Sensitivity Testing of Indicators of Ecosystem Health

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ABSTRACT

The sensitivities of three indicators of ecosystem health were evaluated at several sites in the Jornada Basin of the Chihuahuan Desert in southern New Mexico. The size of bare patches, proportion of total grass cover contributed by long-lived perennial grasses, and soil stability are interdependent indicators of ecosystem functions related to the retention and use of water and nutrients. Sensitivity tests were chosen using data collected along disturbance gradients and then tested using independent, ungrazed exclosures and adjacent grazed pastures. The mean size of bare soil patches was sensitive to anthropogenic disturbance. When bare soil patch data were transformed using natural logarithms, the skewness of the frequency distribution weighted by mean bare patch size could be used to indicate early disturbance to the ecosystem. The proportion of total vegetation that was long lived also was sensitive to anthropogenic disturbance and appears to be a good indicator of ecosystem degradation. The slake test for soil surface stability was extremely sensitive to disturbance and may serve as an early-warning indicator of soil degradation for the coarse-textured soils that were evaluated.

INTRODUCTION

The recognition that ecosystem health is a strong determinant of economic and social health has led environmental managers and policy-makers to be more receptive to the need to incorporate ecological considerations into management strategy. As with issues of human health, the first step in addressing ecosystem health is the detection of a symptom, an indicator, that a problem exists. This is not as simple as it would seem. Ecosystems are dynamic over time and space. Therefore, indicators of ecosystem health should be relatively insensitive to those variables that are a natural component of ecosystem dynamics (e.g., climatic variation) and at the same time are sensitive to disturbances that are anthropogenic in origin. To achieve this, it has been suggested that ecosystem condition be evaluated on the basis of ecosystem function (e.g., Herrick *et al.* 1996). Indicators to assess ecosystem condition have been used by community ecologists for many years (e.g., Clements 1928). More recently, methods for the selection of indicators of rangeland ecosystem condition (e.g., Breckenridge *et al.* 1995, Herrick *et al.* 1996) and sensitivity analysis of indicators (e.g., Tiscareno-Lopez *et al.* 1993, Benkobi *et al.* 1994) have received some attention.

Methods for the selection of indicators for assessing ecosystem health are currently being developed and debated (e.g., Breckenridge *et al.* 1995; Herrick *et al.* 1996). Some criteria for the selection of indicators are rapidity and reliability of measurement, repeatability, and linkage to ecosystem function.

Analogous to indicators of human health, indicators of ecosystem health must change in a predictable, repeatable way when ecosystems shift to a less healthy condition. Sensitive indicators of human health have been selected by comparisons of measurements on large numbers of sick and healthy patients. Indicators of ecosystem health must be tested to determine their sensitivity as measures of ecosystem health before they can be used in general assessments of ecosystem health.

To test the sensitivity of indicators, it is essential to locate sites on similar soils and landscape positions in which large differences in ecosystem functions have been generated by variable intensities and/or durations of exposure to anthropogenic stressors. Such sites are frequently referred to as benchmark or reference sites (National Research Council 1994). By measuring the responses of the indicator variables across a series of sites with documented levels of historical stress, it should be possible to evaluate the sensitivity of a suite of indicators of ecosystem health.

In this article, we provide examples of sensitivity tests for selected indicators of a key rangeland ecosystem function: the conservation of soil and water resources. These indicators include (1) size of bare soil patches, (2) proportions of total vegetative cover that are long-lived or short-lived perennial grasses, and (3) soil surface stability based on slake test measurements. The size of bare soil patches is an indicator of ecosystem fragmentation, which has implications for fluvial and aeolian redistribution of soil. Increases in the size of bare soil patches are associated with increased soil loss due to erosion by both wind and water. Near-surface wind velocity increases with increased gap size, whereas water runs off more quickly and carries more material offsite because there are fewer barriers to divert the flow and trap soil and organic debris.

Short-lived perennial grasses are susceptible to increased mortality during single year growing season droughts (Gibbens & Beck 1988), which reduce their effectiveness in protecting the soil. The long-lived perennial grasses, unlike short-lived perennial grasses, have a lower biomass turnover rate and a more spreading morphology which is more efficient in accumulating and retaining soil and water resources. The long-lived perennial grasses reduce soil losses in three ways. Their longer lifespan results in less frequent gap (bare space) formation following mortality. Second, the long-lived species in the Chihuahuan Desert tend to be stoloniferous, whereas the short-lived species are bunchgrasses. The stoloniferous morphology more effectively reduces raindrop impact and, hence, soil detachment, physical crusting, and runoff. Finally, this morphology tends to trap litter in a more dispersed pattern than does bunchgrass morphology, maintaining a more homogeneous distribution of nutrient resources and soil faunal activity.

Soil surface stability is the system's last defense against soil and water resource losses. The slake test used to measure soil surface stability (the physical integrity of soil) is a measure of the ability of the system to retain soil nutrient resources. The stability of soil crusts as measured by the slake test is a function of physical and biological processes that cement soil particles together. The slake test measures the resistance of the soil surface to dispersion and detachment by raindrop impact and overland flow. Soils that slake easily form crusts more easily, reducing water infiltration, and are also more susceptible to detachment and transport by water.

METHODS

The study sites were in the Jornada Basin of the northern Chihuahuan Desert in southern New

TABLE 1

Characteristics of sites in the Jornada where vegetation cover and soil stability were measured[1]

Site	Plot Code	Distance from Well (m)	Grazing History
	Disturbance Gradients		
Gradient WW	WW0	50	Winter–Spring
	WW1	200	Winter–Spring
	WW2	450	Winter–Spring
	WW3	1050	Winter–Spring
Gradient CW	CW0	50	Winter–Spring
	CW1	200	Winter–Spring
	CW2	450	Winter–Spring
	CW3	1050	Winter–Spring
Gradient MW	MW0	50	Continuous
	MW1	200	Continuous
	MW2	450	Continuous
	MW3	1050	Continuous
	Ungrazed–Grazed Comparisons:		
Exlosure West	EXW	NA	Exclosed 1946
	EWC	NA	Continuous
Exclosure East	EXE	NA	Exclosed 1946
	EEC	NA	Summer
Exlosure Northwest	EXN	NA	Exclosed 1946
	ENC	NA	Continuous

The distance from livestock watering points (wells) produces the disturbance gadient at the three gradient sites. The grazed–ungrazed comparisons are based on 50-year-old exclosures and adjacent grazed pastures.

Mexico, ~40 km north northeast of Las Cruces, New Mexico. Measurements were made at six sites (Table 1). At three of the sites, measurements were made on four plots located along a disturbance gradient (Andrew & Lange 1986; Fusco *et al.* 1995) at increasing distances from stock watering points. At the other three locations, conditions inside and outside of 50-year-old exclosures were assessed. The soils at all six locations are sandy-loams over a layer of caliche (indurated calcium carbonate). The depth to caliche varied from 0.2 to >1 m below the soil surface within individual sites. We developed the sensitivity tests by using data from the three disturbance gradient sites. The sensitivity tests were later evaluated using data from the three exclosure-pairs.

VEGETATION AND BARE SOIL PATCHES

At four points along each disturbance gradient, we established a 1-hectare plot which was sampled by 10 100-m transect lines parallel to the disturbance gradient. The midpoint of each plot along the disturbance gradients was at 50, 200, 450, and 1050 m, respectively, from the watering point. The transects were laid out 10 m apart and parallel to each other. Measurements were made by recording the identity of the plant species or bare patch and the horizontal linear dimensions of the plant or bare patch intercepting the transect (Canfield 1941). From these data, we extracted information on the relative size of bare patches and the cover of long-lived and short-lived perennial grasses.

SOIL SURFACE CONDITION

We used a slake test (e.g., Tongway 1994), modified to account for differences in wet aggregate strength, to determine soil and crust stability. Soil stability was determined on soil from three strata: bare soil, grass, and shrub. At each site, we selected three transects at random from the 10 vegetation transects. Before begining measurements, we generated a list of 25 sorted random numbers spanning the length of each line (e.g., 0–100 for a 100-m transect). We sampled the soil at each point corresponding to a random number along a transect until we obtained three samples for each stratum. If the selected sampling point did not yield a soil fragment (i.e., the soil disintegrated), it was assigned a value of 1.0. If fewer than three samples were obtained for a particular stratum, an additional 25 random numbers were generated and the transect was repeated for that stratum only. Soil was tested from both the upper (0–3 mm) crust layer and just below the crust (15–20 mm). These two measurements, respectively, provide an indication of (1) current stability and (2) potential future stability following crust disturbance. The slake test was done only on air-dry soils because moist soils tend to overestimate relative stability. Soil stability was measured on soil fragments, 6 to 8 mm in diameter and 2 to 3 mm thick. Soil fragments were placed in a small 25-mm-diameter PVC basket with a wire mesh bottom (1.5 mm screen size). The baskets were slowly lowered into a reservoir of distilled water at the same temperature as the soil. Each fragment was observed for 5 minutes. If the soil fragment did not disintegrate within this time, the basket with soil fragment was raised until the bottom of the basket was at the surface of the water and then lowered into the water three times. Soil stability was rated according to the time required for the fragment to disintegrate during the 5-min immersion (classes 1–3) or the proportion of the soil fragment remaining on the mesh after the three extraction–immersion cycles (classes 4–6; Table 2).

TABLE 2

Soil slake classes[1]

Slake Class	Criteria for Assignment to Slake Class
1	50% of soil structure lost within 5 sec of insertion in water
2	50% of soil structure lost between 5–30 sec afer insertion
3	50% of soil structure lost between 30–300 sec after insertion
4	0–25% of soil remaining on sieve after three dipping cycles
5	>25–75% of soil remaining on sieve after three dipping cycles
6	>75–100% of soil remaining on sieve after three dipping cycles

[1]Criteria for assignment of soil slake class are based on (1) the time taken for 50% or more of the soil segment to disintegrate when inserted into deionized water (classes 1–3) or (2) if no disintegration has occured by 300 sec (5 min), the percentage of soil remaining on the sieve following three slow extraction–reinsertion cycles (classes 4–6). Soil that was too unstable to permit sampling was assigned slake class 1.

RESULTS AND DISCUSSION

Indicators are metrics that can sometimes be used directly as indicators, and at other times they must be combined or modified. The size of bare patches, proportion of total grass cover contributed by long-lived perennial grasses, and soil crust stability are interdependent indicators of ecosystem functions related to the retention and use of water and nutrients. These indicators are interdependent because they are measures of variables that affect the resistance to erosion, water infiltration, water storage, and the habitat suitability for a diverse soil flora.

Considering the many and complex interacting ways that these indicators are linked to ecosystem function, it is clear that several indicators that vary in a consistent and repeatable pattern in healthy and unhealthy ecosystems are needed for ecosystem health assessment. To be useable, each indicator must clearly differentiate between healthy and unhealthy systems. In this study, we have examined a small subset of the potential suite of indicators that can be derived from the same set of field measurements, or metrics. We have examined the performance of those indicators in ecosystems for which good historical records exist and where exposure to specific stressors can be examined.

The resistance of a rangeland ecosystem to erosion is a function of a number of ecosystem properties. The functional relationship for wind erosion developed in the Wind Erosion Prediction System model is

$$E = f(I, K, C, L, V), \qquad (1)$$

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 quantity of vegetative cover (Skidmore 1986). Other soil properties also affect wind erosion. These include crust properties, crust cover fraction, loose erodible natural, and soil bulk density (Zobeck 1991). Many of these properties are related to one or more of the metrics reported in this study.

Unvegetated soil in arid and semi-arid regions is unsuitable as habitat for most soil organisms (Santos *et al.* 1978; Santos & Whitford 1983; Elkins *et al.* 1986). The reduced abundance and diversity of soil biota results in few macropores, thereby reducing infiltration (Bevan & Germann 1982). Unvegetated soil has extremely low organic matter content because of the absence of roots and surface litter. In desert rangeland ecosystems, soil nutrient pools and rates of nutrient cycling are positively correlated with organic matter content of soils (Whitford *et al.* 1987).

The size of bare soil patches is an important indicator of ecosystem health. The presence of large bare patches obviously reduces the potential productivity of an ecosystem, but more importantly, it leads to resource loss and affects the long-term health of the ecosystem. Water infiltration rates are usually lower in unvegetated bare soil due to the absence of soil macropores and roots for water channelization into the soil. The subsequent enhanced overland flow leaches nutrients from the ecosystem and may lead to ecosystem deterioration. Aeolian processes become more important as bare soil patch size increases. However, even sparse standing vegetation can significantly reduce soil loss due to wind (van de Ven *et al.* 1989). In summary, systems with small bare patches are healthy and those with large bare patches are less healthy.

A single descriptive statistic such as mean size of bare patch alone may not be a sensitive indicator of ecosystem health. Our measurement of the linear dimension of bare patches yields information on fetch, which correlates with the potential of a site within an ecosystem to erosional degradation by wind and rain. However, the critical fetch length, which depends on factors such as mean canopy height, may vary substantially within an ecosystem and even more so among ecosystems. Also, in the context of potential for soil erosion, mean bare patch size alone yields little useful information. A given mean bare patch size can be generated by an infinite number of bare patch size combinations. A frequency distribution of bare patches of various sizes conveys information more useful for determination of ecosystem health. We hypothesized that a frequency distribution of bare patch sizes with the classic, normal, bell-shaped curve and a mean value less than the critical fetch length would be indicative of a healthy ecosystem. At the early stages of disturbance, we hypothesized that there would be an increase in the number of small bare patches, which would cause the frequency distribution to become skewed to the right; as the ecosystem deteriorated further, larger bare patches would become more common, causing the frequency distribution to skew to the left (Figure 1). However, when the frequency distribution of bare patch sizes was plotted for each of the four plots along each disturbance gradient, there was no clear relationship between the shape

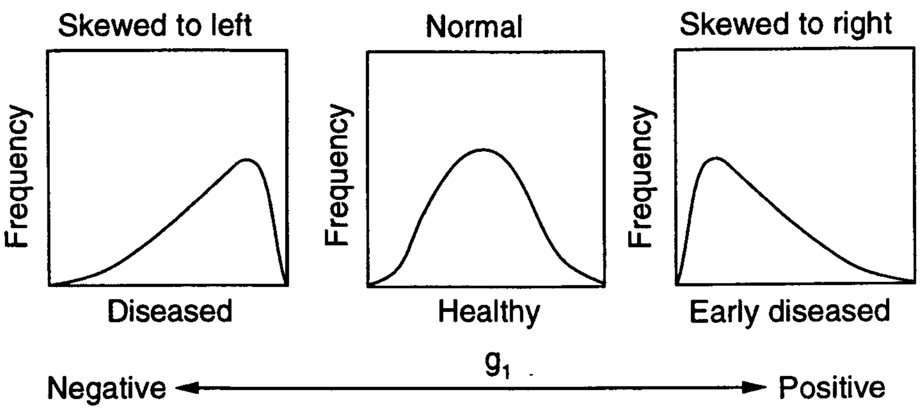

FIGURE 1. Diagram showing suggested interpretation of skewness of log-transformed bare soil patch size data. Increasingly negative skewness indicates increasingly severe degradation. A (near) normal distribution, together with a small mean bare patch size, indicates a healthy ecosystem.

of the distribution curve and distance from the watering point (Figure 2).

A variety of data transformations are available to increase the normality of data and are frequently used in statistical analyses requiring normally distributed data. We used a commonly employed transformation, natural logarithms. This transformation greatly improved the normality of the bare patch size data. With this transformation, the expected pattern for skewness (the g_1 statistic) emerged, linking the shape of the frequency distribution of $\log_n$-transformed bare patch size to degree of disturbance (Table 3).

Because skewness refers only to the shape of the distribution, the mean size of bare patches at any site must also be taken into consideration when interpreting this data. For example, a frequency distribution of transformed bare patch sizes ($LogB_p$) that is severely skewed to the left may yet be representative of a relatively healthy ecosystem if the mean untransformed bare patch size (B_p) is less than the critical fetch value. To generate an indicator matric that takes this into account, we calculated a weighted skewness:

$$S_w = g_1 \times B_p, \quad (2)$$

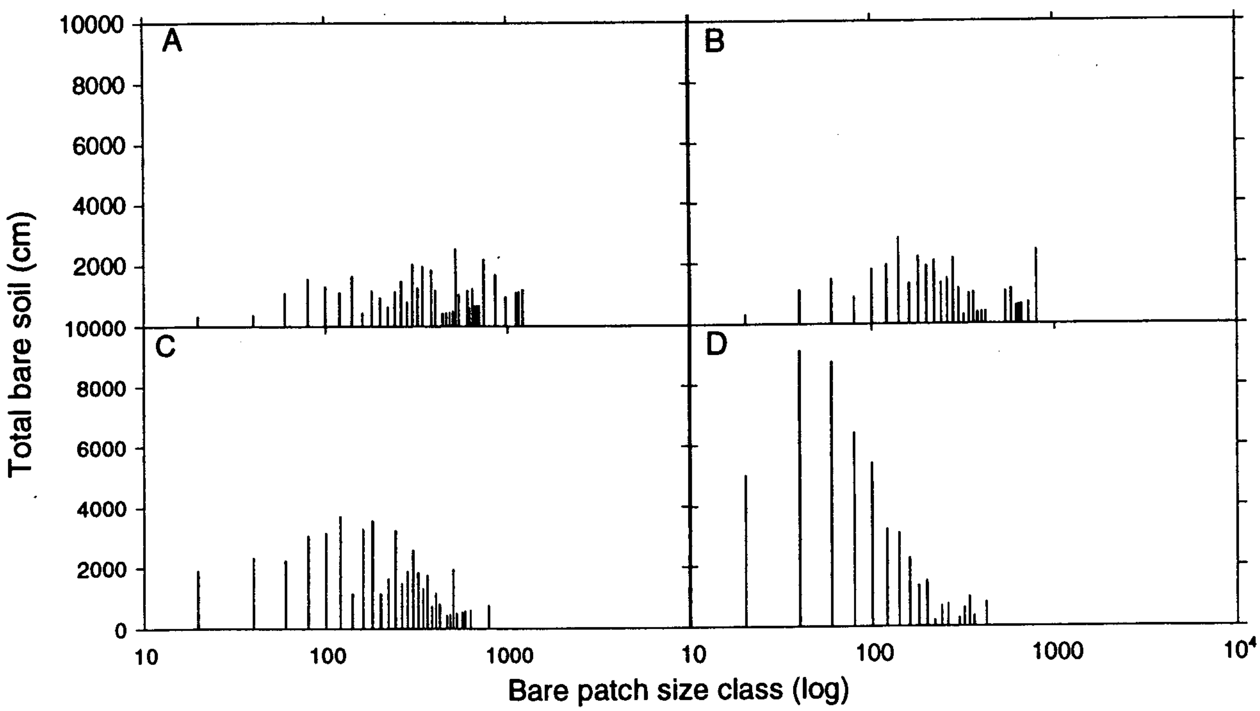

FIGURE 2. Frequency distributions for bare soil patches at Camp Well. A is the distribution at CW0, 50 m; B is the distribution at CW1, 200 m; C is the distribution at CW2, 400 m; and D is the distribution at CW3, 1050 m. The x-axis is a log scale.

TABLE 3

Descriptive statistics for linear bare patch size of raw (untransformed) data and data after transformation to natural logarithms

	Untransformed Data							*Log$_n$-Transformed Data*						
Site	Mean	SD	Min	Max	g_1	g_2	Median	Mean	SD	Min	Max	g_1	g_2	Median
							Disturbance Gradients							
WW0	197.6	225.6	2.0	1204.0	1.996	4.377	111.5	4.6	1.3	0.69	7.1	−0.401	−0.267	4.7
WW1	139.2	143.7	4.0	845.0	2.363	6.867	102.0	4.5	1.0	1.39	6.7	−0.315	−0.245	4.6
WW2	82.1	108.2	1.0	765.0	2.233	6.132	37.0	3.5	1.5	0	6.6	−0.100	−1.045	3.6
WW3	41.0	45.9	2.0	408.0	3.189	15.073	26.0	3.3	1.0	0.7	6.0	0.44	−0.430	3.3
CW0	396.3	668.8	4.0	4515.0	4.143	19.538	192.0	5.2	1.3	1.4	8.4	−0.224	0.281	5.3
CW1	126.0	145.9	2.0	960.0	2.824	9.984	81.0	4.3	1.1	0.7	6.9	−0.409	0.354	4.4
CW2	74.0	125.6	1.0	1000.0	3.410	15.254	20.0	3.1	1.7	0	6.9	0.126	−1.064	3.0
CW3	70.1	67.8	3.0	521.0	2.330	7.856	50.0	3.8	0.9	1.1	6.3	−0.148	−0.384	3.9
MW0	296.4	412.5	5.0	2758.0	3.230	13.432	143.5	4.9	1.3	1.6	7.9	−0.253	−0.365	5.0
MW1	96.6	93.0	4.0	700.0	2.269	7.578	72.5	4.1	1.0	1.4	6.6	−0.399	−0.195	4.3
MW2	86.9	86.0	3.0	600.0	2.069	5.988	60.0	4.0	1.1	1.1	6.4	−0.330	−0.444	4.1
MW3	39.7	52.6	2.0	550.0	3.664	20.462	20.0	3.1	1.1	0.7	6.3	0.233	−0.590	3.0
							Ungrazed–Grazed Comparisons							
EXW	54.1	41.5	7	229	1.669	3.876	42.0	3.7	0.8	1.9	5.4	−0.195	−0.530	3.7
EWC	125.1	134.2	3.0	957.0	2.494	8.520	82.0	4.3	1.1	1.1	6.9	−0.243	−0.340	4.4
EXE	56.6	48.1	5.0	294.0	1.836	3.958	42.0	3.7	0.8	1.6	5.7	−0.185	−0.317	3.7
EEC	118.2	136.1	4.0	1160.0	3.424	18.323	76.0	4.3	1.1	1.4	7.1	−0.279	−0.085	4.3
EXN	50.2	51.0	4.0	321.0	2.705	9.550	36.0	3.5	0.9	1.4	5.8	0.062	−0.239	3.6
ENC	98.4	98.2	2.0	644.0	2.060	5.564	66.0	4.1	1.1	0.7	6.5	−0.353	−0.285	4.2

g_1 = skewness and g_2 = kurtosis.

where S_w is weighted skewness, g_1 is skewness of $LogB_p$, and B_p is the mean of untransformed bare patches. The weighted skewness for the three disturbance gradients was plotted against distance from watering point, yielding a sensitive, positive, linear correlation (Figure 3). In the least disturbed (most distant from watering point) sites along the disturbance gradients, the frequency distribution is skewed to the right (a positive value) and the weighted skewness is a little offset from zero, indicating a small B_p. In the heavily disturbed sites near the watering point, the very negative value of weighted skewness indicates that these sites are strongly skewed to the left and B_p is large. The special case when $g_1 = 0$ can cause misinterpretation of the weighted skewness metric. Therefore, whenever the weighted skewness metric is used, the mean size of bare patch should also be reported.

Also shown in Figure 3 are the relative positions of the ungrazed–grazed site comparisons (mean bare patch size shown in Table 3) based on their weighted skewness metrics. The three ungrazed exclosures have bare patch characteristics closely similar to the relatively less-disturbed plots 400 to 1050 m distant from the watering points, whereas their (grazed) controls are more similar to the more disturbed plots 200 m distant from the watering points.

In desert grasslands, deterioration of the ecosystem may manifest itself in many ways. Although an increase in the size and frequency of bare patches may be intuitively and qualitatively linked to the deterioration of an ecosystem, other indicators may be even more sensitive to disturbance. For example, in the Chihuahuan Desert rangelands, an early response to disturbance appears to be a shift in vegetation type from long-lived pe-

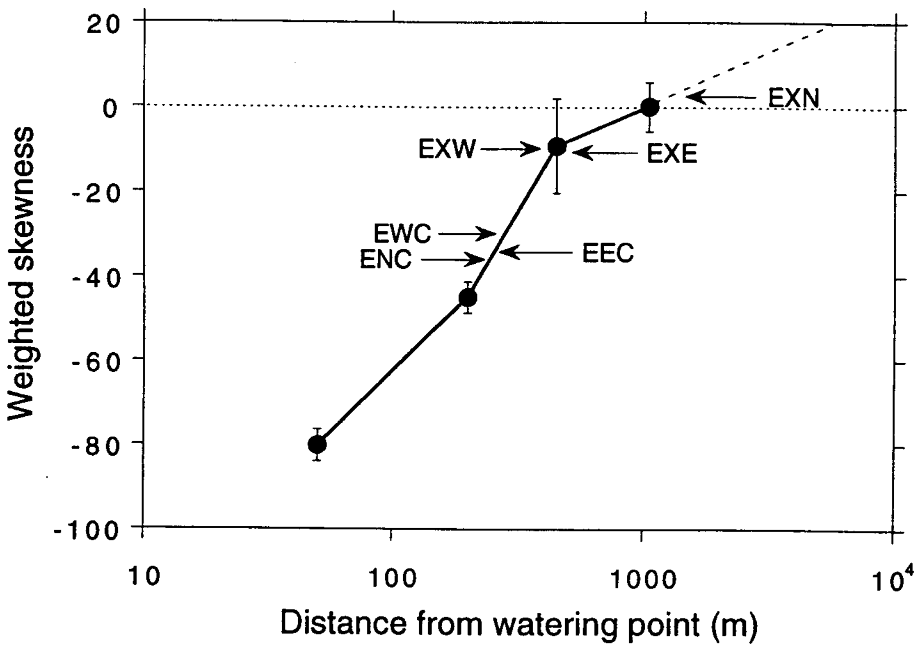

FIGURE 3. Weighted skewness for log transformed data (skewness statistic × mean bare soil patch size) plotted as the mean for the three disturbance gradients. Also shown (arrows) are the relative positions of bare patch weighted skewness for the ungrazed exclosures and the adjacent grazed pastures. Error bars indicate standard error.

rennial grasses to short-lived perennial grasses. Short-lived perennial grasses have a greater biomass turnover rate than do long-lived grasses, and they also have a more clumped morphology, resulting in a less efficient accumulation and retention of soil and water resources. Therefore, an abundance of short-lived perennial grasses relative to long-lived perennial grasses contributes to resource loss which leads to deterioration of the ecosystem.

Long-lived or short-lived grass cover as a proportion of total cover at the three disturbance gradients is shown in Table 4 and Figure 4. Along the disturbance gradients, the least disturbed location had the greatest proportion of long-lived grasses. Short-lived grasses were never a signifi-

TABLE 4

Mean size of bare soil patches, proportion of total cover that is vegetation at the gradient sites, and proportions of total vegetative cover that is composed of long-lived grasses and short-lived grasses

			Proportion of Total Vegetation	
Plot	Mean Bare Patch Size (cm)	ProportionVegetation	Long-Lived	Short-Lived
WW0	197.6	0.1688	0	0.0486
WW1	139.2	0.2402	0.0496	0.0395
WW2	82.1	0.2268	0.3159	0.0595
WW3	41.0	0.3214	0.7700	0.0619
CW0	396.3	0.0641	0	0.0544
CW1	126.0	0.1120	0.0083	0.1665
CW2	74.0	0.1016	0.1802	0.1315
CW3	70.1	0.2514	0.7573	0.0301
MW0	296.4	0.0179	0.0479	0.048
MM1	96.6	0.2048	0.0437	0.0359
MW2	86.9	0.1647	0.0004	0.0510
MW3	39.7	0.2923	0.6518	0.0470

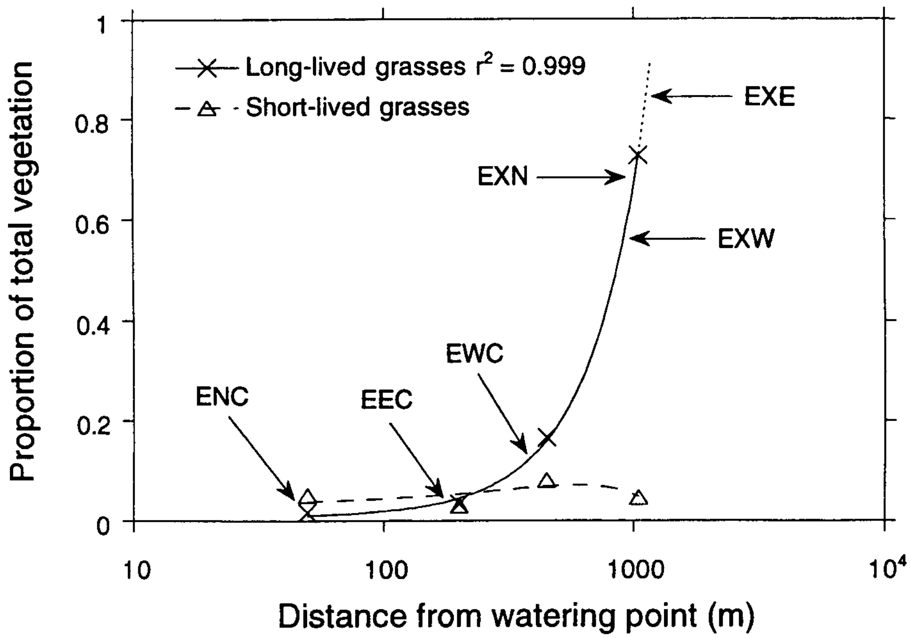

FIGURE 4. Proportions of long- and short-lived perennial grasses plotted as the mean for the three disturbance gradients. The proportion of short-lived grasses shows little change with position along the gradient. The proportion of long-lived grasses is greatest with greatest distance from the origin of the gradient. The relative (extrapolated) positions of long-lived grasses in ungrazed exclosures and adjacent grazed pastures are also shown.

cant proportion of the total vegetation cover, thereby diminishing their usefulness as an indicator. Closer to the watering points, both long-lived and short-lived grasses were an increasingly smaller proportion of total vegetative cover. These data suggest that disturbance leads to a rapid decrease in the proportion of vegetation that is long-lived perennial grasses. When the proportion of long-lived grasses in the grazed–ungrazed comparisons were plotted relative to the disturbance gradient line (Figure 4), the three (grazed) control plot proportions were equivalent to three proportions between 50 and 1050 m. One ungrazed site with the greatest proportion of long-lived grass was plotted on an extrapolated line. This indicator appears to be quite sensitive. Frequency distributions and transformations were not attempted for this data because plant cover was not measured for discrete, individual plants but instead for the contiguous canopies of individual species. Although frequency distributions of plant canopy sizes may yield useful information, the time and resources needed to identify individual genotypes, particularly of grasses, makes it an inviable indicator for general useage.

Average slake test values were higher under vegetation than in bare interspaces in every plot at every site, demonstrating a clear, positive relationship between vegetative cover and soil stability (Table 5). The data were weighted by the relative proportion of bare soil, grass, and shrub to yield an average value for each plot. The weighted averages were uniformly low (< 2) in all but one of the disturbance gradient plots, indicating that the surface soils in these sites are quite unstable and are highly susceptible to erosion. Weighted average values in the exclosures, however, were 2.0 or greater, reflecting weak to moderate soil structure. This comparison suggests that the slake test is very sensitive to the early stages of degradation for these sandy soils but is relatively insensitive to later stages of degradation.

The slake test indicator of soil stability could be made more sensitive to within gradient differences by increasing the resolution at the lower end of the scale (1–3) and by further stratifying the samples by plant species. However, increased sensitivity at the lower end of the scale would not increase the usefulness of the indicator because these differences are functionally insignificant. Any soil with a slake test value of < 2.5 will provide little resistance to detachment by wind or water.

Of the three indicators of rangeland health considered here for highly unstable sandy soils, vegetation type appears to be the most useful, followed by descriptors of bare soil patch size. The slake test of soil surface stability is useful for detecting early changes in rangeland health. In general, long-lived grasses are preferentially grazed by domestic livestock and, with recurrent drought, appear to be the most severely affected. The substantially greater perennial grass cover in the ungrazed plots (~70%) versus adjacent grazed areas (maximum ~25%) suggests that the deteriora-

TABLE 5

Mean values for slake tests on surface and subsurface soil from three strata on the disturbance gradients and from the grazed–ungrazed site pairs

	Surface Soil Slake Index				*Subsurface Soil Slake Index*			
Site	Bare	Grass	Shrub	WMean[1]	Bare	Grass	Shrub	WMean
				Disturbance Gradients				
WW0	1.17	NA[2]	4.11	1.34	1.11	NA	3.78	1.26
WW1	1.00	2.56	5.78	1.87	1.11	1.33	5.56	1.88
WW2	1.00	4.78	5.67	1.91	1.00	3.89	5.11	1.76
WW3	1.00	2.89	5.78	1.69	1.00	2.33	4.33	1.48
CW0	1.06	NA	5.22	1.21	1.11	NA	3.22	1.18
CW1	1.20	1.33	3.78	1.26	1.00	1.33	2.33	1.02
CW2	1.11	2.33	3.22	1.17	1.00	1.56	1.89	1.01
CW3	2.22	4.56	5.78	2.66	1.11	2.89	4.44	1.47
MW0	1.17	NA	4.11	1.19	1.67	NA	3.44	1.64
MW2	1.30	4.22	4.44	1.66	1.60	3.78	4.11	1.53
MW2	1.00	3.67	4.67	1.15	1.00	4.11	4.44	1.18
MW3	1.33	5.11	4.78	2.16	1.00	2.89	3.78	2.40
				Ungrazed–Grazed Comparisons				
EXW	3.67	5.44	4.22	3.29	1.44	3.67	4.56	2.04
EWC	1.22	NA	4.44	1.7	1.00	NA	3.78	1.42
EXE	2.00	4.78	NA	3.06	1.11	3.39	NA	2.00
EEC	1.00	NA	5.22	1.29	1.00	NA	2.78	1.12
EXN	2.44	5.00	4.56	3.42	1.44	4.56	3.78	2.29
ENC	1.00	3.67	4.67	1.83	1.00	3.67	4.76	1.65

[1]WMean, weighted mean.
[2]NA, cover of stratum was < 5%, slake test not done.

tion of rangeland in the Jornada is not due to drought alone and that grazing by livestock has played a major role in exacerbating the effects of drought. As an indicator of rangeland health, the proportion of total vegetation cover that is long-lived grasses appears to be a sensitive and very direct measure of rangeland health. The mean size of bare patches conveys limited albeit useful information about the condition of a rangeland area. By combining the mean of untransformed bare patch size (B_p) with a descriptor of the frequency distribution of bare patch sizes ($LogB_p$) in an area, we derived a metric that was more sensitive to disturbance and less sensitive to rare (and therefore abnormal) data points that could cause the mean of bare patch size to become larger. An example of such an "abnormal" data point would be a large excavation of soil by small burrowing mammals, a natural component of the ecosystem. The slake test of soil surface stability appears to be most useful on these sandy soils for detecting the early stages of degradation.

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