), perennial forbs (PF), perennial grasses (PG) and total vegetation (VG).

and *Formica* spp. were significantly less abundant on sites in poor condition than that on sites in good or fair condition, whereas in Utah, seed harvesters and *Pheidole* spp. were significantly more abundant on sites in poor condition than that on sites in good or fair condition. Also, species richness in Idaho was significantly lower on sites in poor condition than on sites in good or fair condition. The abilities of these metrics to distinguish between extremes in condition represent a 'first cut' in identifying a potential indicator (Fore *et al.*, 1996). However, no differences were evident for any metric between sites in good versus fair condition. The ability of an indicator to make such a distinction is probably critical in rangeland systems because the condition threshold for irreversible degradation is thought to occur in fair or better conditions (Westoby *et al.*, 1989). Moreover, the ability of ant community metrics to distinguish only between extreme conditions may not improve upon the non-faunal indicators in use or recommended for investigation in rangeland systems (NRC, 1994). Thus, ant community metrics appear to have limited utility as indicators of rangeland condition in the Great Basin.

In comparison to other studies, an absence of differences in ant communities between sites in good versus fair rangeland condition was also observed by Crist & Wiens (1996) in a single-species study. In short-grass prairie systems, they reported overall lower densities of harvester ants (*P. occidentalis*) in heavily grazed pastures in comparison to light and moderately grazed pastures. They also reported the lowest variability in colony density in areas subjected to heavy grazing. The responses of several functional groups of ants to poor range condition in the present study and Crist & Wiens (1996) indicates that soil and vegetation changes resulting from intense grazing by livestock often affects ant communities. In contrast, in the Chihuahuan Desert of southern New Mexico and south-eastern Arizona, ant species richness and species composition exhibited no consistent patterns of response to disturbance even in the most degraded rangeland sites (Whitford *et al.*, 1999).

At the altered sites in Idaho, it is interesting that the striking changes in vegetation cover (i.e. reduced shrub cover and increased cover for a single grass species) did not appear to result in much change in ant community composition. That is, the relative

abundance for all ants, homopteran tenders, and *Formica* spp. on altered sites were generally similar to abundances on sites in good and fair condition. Nevertheless, management treatment of the altered sites did cause a significant reduction in the abundance of generalist species.

The general absence or weakness of relationships between ant community metrics and rangeland condition supports the conclusions of Whitford *et al.* (1999) that rangeland condition has relatively little effect on ant communities. In the present study, sites were assigned to rangeland condition classes based on the extent of vegetation change from the original state due to grazing management. The conceptual framework for assigning range condition in this manner was provided by Dyksterhuis (1949). Condition is based on the successional status of the vegetation of a site relative to an hypothesized climax for the unique soil and climate combinations at the site. The parameters included in this evaluation of 'condition' (vegetation change) appear to be less important as determinants of ant species richness than soil properties, landscape position, and structural characteristics of vegetation that affect soil microclimate (Crist & Wiens, 1996).

#### *Influence of vegetation and bare patch size on ant communities*

The significant relationships between relative abundances and rangeland condition for some taxa reflect either differences in colony densities or in extent and intensity of foraging from colonies (Crist & Wiens, 1996; Andersen, 1997). The *Formica* spp. are behaviorally dominant, cool-temperate species (Andersen, 1997). Sites in poor range condition are characterized by larger bare patches and sparser cover of shrubs and grasses than sites in good condition (de Soyza *et al.*, 1997; Nash *et al.*, 1999). One of the most dramatic effects of intense grazing by livestock is the increase in the sizes of bare patches of soil (areas devoid of vegetation cover). Open sites with large bare patches present ants with a temperature mosaic in which there are large areas of high soil surface temperatures interspersed with small, cool patches under shrubs. The large bare patches probably prevent ants like *Formica* spp. from traversing the patches during most of the daylight hours during the summer. A marked reduction in time available for foraging and storing food supplies for the long winter at these latitudes may eliminate colonies of cool-temperate ant species from sites in poor range condition. In contrast, the significant reduction in homopteran tenders in Idaho sites in poor condition is probably more a function of reduced primary production and low cover of vegetation that supports homopterans than of a soil surface microclimate. This probably also applies to the reduction in liquid feeding species in the Utah sites in poor condition. The negative relationship between bare patch size and relative abundance of *Formica* spp. and homopteran tending ants at the Idaho sites may also reflect differences in the quality and cover of the dominant shrubs as a substrate for homopterans and other insects.

The abundance of *Pogonomyrmex* seed harvesters was generally higher at the sites with large bare patches. Crist & Wiens (1996) reported that densities of *P. occidentalis* colonies were lowest under heavy grazing. Heavy grazing reduces vegetation cover and increases the average size of bare patches (de Soyza *et al.*, 1997). Seed harvesting ants select unvegetated patches for their nests, and clear vegetation from around the nest disk (Hölldobler & Wilson 1990). Thus, increased cover of bare soil and large size of bare patches may actually favor increases in densities of seed harvester ant colonies. Seed harvesters collect seeds of annual grasses, perennial grasses, and herbaceous plants. Annual grasses and herbaceous plants are generally not adversely affected by heavy grazing nor by large areas devoid of perennial vegetation (Nash *et al.*, 1999; Kelt & Valone, 1995). Therefore, rangeland in poor condition may produce as much food for seed harvesting ants as rangeland in excellent condition.

*Ant communities in Idaho versus Utah*

Ant communities differed considerably between sites in Idaho and Utah. Species richness was greater in Idaho, species overlap between the two areas was low, and relative abundances for several groups differed between the areas (e.g. generalists and homopteran tenders were relatively scarce at the Utah sites). The Utah sites also differed from the Idaho sites and from other areas reported in the literature by the absence of one or two clearly dominant species (Andersen, 1997). Some of these differences in species composition may be related to differences in the seasonal temperature characteristics of these locations. The ant genera that were most abundant at the Idaho sites (*Formica* and *Leptothorax*) are considered to be cold climate specialists, whereas the genera that dominated at the Utah sites are considered to belong to the hot-climate specialists and the generalized myrmicine functional groups by Andersen (1997). These differences correspond with the lower ambient temperatures during the growing season and the shorter growing season at the Idaho sites compared to the Utah sites.

The species richness-latitude relationship for the two regions represented in this study was the reverse of that found for European ant assemblages, where the number of species decreases linearly with increasing latitude (Cushman *et al.*, 1993). Species richness at the southern (Utah) sites was 58% of that in northern (Idaho) sites (6.9 *vs.* 12.0). The studies of variation in species richness with latitude cited by Cushman *et al.* (1993) were conducted on latitudinal gradients where the structural difference among habitats varies gradually along the gradient. In the present study, the latitudinal temperature differences apparently are not as important in determining species richness as are the rainfall and productivity differences. The mean annual rainfall at the Idaho sites is 280 mm in contrast to only 160 mm at the Utah sites. These differences in rainfall are associated with generally lower productivity of the Utah sites (Blaisdell & Holmgren, 1984). The saline soils and specialized vegetation communities that develop on soils at the Utah sites also contribute to the lower productivity of those sites in comparison to the Idaho sites. Lower productivity reduces the food base for ants. Thus, the large differences in the ant communities between the Idaho and Utah sites are probably a function of both climatic and edaphic differences between these areas.

Ants were identified by Walter Smith, Justin Van Zee, and Gregory Forbes of the USDA-ARS Jornada Experimental Range, which also maintains the reference collections from this study. Field assistance was provided by Randall Balice, James Glennon, Kathy Hickey, William Mader, Cynthia and Michael O'Hara, and G. Renshaw employed by Lockheed Environmental Systems & Technology Co. (LESAT). Representatives of the US Forest Service, US Bureau of Land Management, and USDA Agricultural Research Service provided substantial assistance in site selection. Staffs of the Desert Experimental Range, Idaho National Engineering Laboratory, and U.S. Sheep Station also provided assistance in the use of their facilities, and LESAT assisted the project in many ways. Robert P. Breckenridge, William G. Kepner, and Glen R. Miller were particularly helpful throughout the project. The project plan benefitted from external review by Harry Coulombe, James R. Karr, Douglas H. Johnson, and William P. MacKay. The U.S. Environmental Protection Agency (EPA), through its Office of Research and Development, funded the research described here. It has been subjected to the Agency's review and approved for publication.

### References

- Andersen, A.N. (1990). The use of ant communities to evaluate change in Australian terrestrial ecosystems: a review and a recipe. *Proceedings of Ecological Society Australia*, **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.

- Herbers, J.M. (1989). Community structure in north temperate ants: temporal and spatial variation. *Oecologia*, **81**: 201–211.
- 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 Naturalist*, **36**: 89–93.
- Hölldobler, B. & Wilson, E.O. (1990). *The Ants*. Cambridge, Mass: Belknap Press.
- James, C.D., Landsberg, J. & Morton, S.R. (1999). Provision of watering points in the Australian arid zone: a review of effects on biota. *Journal of Arid Environments*, **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.
- Johnson, R.A. (1998). Foundress survival and brood production in the desert seed-harvester ants *Pogonomyrmex rugosus* and *P. barbatus* (Hymenoptera, Formicidae). *Oecologia*, **45**: 255–266.
- Johnson, R.A. (2001). Biogeographical and community structure of North American seed-harvester ants. *Annual Review of Entomology*, **46**: 1–30.
- Jones, K.B. (1981). Effects of grazing on lizard abundance and diversity in western Arizona. *Southwestern Naturalist*, **26**: 107–115.
- Karr, J.R. (1991). Biological integrity: a long-neglected aspect of water resources management. *Ecological Applications*, **1**: 66–84.
- Karr, J.R. (1993). Defining and assessing ecological integrity: beyond water quality. *Environmental Toxicology and Chemistry*, **12**: 1521–1531.
- Kelt, D.A. & Valone, T.J. (1995). Effects of grazing on the abundance and diversity of annual plants in Chihuahuan Desert scrub habitat. *Oecologia*, **103**: 191–195.
- Knick, S.T. & Rotenberry, J.T. (1995). Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. *Conservation Biology*, **9**: 1059–1071.
- Lobry de Bruyn, L.A. (1993). Ant composition and activity in naturally-vegetated and farmland environments on contrasting soils at Kellerberrin, *Western Australia*. *Soil Biology and Biochemistry*, **25**: 1043–1056.
- McGeoch, M.A. (1998). The selection, testing and application of terrestrial insects as bioindicators. *Biological Reviews*, **73**: 181–201.
- MacMahon, J.A. (1985). *Deserts*. New York: Alfred A. Knopf 638 pp.
- Majer, J.D. (1984). Recolonisation by ants in rehabilitated open cut mines in Northern Australia. *Reclamation and Revegetation Research*, **2**: 279–298.
- Majer, J.D. (1985). Recolonisation by ants of rehabilitated mineral sand mines on North Stadbroke Island Queensland with particular reference to seed removal. *Australian Journal of Ecology*, **10**: 31–48.
- Majer J.D. & Nichols, O.G. (1998). Long-term recolonization patterns of ants in Western Australian rehabilitated bauxite mines with reference to their use as indicators of restoration success. *Journal of Applied Ecology*, **35**: 161–182.
- 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. *Journal of Applied Ecology*, **10**: 31–48.
- 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. *Ecological Applications*, **9**: 814–823.
- NRC (National Research Council) (1994). Rangeland health: new methods to classify, inventory, and monitor rangelands. National Academy Press. Washington, D.C.
- Peck, S.L., McQuaid, B. & Campbell, C.L. (1998). Using ant species (Hymenoptera: Formicidae) as a biological indicator of agroecosystem condition. *Environmental Entomology*, **27**: 1101–1110.
- Rissing, S.W. (1988). Dietary similarity and foraging range of two seed-harvester ants during resource fluctuations. *Oecologia*, **75**: 362–366.
- SAS. (1998). Stat User's Guide. SAS Institute. Inc. Cary, NC, USA.
- Turner, R.M. (1982). Great Basin desert scrub. In: Brown, D.E. (Ed.), *Biotic Communities of the American Southwest—United States and Mexico*, pp. 145–155. Desert Plants vol. 4 Numbers 1–4, pp. 342.

- USDA Soil Conservation Service. (1976). National Range Handbook, U.S. Department of Agriculture, Washington, D.C.
- West, N.E. (1982). *Intermountain Salt-desert Shrublands Ecosystems of the World*, Vol. 5. pp. 375–398. Amsterdam Elsevier Scientific.
- Westoby, M., Walker, B. & Noy-Meir, I. (1989). Opportunistic management for rangelands not at equilibrium. *Journal of Range Management*, **42**: 266–274.
- Whitford, W.G. (1978). Structure and seasonal activity of Chihuahuan Desert ant communities. *Insects Sociaux*, **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), pp. 273–293. *Evaluating and Monitoring the Health of Large-scale Ecosystems*. NATO ASI Series, Vol. 128, Berlin: Springer-Verlag 454 pp.
- Whitford, W.G. (1997). Desertification and animal biodiversity in the desert grasslands of North America. *Journal of Arid Environments*, **37**: 709–720.
- Whitford, W.G. & Gentry, J.B. (1981). Ant communities of southeastern longleaf pine plantations. *Environmental Entomology*, **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. *Environmental Monitoring and Assessment*, **54**: 143–171.
- Wisdom, W.A. & Whitford, W.G. (1981). Effects of vegetation change on ant communities of arid rangelands. *Environmental Entomology*, **10**: 893–897.



## AUTHOR QUERY FORM

**HARCOURT  
PUBLISHERS**

JOURNAL TITLE: JARE  
ARTICLE NO. : 20010824

DATE:3/9/2001

### *Queries and / or remarks*

Manuscript Page/line	Details required	Author's response
24	Turner 1982 — supply name and location of publisher.	
24	West 1982 — supply total number of pages.  Pl. check the running head. Is it OK?	