# VEGETATION, SOIL, AND ANIMAL INDICATORS OF RANGELAND HEALTH

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**Abstract.** We studied indicators of rangeland health on benchmark sites with long, well documented records of protection from stress by domestic livestock or histories of environmental stress and vegetation change. We measured ecosystem properties (metrics) that were clearly linked to ecosystem processes. We focused on conservation of soil and water as key processes in healthy ecosystems, and on maintenance of biodiversity and productivity as important functions of healthy ecosystems. Measurements from which indicators of rangeland health were derived included: sizes of unvegetated patches, cover and species composition of perennial grasses, cover and species composition of shrubs and herbaceous perennials, soil slaking, and abundance and species composition of the bird fauna. Indicators that provided an interpretable range of values over the gradient from irreversibly degraded sites to healthy sites included: bare patch index, cover of long-lived grasses, palatability index, and weighted soil surface stability index. Indicators for which values above a threshold may serve as an indicator of rangeland health include: cover of plant species toxic to livestock, cover of exotic species, and cover of increaser species. Several other indicator metrics were judged not sensitive nor interpretable. Examples of application of rangeland health indicators to evaluate the success of various restoration efforts supported the contention that a suite of indicators are required to assess rangeland health. Bird species diversity and ant species diversity were not related to the status of the sample site and were judged inadequate as indicators of maintenance of biodiversity.

## 1. Introduction

There is a clear need to develop assessment and monitoring systems for the rangelands of North America (National Research Council, 1994). Rangeland health assessments are needed to evaluate the status of the component ecosystems of the rangelands. Monitoring systems are needed to detect changes that are related to the probability of degradation (i.e. risk of crossing a threshold to an alternate but less desirable state (Westoby *et al.* 1989). Monitoring is also needed to track the success or failure of restoration efforts. For both assessment and monitoring, it is necessary to have a suite of indicators that are sensitive to environmental stress, that focus on risk of degradation, and that are related to ecosystem function (Herrick *et al.* 1995).

"Rangeland health may be defined as the degree to which the integrity of the soil and the ecological processes of rangeland ecosystems are sustained" (National Research Council 1994). While this definition is useful, the concept of integrity is difficult to quantify. Examination of other descriptions of ecosystem health (where ecosystems are broadly defined to include humans and all of their activities) reveals a common set of characteristics of healthy ecosystems: (1) they are free from ecosystem distress syndrome

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(a common set of signs that characterize the most heavily damaged ecosystems (Rapport *et al.* 1985), (2) they are resilient (rebound quickly to pre-disturbance state) following normal perturbations (i.e. disturbances normally encountered in their evolutionary history such as fire, floods, and drought), (3) they are self-sustaining (i.e. they can be perpetuated without subsidies or drawing down natural capital), (4) management practices and ecosystem processes do not impair adjacent systems or systems at some distance, (5) they are economically viable and (6) they sustain healthy human communities. For rangeland health the indicators must provide information which can be used to evaluate the capacity of the system to conserve and efficiently use water and nutrients, and to support biodiversity, economic production, and recreational uses.

The indicators of rangeland ecosystem health must meet criteria similar to those applied to judgments of which tests should be done in an assessment of human health. Indicators of rangeland health should (1) reflect the status of a critical ecosystem process, important ecosystem property, or an economic-social value, (2) be unambiguous (i.e., the trajectory of the measure is unidirectional in response to ecosystem stressors of increasing intensity), (3) be applicable in the range of ecosystems encountered in rangeland landscapes, and (4) be readily and inexpensively measured. There are ecosystem properties that provide information on the characteristics of most ecosystem processes. Ecosystem properties such as structural characteristics of the vegetation, spatial distribution patterns of plant species, morphological characteristics of plants, and physical and biological characteristics of the soil surface do not change rapidly over time and the quantitative values of measurements of these properties are frequently directly related to one or more ecosystem processes. The properties chosen as indicators in our studies met the criteria listed above and there was sufficient information in the literature to clearly link indicators and ecosystem processes.

Potential indicators need to be evaluated for their sensitivity to disturbance and stress. Disturbance is used here as a variable to which the ecosystem has been exposed over evolutionary time while a stress is a variable to which the ecosystem has had no evolutionary experience. In rangelands in the western U.S., drought is a disturbance and poorly managed grazing by domestic livestock is a stress. A sensitive indicator is one that yields very different quantitative values when measured at locations that are known to be degraded and at locations that are known to be in healthy condition (deSoyza *et al.* 1997). A sensitive indicator also yields very different quantitative values when measured at locations known to be exposed to very different levels of environmental stress. If the indicator is sufficiently sensitive, it should be possible to use that same indicator to evaluate the relative success of restoration efforts. Our studies were designed to test the sensitivity of a series of indicators derived from measurements of ecosystem properties for use in assessing rangeland health and in evaluating the health status of restoration efforts. For each indicator we hypothesized the response to stress or change in state (Table I).

Table I
Hypothesized responses of indicator metrics to exposure to environmental stress or ecosystem change resulting from exposure to stress and as measures of recovery resulting from restoration efforts.

| Indicator | Exposure to Stress/Change | Restoration Efforts |
|---|---|---|
| **Water and Nutrient Conservation** | | |
| percent cover long-lived grasses | < | > |
| bare patch size (bare patch index) | > | < |
| percent cover vegetation | < | > |
| soil surface stability - slake tests | < | > |
| percent cover "increasers" | > | < |
| percent cover shrubs | > | < |
| **Productivity (commodity yield)** | | |
| percent cover species preferred by livestock | < | > |
| percent cover toxic species | < | > |
| percent cover shrubs | > | < |
| percent cover exotics | ? | ? |
| **Biodiversity** | | |
| perennial plant species richness | < | > |
| perennial plant species diversity (H') | < | > |
| breeding bird species richness | < | > |
| breeding bird species diversity (H') | < | > |

## 2. Methods

The study sites were located in the Jornada Basin including the slopes of the San Andres mountains and Dona Ana mountains and the Jornada plain on the USDA-ARS Jornada Experimental Range approximately 40 km north of Las Cruces, N.M. Measurements were made on 44 sites. Seven sites were selected on the basis of historical records of vegetation change and land use (Buffington and Herbel 1965, Gibbens and Beck 1988, Gibbens *et al.* 1992). Four sites that had been subjected to different restoration practices in the mid to late 1970's (Herbel *et al.* 1958, 1973, 1983, 1985) were paired with sites that were grazed and also with plots within grazing exclosures. We included plots established at varying distances from stock watering points. These provide a distinct disturbance gradient from very intense at the water point to no perceptible disturbance at distances greater than 1 km

(Andrew and Lange 1986, Fusco *et al.* 1995). We made measurements outside and within three 50 year old grazing exclosures. Measurements were also made on four sites in southeastern Arizona.

At each site, we established a 1 ha plot with the baseline centered on a randomly selected point. Vegetation and soils were measured along ten, 100 m lines spaced at 10 m intervals along the baseline. At sites with a perceptible slope, the sample lines were oriented with the slope. On the flat areas, the transect lines were parallel to the disturbance gradient (if present). We recorded the intercept length of canopy cover by species and the length of bare (unvegetated) patches (Canfield 1941).

A slake test (Tongway 1994) modified to account for differences in wet aggregate strength, was used to measure soil crust stability. Soil stability was measured from three different strata (bare soil, grass clump, and under shrub canopy). At each site we selected three transects at random and for each transect generated 25 random numbers between 0 and 100. We sampled the soil at each point corresponding to a random number until we had obtained three samples for each stratum. If the soil at a point did not yield a soil fragment (i.e. the soil disintegrated), it was assigned a value of 1.0. If fewer than 3 samples were obtained for a stratum, an additional 25 random numbers were generated and the transect was repeated for that stratum only. Soil was tested from surface (0 - 3 mm depth). The slake values provide a measure of current stability. The soil stability test was done only on air dried soils because moist soil tends to yield an overestimate of stability. Stability was measured on soil fragments (6mm - 8mm in diameter) that were 2 - 3 mm thick. Soil fragments were carefully placed in a small, 25 mm diameter PVC basket with a wire mesh bottom (1.5 mm mesh size). The baskets were slowly lowered into a reservoir of distilled water. The disintegration of each fragment was observed for 5 minutes. If the soil fragment remained intact at the end of this time, the basket was raised and lowered three times. Soil stability was ranked according to the time required for the fragment to disintegrate during the five minute immersion or on the proportion of the fragment remaining intact after three immersion cycles.

Breeding birds were censused in areas centered on sites where detailed vegetation and soil indicator measurements were made. Breeding birds were censused in early June, the peak of the breeding season in the Chihuahuan Desert. Nine circular plots of 50 m radius were established in good condition grassland, a mixed grass - mesquite shrub mosiac, creosotebush shrubland, tarbush shrubland, and mesquite coppice dune area. Based on historical records, these areas provided a gradient from minimally changed (healthy rangeland) to maximally degraded (changed) rangeland. Breeding birds were also censused in southeastern Arizona on landscape units dominated by Lehmann lovegrass and compared with censuses conducted on native grasslands. The center of each circular plot was 300 m from the center of all other plots. Plots were arranged in a 3 x 3 grid. Birds that were seen or heard singing within the 50 m circle were recorded by two observers standing at the center point. Observations were made for 10 minutes at each center point.

Indicator metrics applicable to assessment of rangeland health were plotted against an axis of percent grass cover. All of the sites which were selected for these measurements were in areas on the Jornada Basin that were classified as grassland in the original surveys and which have been documented to have undergone some change during the past 150 years (Buffington and Herbel 1965). By selecting grass cover as the independent variable, we provide a measure of departure from the least disturbed (least changed) landscape units. If the indicator metrics that we have selected are sensitive measures of the health of these rangeland ecosystems, they should vary in a systematic way on a gradient of change measured as percent grass cover.

## 3. Results

Several of the hypothesized indicators failed to change in a systematic fashion on a gradient from most changed to least changed ecosystem. Although there was a cluster of sites with low percent of vegetative cover on the maximally changed end of the gradient, there were 5 sites that had vegetative cover equivalent to sites in the intermediate change category (Figure 1). Total vegetative cover is therefore not a sensitive indicator of exposure to stress. Two other indicators that produced patterns that were not consistent with the hypothesized responses and that were not easily interpreted were perennial plant species richness and Shannon's diversity index for perennial plant species (Figures 2 and 3). Percent shrub cover was hypothesized to increase as sites were exposed to increased stress since the historical trend in the degradation (desertification) of Chihuahuan Desert rangelands has been "invasion" by woody shrubs (Buffington and Herbel 1965, Grover and Musick 1990). Shrub cover varied from zero to twenty percent on degraded sites and one of the "healthy" sites had shrub cover of fifteen percent (Figure 4). Thus shrub cover is not a sensitive indicator for incorporation into a rangeland health assessment system for Chihuahuan Desert rangelands.

An indicator based on the size of unvegetated patches that is identified as a bare patch index was calculated as: bare patch index = mean bare patch size x proportion of line that is bare. The bare patch index provided one of the most sensitive metrics for assessing change (Figure 5). The inflection point in the bare patch index at a value of 80 may represent the threshold value for this metric (Figure 5). A vegetation metric that was very sensitive for assessing change was percent cover of long-lived grasses. These grasses are species that are very drought resistant and that live for several decades. The percent cover of these grasses decreased to zero or virtually zero on the maximally changed sites (Figure 6). The surface soil stability index was calculated from the slake test data by multiplying the mean slake value for grass, shrub, and bare soil by the cover of each of those types to obtain a weighted mean for each of the three sub-locations sampled. The three weighted values were summed to obtain the surface soil stability index for the site. This index provided a clearly interpretable pattern that was consistent with the hypothesized response (Figure 7). The surface soil stability index combined with the bare patch index provide

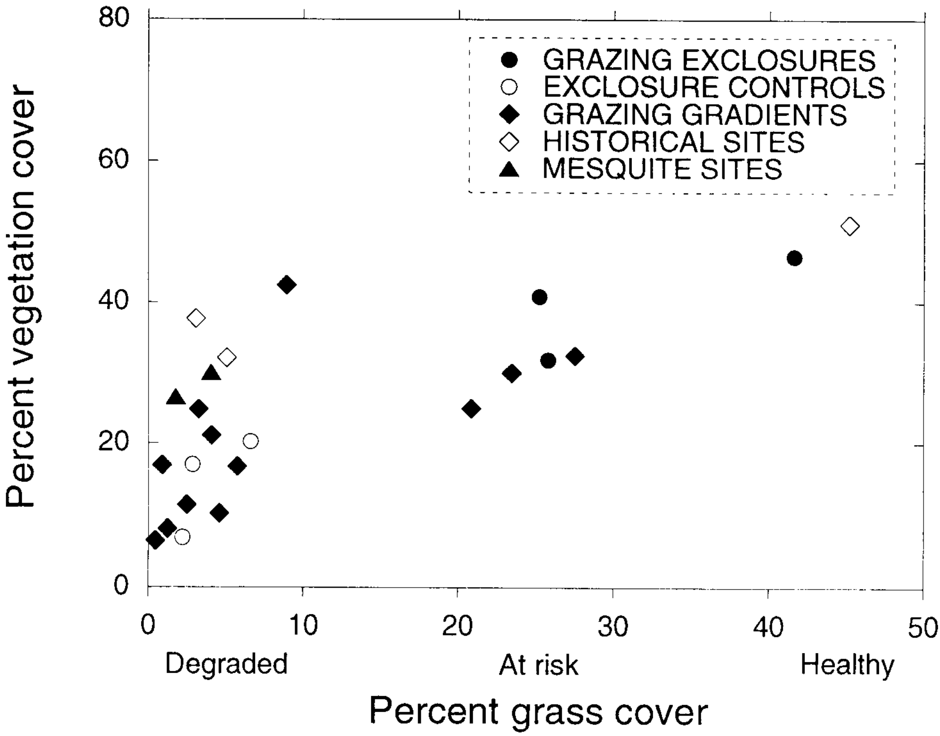


*Fig. 1.* Average total vegetative cover at sites ranging from irreversibly degraded to healthy.

metrics that are related to erosion potential of a site.

The indicator that provides information on the productive capacity of the system, the index of relative preference of vegetation by livestock, incorporates data on cover by species and the relative palatability of that species to livestock. The relative preference index is calculated as: percent cover (by species) x forage quality (good = 1.0, fair = 0.75, poor = 0.25, toxic or not eaten = 0.0) x fraction of the year the species is eaten by cattle (primary data source, Stubbendieck *et al.* 1993). The relative preference index is a very sensitive indicator of one function related to the health of rangeland ecosystems (Figure 8). Other metrics that can be derived from the basic data set include: cover of toxic species, cover of exotic (non-native) species and cover of invader species, i.e. those species that rapidly occupy sites outside their pre-stress habitats (examples are mesquite, Prosopis glandulosa and creosotebush, Larrea tridentata)(Gardiner 1951, Buffington and Herbel 1965, Grover and Musick 1990, Gibbens *et al.* 1992, Dick-Peddie 1993). All of these metrics are related to the productivity of rangelands. The percent cover of perennial plant species that are toxic to livestock was similar on many of the degraded, at risk, and healthy sites (Figure 9). The only sites where cover of toxics exceeded 5% were sites currently exposed to activity of large numbers of livestock for a large part of each year (Figure 9).

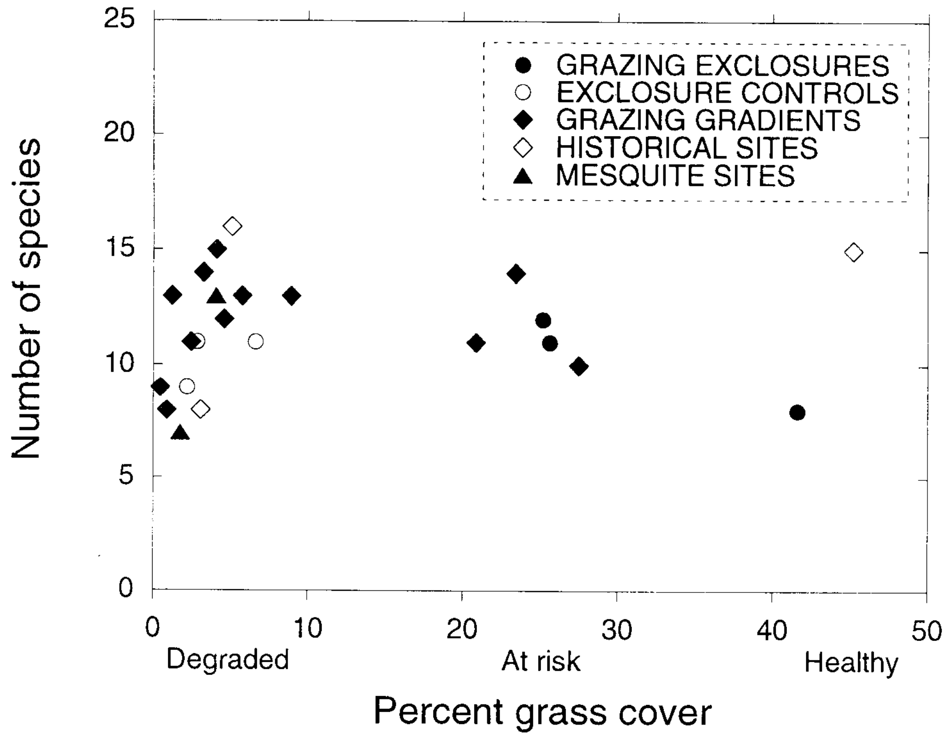


*Fig. 2.* Perennial plant species richness at sites ranging from irreversibly degraded to healthy.

A measurement with these characteristics may be used as an indicator of exposure to grazing stress but not as an indicator of desertification. The distribution of values of cover for toxic species suggests that cover of toxics exceeding a 5% cover threshold may be considered as an indicator of degradation with the caution that values below 5% should not be considered as an indicator of a healthy site. Here we identify the threshold as values above those recorded on sites at the healthy end of the spectrum.

The percent cover of increasers (species that establish in areas not occupied before disturbance and increase in density and cover following establishment) was higher at most of the degraded sites but there were degraded sites where cover of increasers was less than 5% (Figure 10). This is another indicator where values above a threshold, i.e., cover of 10%, may be a useful indicator of degradation in a multi-metric assessment. However cover of increasers less than 10% must not be judged as an indicator of a healthy rangeland site. None of the sites in the Jornada Basin had significant cover of exotics with the exception of one of the restoration site controls. We have data on exotics from only two sites in southeastern Arizona where there was documentation of recent expansion of the

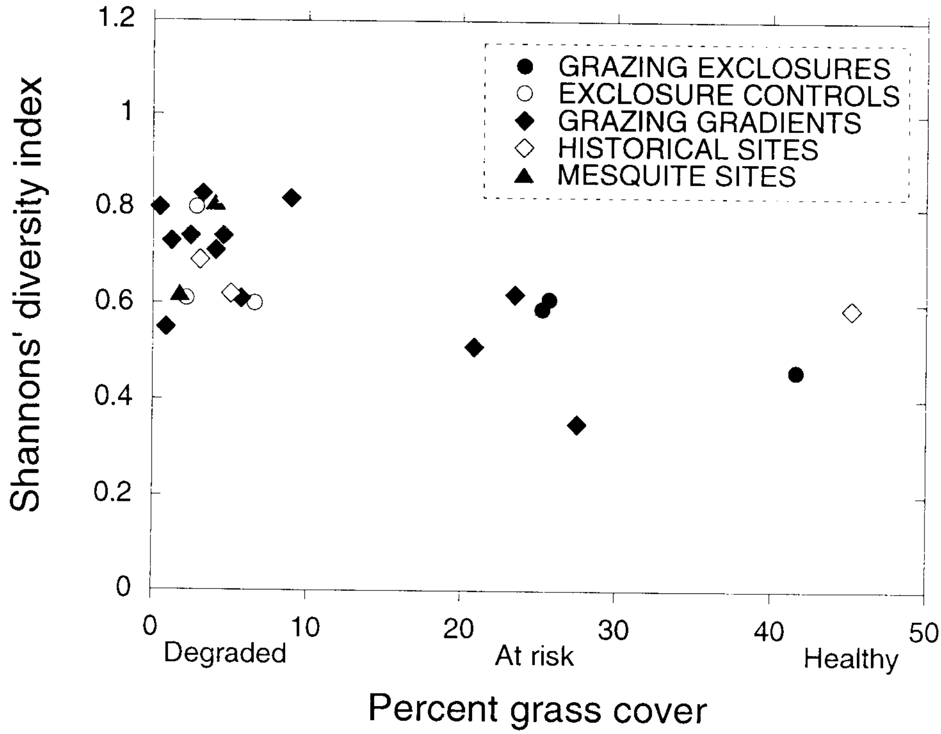


*Fig. 3.* Shannon's species diversity index (H') at sites ranging from irreversibly degraded to health.

South African Lehmann's lovegrass, *Eragrostis lehmanniana* (Anable *et al.* 1992). At these sites E. lehmanniana accounted for 88-97 % of the vegetative cover. On native grassland sites in the same areas, grass cover was 10.4-19.1% and Lehmann lovegrass accounted for less than 5% of that cover. Percent cover of exotic species is an indicator like the cover of increasers and cover of toxics where a cover value above some threshold may be considered an indicator of degradation.

## INDICATORS OF RESTORATION SUCCESS

The applicability of rangeland health indicators as measures of restoration success was examined on a series of paired sites in mesquite coppice dunes and creosotebush-tarbush shrub sites. Restoration efforts in the Chihuahuan Desert have focused on reducing or eliminating shrubs. These efforts have included treatment with chemical herbicides and mechanical treatments such as root plowing and bulldozing (Herbel *et al.* 1973, Herbel *et al.* 1983, Herbel *et al.* 1985). All of the restoration treatments had been in place for more than 25 years when the indicator measurements were made. In the mesquite coppice dunes, the bare patch index was greatly reduced by bulldozing that flattened the dunes and

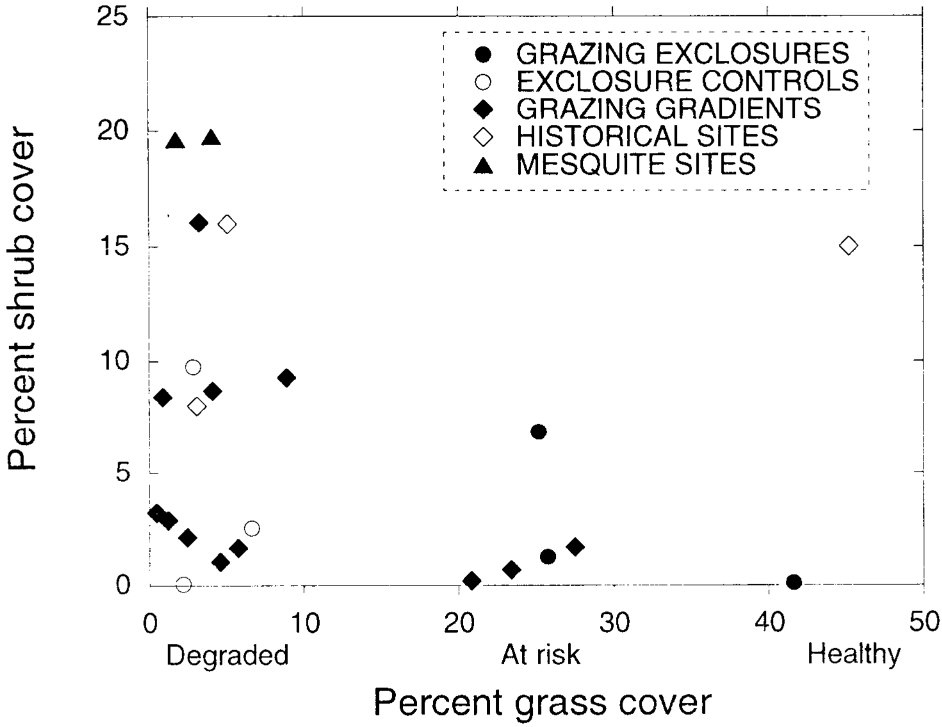


*Fig. 4.* Average shrub cover at sites ranging from irreversibly degraded to healthy.

removed the coppiced shrubs and the percent grass cover was increased (Figure 11). Bulldozing the dunes did not result in increased productivity as measured by the relative preference index despite the reduction in shrub cover and increase in grass cover (Figure 11).

Herbicide treatment of mesquite coppice dunes did not change the bare patch index nor the percent grass cover and had only a small effect on the relative preference index (in the direction opposite that hypothesized). Herbicide application in tarbush dominated sites produced mixed results (Figure 12). Bare patch index was higher on the grazed sites and lowest on the ungrazed, herbicide treated plot. Grass cover was equivalent to the best condition sites on the ungrazed, herbicide treated plots. Shrub cover was the same on the grazed sites and remained significantly depressed only on the ungrazed, herbicide treated plots (Figure 12). Despite changes in some of the key indicators, the productivity indicator (relative preference index) was not dramatically increased by the herbicide treatment. Root-plowing and seeding of creosotebush slope was relatively more effective than herbicide treatment in shifting indicators toward the healthy end of the spectrum. With the exception of percent long-lived grasses, the indicator values for the root-plowed site were healthy especially when compared to the untreated sites (Figure 13). The high cover of

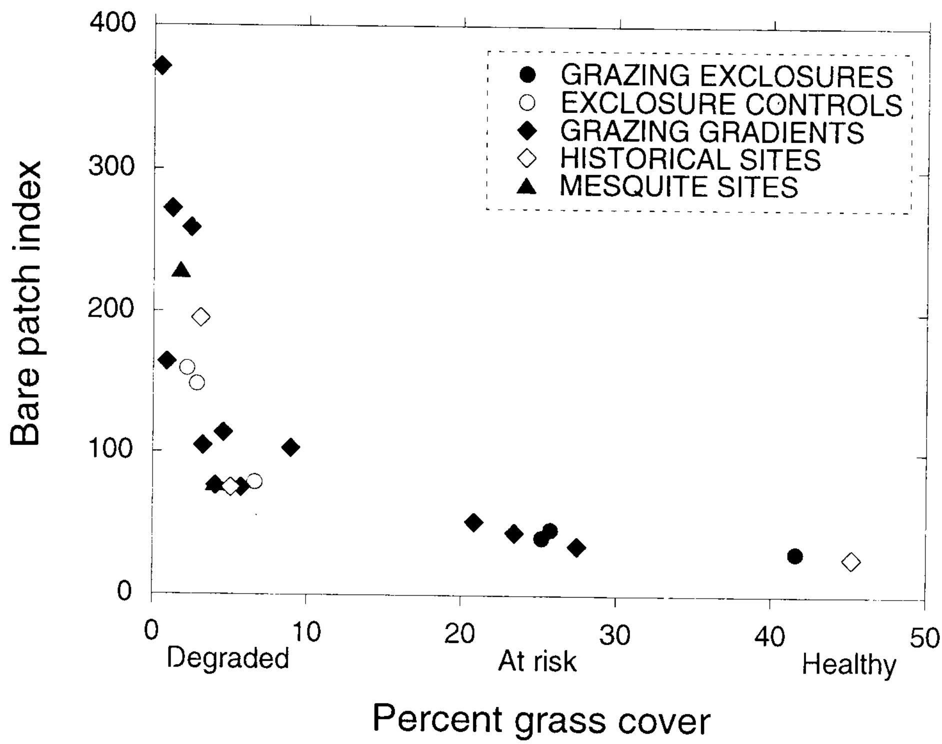


*Fig. 5.* Bare patch index at sites ranging from irreversibly degraded to healthy.

long lived grasses and low shrub cover on the ungrazed site on the creosotebush slope was due to bush muhly grass, *Muhlenbergia porteri*, which grew around the base of virtually every shrub in the exclosure.

## BIODIVERSITY INDICATORS

The diversity indices based on breeding birds reflected the vegetation structural diversity and not the relatively recent change from the pre- grazing stress vegetation. The lowest species richness was recorded in the black grama (Bouteloua eriopoda) grassland and the lowest diversity index was in the tarbush (Flourensia cernua) (Figure 14). These landscape units are dominated by shrubs or grasses that are less than 1 m in height. In each unit, there was one plot that had taller shrubs or yucca (Yucca elata). The highest diversity of breeding birds was in the mesquite - grass mosaic landscape unit that was centered on a small grass covered dry lake with a fringe of mesquite shrubs that were between 5m and 7 m tall (Figure 14) The breeding bird species richness and diversity was very similar in the other degraded landscape units. The pattern of abundance of breeding birds (no.. $ha^{-1}$) was the same as species richness. Species richness and diversity were higher in the area

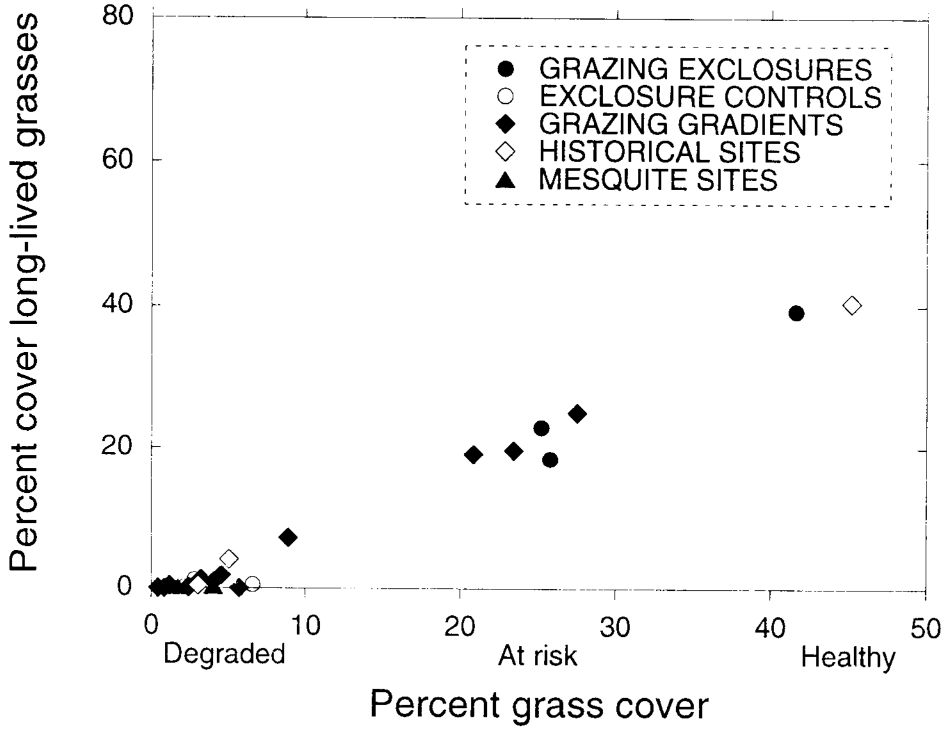


*Fig. 6.* Average percent cover of long-lived grasses at sites ranging from irreversibly degraded to healthy.

dominated by Lehmann lovegrass at the Empire Cienega Ranch but were higher in the native grassland at the Santa Rita Experimental Range (Figure 14).

## 4. Discussion

The National Research Council (1994) report on rangeland health emphasized the need for multiple indicators and for the use of benchmark sites for evaluating the efficacy of indicators. Our studies approached these needs by screening a suite of indicators on sites with long term vegetation cover records and on sites that had been subjected to one of several restoration efforts. These studies demonstrated that some potential indicators did not provide interpretable responses across a series of sites representing a gradient from healthy rangeland to degraded rangeland.

The National Research Council (1994) described the problems encountered in comparisons of plant composition to desired or site potential vegetation. They also cautioned about extrapolating from benchmark sites to all rangeland sites in a region. By focusing on indicators of ecosystem function rather than on indicators of desired or

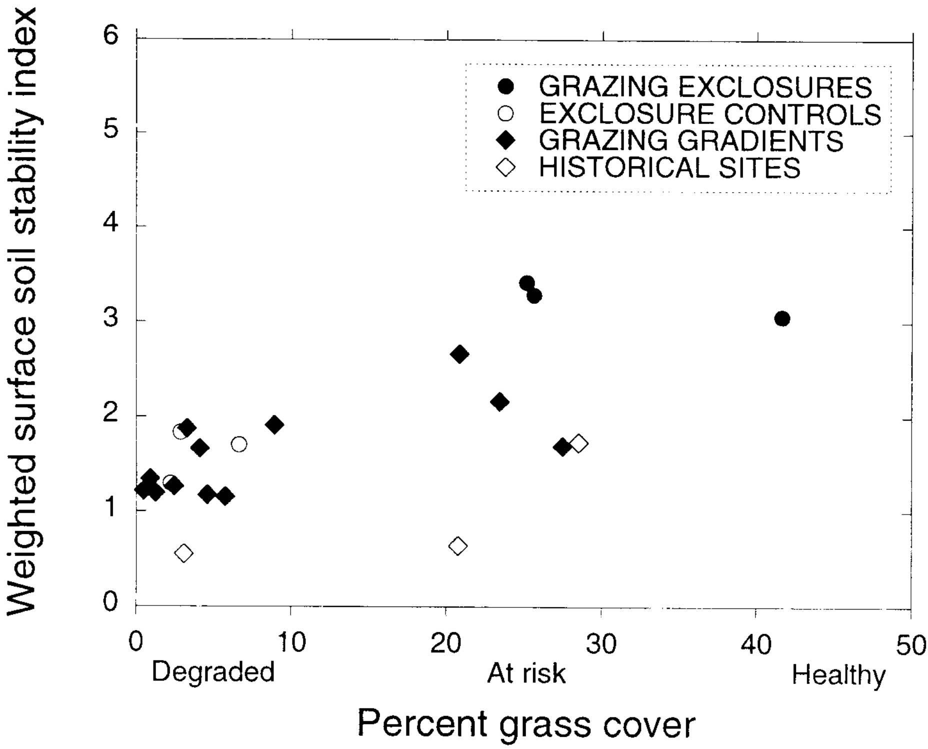


*Fig. 7.* Soil surface stability index at sites ranging from irreversibly degraded to healthy.

potential state, we avoid the problem of pre-judging the potential of a site. Indicators of ecosystem function allow a manager to evaluate the health of an ecosystem by assigning variable weights to the indicators that reflect management goals or community values (Herrick *et al.* 1995). Indicators of ecosystem function are general indicators because they are directly linked to ecosystem processes and are applicable to most if not all rangeland ecosystems. Since the most important feature of the indicators presented here is the linkage to ecosystem processes, we provide a discussion of those linkages.

LINKAGE TO ECOSYSTEM PROCESSES: The indicators for retention and use of soil and water resources *in situ* and the indicators for economic productivity are interdependent. Proportional vegetative cover of shrubs and grasses is directly related to the degree of linkage between precipitation and nitrogen mineralization and nitrogen immobilization. In grasses there is a direct coupling of water availability and the availability of nitrogen and other nutrients resulting in a proportional increase in the production response following rainfall (Stephens and Whitford 1993, Whitford and Herrick 1996). In shrub systems, there is frequent decoupling of rainfall and plant growth resulting from immobilization of nutrients in decomposing roots of ephemeral plants that grow in high densities under the shrub canopies (Whitford and Herrick 1996). Redistribution of rainfall also varies with the

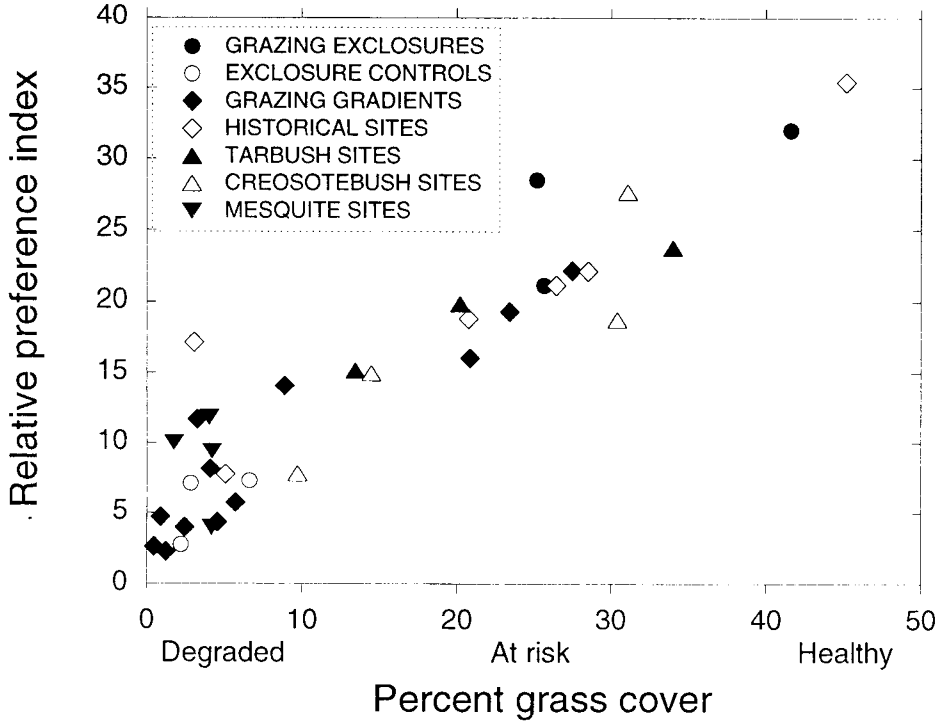


*Fig. 8.* Relative preference index (by livestock) at sites ranging from irreversibly degraded to healthy.

proportional cover of shrubs and grasses. Water is translocated deeper into the soil profile by shrubs than by grasses (Martinez-Meza and Whitford 1996). Preference of vegetation by livestock is in general related to growth form with grasses being generally more preferred than shrubs (Stubbendieck *et al.* 1993). However because of the differences in seasonal preference for different species of grasses, grass cover alone is not a good measure of the potential for livestock production. The relative preference index uses the information on seasonal use of each plant species at a site, thereby incorporating this variability into the calculation of the indicator.

An important component of retention of water and soil resources *in situ* is the efficacy of the vegetation in disrupting overland flow during intense rainfall events and of the vegetation to disrupt airflow during high winds. Grass clumps are generally far more effective in disrupting overland water movement than are shrubs or trees (Tongway and Ludwig 1997) and the efficacy of the grasses in slowing water movement is directly proportional to the grass cover. In desert rangelands where periodic drought is a characteristic of the climate, long-lived grasses afford greater protection to the soil than do the short lived grasses (i.e. perennial grasses that live less than a decade). Short lived grasses exhibit large swings in cover values in wet years following drought (Herbel and

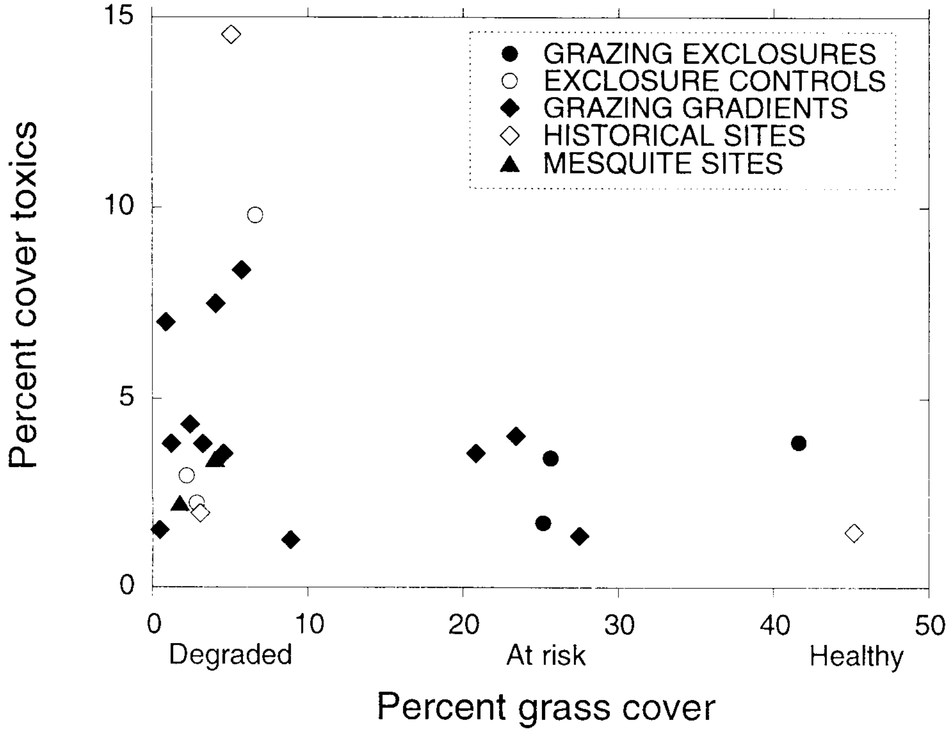


*Fig. 9.* Average percent cover of perennial plant species that are toxic to livestock at sites ranging from irreversibly degraded to healthy.

Gibbens 1996). Cover values of long-lived grasses are also related to the long-term protection of the soil surface from wind erosion.

The resistance of rangeland ecosystems to erosion is a function of a number of ecosystem properties. Both vegetation cover and bare patch size are important variables in the WEPS (Wind Erosion Prediction System) model. The functional model for wind erosion is: E = F(I, K, C, L, V) where E is potential annual soil loss per unit area, I is a soil erodibility index, K is a soil roughness factor, C is a climatic factor, L is the unsheltered median travel distance of wind across an unvegetated space and V is the quantity of vegetative cover (Skidmore 1986). Included in the soil erodibility index (I) are a number of other soil properties that affect wind erosion: crust properties, crust cover fraction, loose erodible material and soil bulk density (Zobeck 1991). This set of properties is related to the rangeland health soil stability indicator and by measures of cover of cryptogams, stones, and litter (variables measured on detailed lines not reported here). The most important variable in the WEPS model is the wind fetch, or unvegetated patch size. Unvegetated patch size is also an important variable with respect to water infiltration and storage and soil nutrients. Patches devoid of vegetation have very low soil organic matter,

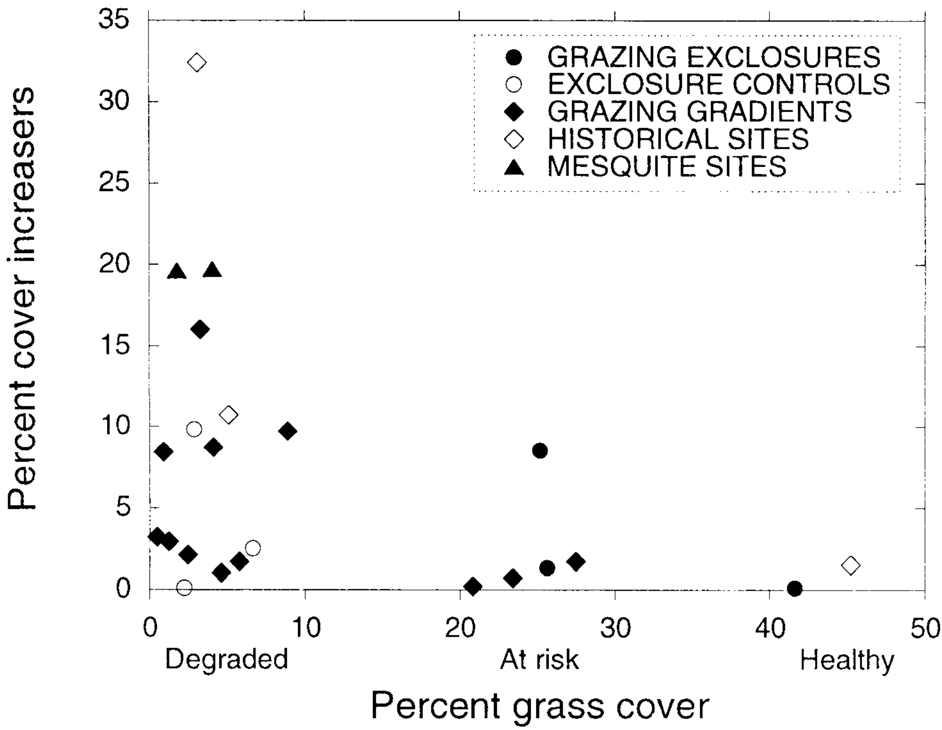


*Fig. 10.* Average percent cover of shrubs that are increasers at sites ranging from irreversibly degraded to healthy.

low total nitrogen and low rates of mineralization (Fisher *et al.* 1990). Unvegetated patches are also unsuitable habitat for most soil organisms (Santos *et al.* 1978, Elkins *et al.* 1986, and unpublished observations). For example subterranean termites and ants were absent from large unvegetated patches in Australian mulga woodlands (Whitford *et al.* 1992). Since soil organisms are responsible for the production of soil macropores which enhance water infiltration, the size and frequency of unvegetated patches also have a direct effect on infiltration and water storage (Bevan and Germann 1982).

The preceding discussion, while not an exhaustive review of linkages between the indicators based on ecosystem properties and ecosystem processes, does serve to demonstrate that properties that are rapidly and easily quantified in the field can be used as indicators of ecosystem processes.

ECONOMIC HEALTH AND PRODUCTIVITY: In arid lands around the world, productivity that is usable by humans is in the form of meat, fiber, or milk products from domestic livestock. Low quantity and temporally unpredictable rainfall precludes rain-fed crop production. Thus most of the arid and semi-arid lands of the world are referred to as rangelands where livestock production is virtually the only industry. The productivity

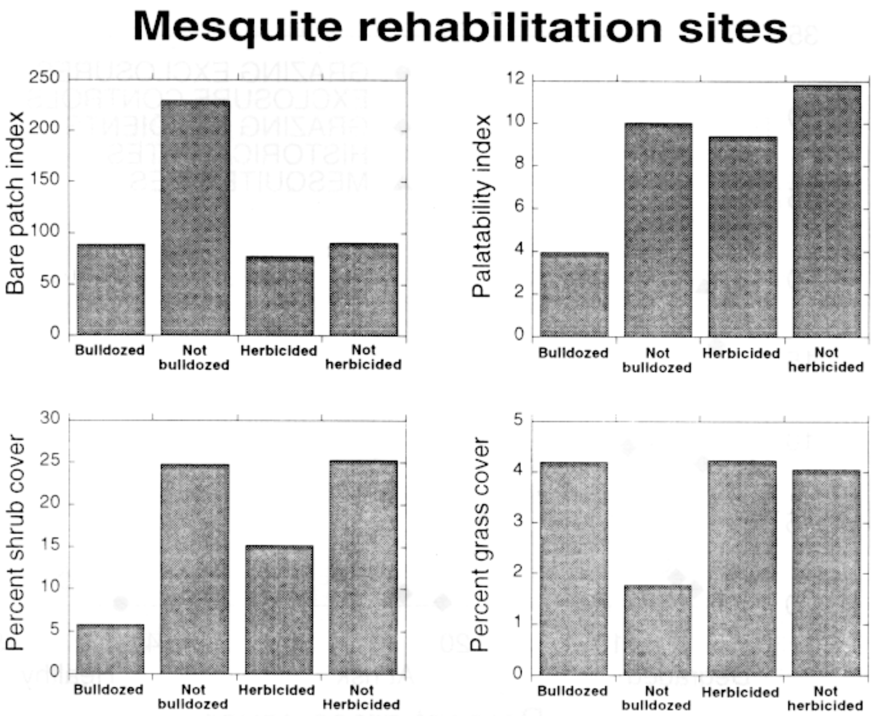


*Fig. 11.* Comparisons of values of selected indicators at mesquite, Prosopis glandulosa, coppice dune sites subjected to herbicide application restoration efforts and untreated sites.

measure that we chose to investigate is one that relates directly to livestock production: cover of preferred vegetation. This indicator was very sensitive to vegetation change due to desertification and to acute exposure to grazing by domestic livestock.

BIODIVERSITY: The lack of a pattern of reduction in biodiversity of perennial plants related to the change in vegetation resulting from exposure to grazing or as the result of long-term desertification processes may be in part attributable to the sampling technique used. The frequency of plants encountered along a line is not as good a measure of species richness as is obtained by careful census of nested quadrats ( T. J. Stohlgren, personal communication).

The lack of effects of vegetation change and stress on ecosystems from the grazing of domestic livestock was clearly documented in a study of ant communities on all of the sites sampled in testing indicators of rangeland health. There were no differences in species richness nor H' among protected, healthy sites and sites that were exposed to grazing stress or that had experienced dramatic change in composition and cover of the vegetative community within the past century (Whitford *et al.* in press). In addition to the absence of effects on diversity, there were no interpretable patterns of difference in a variety of metrics based on ant species feeding and life history characteristics.

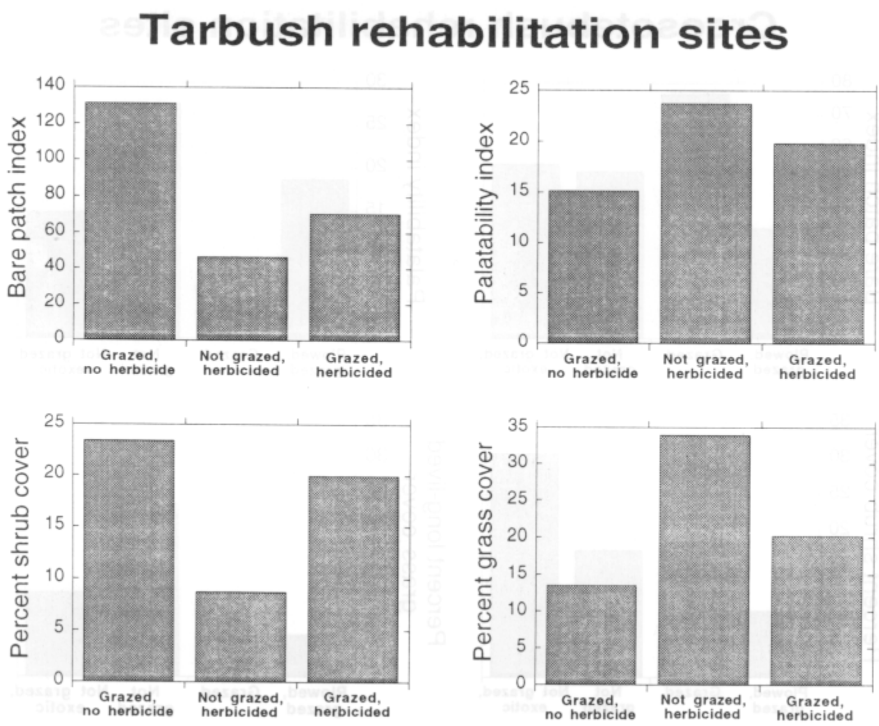


*Fig. 12.* Comparisons of values of selected indicators at tarbush, Flourensia cernua, dominated sites subjected to herbicide application restoration efforts and untreated sites.

Our data on rangeland health and breeding birds are similar to data from other studies in desert shrubland-grassland in the western U.S.. In a study of bird species assemblages as indicators in Great Basin rangelands, Bradford *et al.* (1997) found that two metrics (species richness and dominance) exhibited little overlap between values for heavily impacted sites and relatively unimpacted sites. They concluded that these measures were potentially good indicators of biological integrity. Bradford *et al.* (1997) added the caution that the sensitivity of these metrics suggested that they may be of limited usefulness in distinguishing between sites with light to moderate impacts. Our data on breeding birds demonstrated that species richness did not distinguish between sites except for the extreme example of a site dominated by an exotic grass species.

There are few species of breeding birds in desert grasslands and several of these species utilize the inflorescence stalks or leaf crowns of soaptree yucca, *Yucca elata*, as nesting sites (Wiens 1973, Naranjo and Raitt 1993). In 1970, Wiens (1973) reported 6 species of breeding birds from the desert grassland on the Jornada which we sampled in 1994 in this study. Of the 6 species that he reported, only four of those species were recorded in our census and we recorded 2 species that Wiens did not report (Scott's Oriole and Black-throated Sparrow). The higher diversity of breeding birds in the shrub

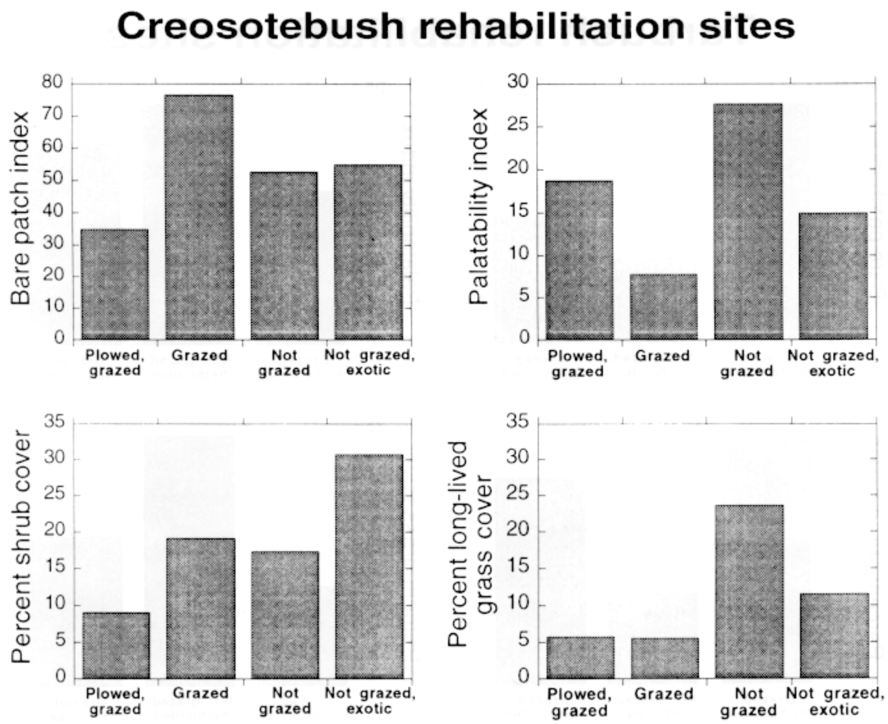


*Fig. 13.* Comparisons of values of selected indicators at creosotebush, Larrea tridentata, dominated sites subjected to root-plowing restoration efforts and untreated sites.

dominated habitats appears to be related to the architectural complexity of the vegetation. The highest species richness and diversity were recorded at a site with large mesquite trees, clumps of mesquite shrubs, scattered creosotebushes and dense patches of grass. The creosotebush, and tarbush shrub sites included patches of taller shrubs and patches of grasses within the relatively uniform height and density stands of the dominant shrubs. All of the sites were grazed at approximately the same stocking rates; hence, we cannot separate the responses of the avifauna to exposure to grazing from their responses to vegetation change resulting from desertification. The clear differences in species richness and diversity on the Lehmann lovegrass dominated site compared with the native grass site at the Santa Rita Range in southeastern Arizona is instructive. The reduction in diversity and abundance in the Lehmann lovegrass area at the Santa Rita is probably the result of changes in cover characteristics and food availability. At the Empire Cienega site, Lehmann lovegrass cover was not as high nor was the lovegrass distributed widely over all of the 50 m radius census plots. At Empire Cienega, four of the census circles in the native grassland had dense stands of large mesquite which were absent on the Lehmann lovegrass plots. At the Santa Rita Experimental Range, the Lehmann lovegrass had occupied the area for years while at the Empire Cienega, the Lehmann lovegrass was mixed with native grasses and had established less than 10 years prior to our studies.

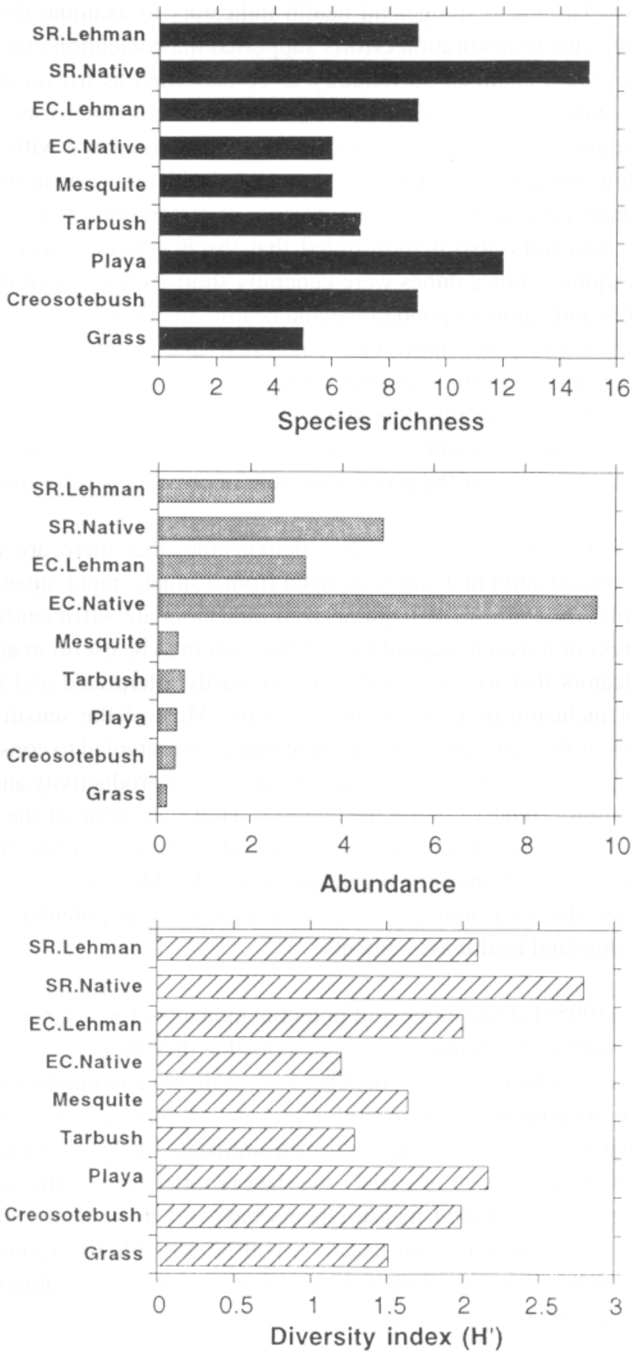


*Fig. 14.* Mean species richness, abundance, and Shannon's diversity index for breeding birds at a variety of sites exposed to environmental stressors and sites that exhibit few signs of environmental stress.

RESTORATION: The use of rangeland health indicators to examine the recovery of degraded sites subjected to restoration efforts supported the contention that the mesquite coppice dune sites were changed sufficiently to be classified as irreversibly degraded (Whitford 1995). Only a small fraction of the indicators showed change toward the healthy values in the mesquite coppice dune restoration sites when compared with the untreated controls. Those few indicators were primarily associated with reduction in shrub cover but not with soil surface stability, resistance to erosion, or restoration of productivity. The rangeland health indicators also demonstrated that the beneficial effects of herbicide application to mesquite coppice dunes were generally short-lived and that the herbicided dunes exhibited few indicators of good rangeland health. In the creosotebush and tarbush dominated degraded sites, the indicators suggested that herbicide application and root-plowing restored some of the ecosystem functions of healthy rangelands but did not completely restore the areas to a healthy state. The suite of indicators examined in this study do provide a means of quantifying the relative success of restoration efforts and demonstrating what functions of the ecosystem were improved by restoration.

GENERAL CONCLUSIONS: Our studies demonstrate that there are a number of indicators of rangeland health that can be derived from a simple, rapid, quantitative set of field measurements. Not all of the hypothesized indicators are sufficiently sensitive to environmental stress or to recent degradation of the system to be useful in an assessment. Some of the indicators that we examined were not easily interpreted and therefore not recommended for inclusion in an assessment system. Most of the sensitive indicators provided measures of the capacity of the patch of rangeland sampled to conserve soil and water resources. There was at least one sensitive indicator of productivity and of potential economic sustainability (relative preference index). However, none of the indicators of biodiversity examined to date were sensitive to ecosystem stress resulting from livestock grazing nor to recent large changes in vegetation and soils. Measures other than species richness or species diversity indices need to be examined as potential indicators of biodiversity for rangeland health assessments.

Herrick *et al.* (1995) listed a number of potential indicators for use in assessing and/or monitoring rangeland health. Some of the indicators that they listed are more appropriate for a monitoring system than for assessment because of the time requirements to make the measurements (i.e. infiltration capacity, soil aggregate stability, and soil texture). Other indicators are derived from a set of detailed measurements made on randomly selected small segments of the lines used to collect the data reported here. The sensitivities of indicators based on these detailed measurements have yet to be analyzed. When the complete suite of indicators has been tested, it will be possible to examine how these indicators can be combined and incorporated into a scoring system for evaluating rangeland health (Herrick *et al.* 1995).

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## References

Anable, M. E., M. P. McClaran, Ruyle, G. B. 1992. "Spread of introduced Lehmann lovegrass (Eragrostis lehmanniana Nees.) in southern Arizona, USA." *Biol. Cons.* **61**:181-188.

Andrew, M. H., Lange, R. T. 1986. "Development of a new piosphere in arid chenopod shrubland grazed by sheep. 2. Changes to the vegetation". *Australian J. Ecology* **11**: 411-424.

Bevan, K., Germann, P. 1982. "Macropores and water flow in soils." *Water Resources Research* **18**:1311-1325.

Bradford, D. F., Franson, S. E., Neale, A. C., Heggem, D. T., Miller, G. R., Canterbury, G. E. 1997. "Bird species assemblages as indicators of biological integrity in Great Basin rangeland." *Environ. Monitor.Assess.* (In Press)

Buffington, L. C., Herbel, C. H. 1965. "Vegetation changes on a semi-desert 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.

De Soyza, A. G., Whitford, W. G., Herrick, J. E. 1996. "Sensitivity testing of indicators of ecosystem health." *Ecosystem Health* (In press).

Dick-Peddie, W. A. 1993. "New Mexico Vegetation, Past, Present and Future." University of New Mexico Press. Albuquerque, N.M.

Elkins, N. Z., Sobal, G. V., Ward, T. J., Whitford, W. G. 1986. "The influence of subterranean termites on the hydrological characteristics of a Chihuahuan Desert ecosystem." *Oecologia* **68**:521-528.

Fisher, F. M., Freckman, D. W., Whitford, W. G. 1990. "Decomposition and soil nitrogen availability in Chihuahuan Desert field microcosms." *Soil Biol. Biochem.* **22**:241-249.

Fusco M., Holachek, J., Tebo, A., Daniel, A., Cardenas, M. 1995. "Grazing influence of watering point vegetation in the Chihuahuan Desert." *J. Range Mgmt.* **48**: 186-192.

Gardiner, J. L. 1951. "Vegetation of the creosotebush area of the Rio Grande valley in New Mexico." *Ecological Monographs* **21**:397-403.

Gibbens, R. P., Beck, R. F. 1988. "Changes in grass basal area and forb densities over a 64-year period on grassland types of the Jornada Experimental Range." *J. Range Mgmt.* **41**:186-192.

Gibbens, R. P., Beck, R. F., McNeely, R. P., Herbel, C. H. 1992. "Recent mesquite establishment in the northern Chihuahuan Desert." *J. Range Mgmt.* **45**:585-588.

Grover, H. D., Musick, H. B. 1990. "Shrubland encroachment in southern New Mexico, U. S. A.: An analysis of desertification processes in the American southwest." *Climatic Change* **17**:305-330.

Herbel, C. H., Ares, F., Bridges, J. 1958. "Hand-grubbing mesquite in the semidesert grassland." J. Range Mgmt. **11**: 267-270.

Herbel, C. H., Abernathy, G. H., Yarbrough, C. C., Gardner, D. K. 1973. "Rootplowing and seeding arid rangelands in the southwest." *J. Range Mgmt.* **26**: 193-197.

Herbel, C. H., Gould, W. L., Leifeste, W. F., Gibbens, R. P. 1983. "Herbicide treatment and vegetation response to treatment of mesquites in southern New Mexico." *J. Range Mgmt.* **36**: 149-151.

Herbel, C. H., Morton H. L., Gibbens, R. P. 1985. "Controlling shrubs in the arid southwest with tebuthiuron." *J.Range Mgmt.* **38**: 391-394.

Herbel, C. H., Gibbens, R. P. 1996. "Post-drought vegetation dynamics on arid rangelands in Southern New Mexico. New Mexico State Univ. Agric. Exper. Sta. Bull. 776. Las Cruces, NM.

Herrick, J. E., Whitford, W. G., deSoyza, A. G., Van Zee, J. 1995. "Soil and Vegetation Indicators for Assessment of Rangeland Ecological Condition." (pp.157-166) In: Celedonio, A. G., ed. *North American Workshop on Monitoring for Ecological Assessment of Terrestrial and Aquatic Ecosystems. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station.* 305 p.

Martinez-Meza, E., Whitford, W. G. 1996. "Stemflow, throughfall and channelization of stemflow by roots in three Chihuahuan Desert shrubs." *J. Arid Environ.* **32**:271-287.

Naranjo, L. G., Raitt, R. J. 1993. "Breeding bird distribution in Chihuahuan Desert habitats." *Southwest. Nat.* **38**:43-51.

National Research Council. 1994. "Rangeland Health: New Methods to Classify, Inventory, and Monitor Rangelands." National Academy Press. 180 p.

Rapport, D. J., Regier, H. A., Hutchinson, T. C. 1985. "Ecosystem behaviour under stress." *Am. Nat.* **125**:617-640.

Santos, P. F., Depree, E., Whitford, W. G. 1978. "Spatial distribution of litter and microarthropods in a Chihuahuan Desert ecosystem." *J. Arid Environ.* **1**:41-48.

Skidmore, E. L. 1986. "Wind erosion control." *Climatic Change* **9**:209-218.

Stephens,G., Whitford, W. G. 1993. "Responses of Bouteloua eriopoda to irrigation and nitrogen fertilization in a Chihuahuan Desert rassland." *J. Arid Environ.* **24**:415-421.

Stubbendieck, J., Hatch, S. L., Butterfield, C. H. 1993. North American Range Plants (4th ed.) University of Nebraska Press, Lincoln

Tongway, D. J., Ludwig, J. A. 1997. "The conservation of water and nutrients within landscapes." In. Ludwig, J., Tongway D., reudenberger D., Noble, J., Hodgkinson, K. (eds.) *Landscape Ecology: Function and Management.* CSIRO Publishing, Collingwood, Victoria

Tongway, D. 1994. "Rangeland Soil Condition Assessment Manual." CSIRO, Division of Wildlife and Ecology.Australia.

Westoby, M., Walker, B., Noy-Meir, I. 1989. "Opportunistic management for rangelands not at equilibrium." *J. Range Manage.* **42**:266-274.

Whitford, W. G., Ludwig, J. A., Noble, J. C. 1992. "The importance of subterranean termites in semi-arid ecosystems in south-eastern ustralia." *J. Arid Environ.* **22**:87-91.

Whitford, W. G. 1995. Desertification: Implications and limitations of the ecosystem health metaphor. In: Rapport, D. J., Gaudet, C. L., alow, P. (eds) Evaluating and Monitoring the Health of Large-Scale Ecosystems. NATO ASI Series, Vol. 128. Springer-Verlag, Berlin pp.271-296.

Whitford, W. G., Herrick, J. E. 1996. "Maintaining soil processes for plant productivity and community dynamics." In.West, N. E. (ed) Rangelands in a sustainable biosphere (vol. 2) Proceedings of the Fifth International Rangeland Congress. Society for Range Management. Denver, CO pp. 33-37.

Whitford, W. G., Van Zee, J., Nash, M. S., Smith, W. E., Herrick, J. E. 1997. "Ants as indicators of exposure to environmental stressors in North American desert grasslands." *Environ. Monitor. Assess.* In Press.

Wiens, J. A. 1973. "Pattern and process in grassland bird communities." *Ecol. Monogr.* **43**:237-270.

Zobeck, T. M. 1991. "Soil properties affecting wind erosion." *J. Soil Water Conserv.* **46**:112-118.