ASSESSING AND MONITORING THE HEALTH OF WESTERN RANGELAND WATERSHEDS

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Abstract. The most important function of watersheds in the western U.S. is the capacity to retain soil and water, thereby providing stability in hydrologic head and minimizing stream sediment loads. Long-term soil and water retention varies directly with vegetation cover. Data on ground cover and plant species composition were collected from 129 sites in the Rio Grande drainage of south-central New Mexico. This area was previously assessed by classification of Advanced Very High Resolution Radiometry (AVHRR) imagery. The classification of irreversibly degraded sites failed to identify most of the severely degraded sites based on size of bare patches and 35% of the sites classified as degraded were healthy based on mean bare patch size and vegetation cover. Previous research showed that an index of unvegetated soil (bare patch size and percent of ground without vegetative cover) was the most robust indicator of the soil and water retention function. Although the regression of mean bare patch size on percent bare ground was significant ($p < 0.001$), percent bare ground accounted for only 11% of the variability in bare patch size. Therefore bare patch size cannot be estimated from data on percent bare ground derived from remote sensing. At sites with less than 25% grass cover, and on sites with more than 15% shrub cover, there were significant relationships between percent bare soil and mean bare patch size ($p < 0.05$). Several other indicators of ecosystem health were related to mean bare patch size: perennial plant species richness ($r = 0.6$, $p < 0.0001$), percent cover of increaser species ($r = 0.5$, $p < 0.0001$) and percent cover of forage useable by livestock ($r = 0.62$, $p < 0.0001$). There was no relationship between bare patch size and cover of species that are toxic to livestock. In order to assess the ability of western rangeland watersheds to retain soil and water using remote sensing, it will be necessary to detect and estimate sizes of bare patches ranging between at least 0.5 m in diameter to several meters in diameter.

Keywords: Soil and water retention, bare patch size, percent bare soil, grass, shrub, remote sensing.

1. Introduction

The ability to accurately monitor and assess ecosystem health is essential in order to guide sustainable management of western rangelands. The landmark National Resource Council publication on ecosystem health established the multi-faceted nature of ecosystem health (National Research Council 1994). This included the need to balance the various facets (functions) of an ecosystem based on an understanding of the potential uses and sustainability of the ecosystem. The central

Environmental Monitoring and Assessment **64**: 153–166, 2000.

function of any healthy ecosystem is the ability of that system to retain soil and water resources. Retention of water and soil is crucial to maintaining the health of upland watersheds. When rangeland watersheds are degraded and fail to retain these resources, perennial streams become ephemeral and carry high sediment loads during episodic floods (Dregne 1983). Properly functioning watersheds retain the maximum amount of water and soil consistent with the production limits imposed by the climate of the region. Soil and water retention is affected by various biotic (e.g., vegetation cover, life-form [grass, shrub, cryptogamic crust], longevity of dominant species) and abiotic (e.g., climate, soil stability, etc.) characteristics of the ecosystem, and several indicators that quantify resource retention capacity have been described (e.g., De Soyza et al. 1998, Whitford et al. 1998).

Indicators derived from ground-based measurements appear to accurately describe ecosystem health for a particular area. However, the heterogeneity of landscapes and the large land area occupied by western rangelands compromises the feasibility of evaluating all rangelands using these methods. Satellite or aerial imagery (remote sensing) may allow more rapid quantification of the health of rangeland landscapes and may provide an economically viable alternative for evaluating landscape ecosystem health. However, the relatively low spatial resolution, the inability to distinguish among plant species, and other factors may compromise the usefulness of remotely sensed imagery for evaluating the potential of an area to retain soil and water resources (i.e., rangeland health). For example, most techniques for utilizing remotely sensed data to describe landscapes rely on the calculation of percent cover by vegetation (generally characterized to the level of major species, lifeform, or major functional group such as C3 [cool season species] and C4 [warm season species]), and its complement, the percent unvegetated area.

Satellite imagery has been classified to distinguish those areas of rangeland that are considered to be irreversibly degraded (Eve et al. 1999). Irreversibly degraded areas in the northern Chihuahuan Desert of North America are characterized by large bare patches surrounding coppiced mesquite (*Prosopis glandulosa*), or between scattered creosotebush (*Larrea tridentata*) shrubs with little understory vegetation. If satellite imagery can accurately detect irreversibly degraded areas, on-the-ground assessments can be reduced accordingly. The accuracy of classification of C3 shrub dominated areas and C4 grass dominated areas was examined by Johnson et al. (in review). In this study we examine the accuracy of remote sensing classification of irreversibly degraded areas based on average size of bare patches.

Unvegetated soil patches (bare patches = fetch length) are important for rangeland health because most soil and water resource loss occurs from these "unprotected" areas (Goff et al. 1993, Skidmore 1986). Resource loss is higher from larger bare patches. Therefore, an area with many small bare patches should retain soil and water resources more efficiently than an area with a few large bare patches. Unfortunately, current methods for rapid assessment of site characteristics, including remote sensing, are unable to accurately measure bare patch sizes, instead using the surrogate percent bare soil as an indicator of ecosystem health. However, an identical percent bare soil value can result from vastly different sizes of bare patches. Here we also investigate the validity of using surrogate indicators, such as percent bare soil, in place of those indicator values that are mechanistically linked to the functioning of ecosystems, such as bare patch size.

Since gross differences in plant community composition may be detectable by remotely sensed imagery, we also studied the potential for improving surrogate indicators by stratifying sites by community type. We also examine the relationships between other indicators of rangeland health (Whitford et al. 1998) and the bare soil parameters. These relationships are used to identify rangeland health indicators that may be obtained from remote sensing measurements of bare soil parameters. Because we intensively sampled the rangeland watersheds of a large portion of the Rio Grande drainage system, we provide estimates of time costs for obtaining and processing ground data for indicators of rangeland health.

2. Methods

Study sites were selected from an area covering much of south-central New Mexico, ranging from 32.00° to 34.21° N and from 105.74° to 107.78° W (Figure 1). Non rangeland areas including montane forests, woodlands, irrigated agriculture and urban areas) were excluded from the study. Remotely sensed images (AVHRR) that had been classified as part of an earlier study (Eve et al.1999) on the basis of changes in the normalized difference vegetation index (NDVI) over the 1990–1993 growing seasons, were used to identify a gradient of pixel classes from grassland dominated to increasingly shrub-dominated sites. We selected 129 sites at random to obtain a representation of the five gradient classes (Turner et al. in review, Johnson et al. in review). We surveyed the vegetation at 109 of these sites in 1996 and at an additional 20 sites in 1998.

Study sites were positioned at the centers of the selected pixels using a Global Positioning System (GPS) with differential correction. At each site, perennial vegetative cover was measured along three 100 m transects radiating out from the plot center, 120° apart, with the direction for the first transect chosen at random. We recorded the identity of plant species or unvegetated area intercepting the transect, measured to the nearest 1 cm (Canfield 1941, DeSoyza et al. 1997, DeSoyza et al. 1998). We differentiated between living and standing dead vegetation and between dominant overstory and understory canopies, and the current study is based solely on live plants of the overstory. The initial vegetation survey was conducted from April through July 1996 and the additional 20 grassland sites were surveyed in April 1998.

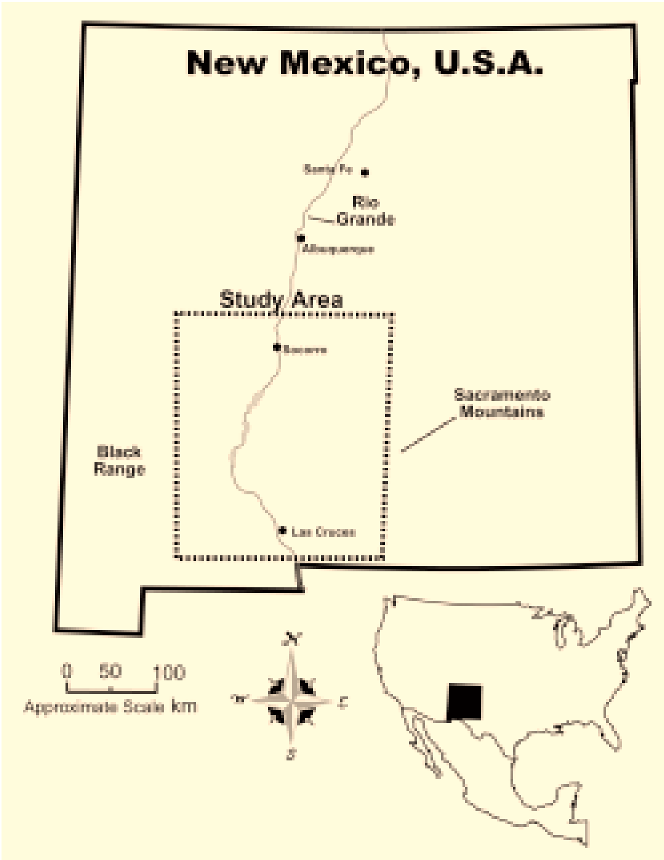

Figure 1. *A map of the study area in which 129 sites were selected for intensive measurement of vegetation cover and composition and soil surface properties.*

Using computer-based spread sheets, we calculated basic univariate statistics (mean, standard deviation, coefficient of variation) on the percent cover by species, life form, bare soil, litter, and cryptogams for each site. We investigated relationships between variables that can be measured remotely [e.g., percent unvegetated (bare area)] and those not available from the remotely sensed images, but which are better indicators of ecosystem function (e.g., mean size of bare patches) using simple linear regression models. We also used linear regression models to examine the relationships between other indicators of rangeland health (Whitford et al. 1999) and mean size of bare patches and percent cover of bare ground.

In order to investigate the relationship between vegetation type and bare soil indicators, sites were classified by dominant vegetation life form. These data were subsequently examined by regression analysis. Sites were assigned to four classes (grass, shrub, grass + shrub, or none) based on the occurrence of at least 5% grass cover only (a grass site), or 5 % shrub cover (a shrub site) or 5% grass cover and

5% shrub cover (a grass + shrub site). Sites that did not have a minimum of 5% grass cover or 5% shrub cover were assigned to the none category. Thus each of the 129 sites were assigned to one of four cover-type classes. This procedure was repeated with all 129 sites used at each iteration. Thus we used 10% minimum cover, 15% minimum cover etc. to 30% minimum cover. We did not use minimum cover values of 35% or greater because at these cover classes all 129 sites fell into the none category.

In order to validate the relationships between percent bare cover and mean bare patch size we used line transect measurements of sites reported in earlier studies (DeSoyza et al. 1997, DeSoyza et al. 1998) which were not used in deriving the current regressions. These sites were chosen to include a range of mean bare patch sizes, percent bare soil, and percent grass and shrub cover. The measured percent bare soil value for each validation site was used in the appropriate regression model to predict a mean bare patch size, and its percent error was calculated as: (Actual mean bare - Predicted mean bare x 100) ÷ (Actual mean bare).

3. Results

The relationship between mean bare patch size and total percent bare (not covered by vegetation) was: mean bare patch (cm) = 64.6 + 0.041 percent cover unvegetated (Figure 2). This linear regression model yielded a significant relationship ($P < 0.0001$), but explained only 11% of the variability between mean bare patch size and percent bare ($r^2 = 0.11$). These data also show that some sites that were classified as irreversibly degraded by Eve et al. (1999) are characterized by small mean bare patches and that many sites characterized by very large bare patches were not classified as irreversibly degraded (Figure 2).

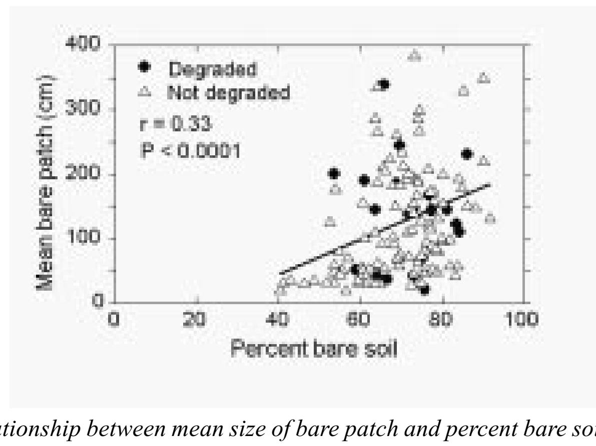

Figure 2. The relationship between mean size of bare patch and percent bare soil at 129 rangeland sites in south-central New Mexico. Dark circles represent sites that were classified as irreversibly degraded, and triangles represent sites that were classified as not irreversibly degraded by Eve et al. (1999).

When these data were stratified by dominant vegetative life-forms, forbs and succulents had very low cover and were always less than 5% cover. The results of increasing the cutoff value for percent cover for the four life-form classes (grass, shrub, grass+shrub, none) by 5% increments is shown in Figure 3. Increasing the minimum percent cover needed to assign sites to a vegetative life-form class increased clustering of grass, shrub and none sites (Figure 3). At high cutoff percentages, the shrub and none sites occupied a similar range of mean bare patch sizes, but the shrub sites had lower percent bare soil (Figure 3).

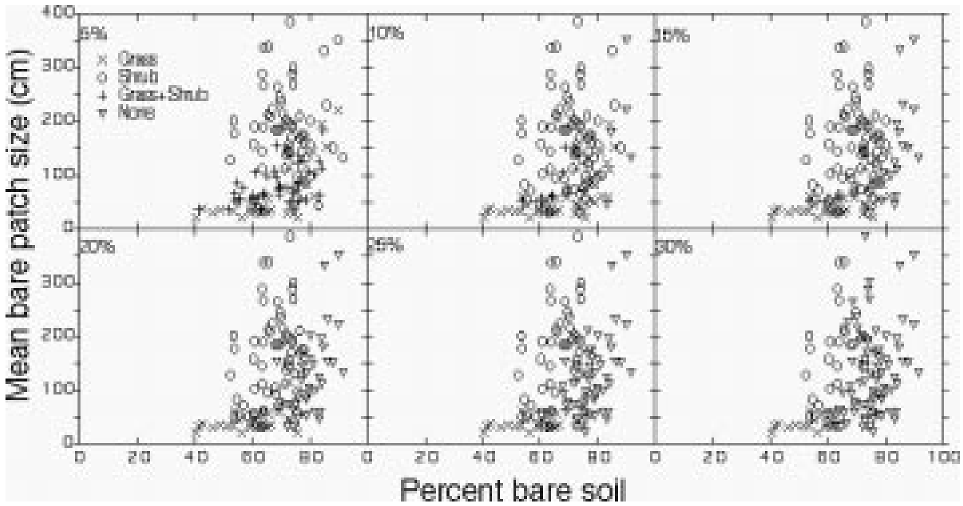

Figure 3. *The relationships of mean size of bare patches and percent bare soil at sites categorized by minimum cover values ranging from 5% to 30%. Each cover category summarizes data for 129 sites. Not all cover categories are included in each panel because of the small sample size (see Table I).*

A simple linear regression model yielded significant relationships between percent bare soil and mean size of bare patches for the four life-form classes at several minimum cover classes (Table I). Notably, grass and none sites yielded the best relationships (highest correlation coefficients, r) when site assignment was based on lower minimum cover values, and shrub sites yielded the best relationships when site assignment was based on higher minimum cover values (Table I).

Although the composition of the site classes changed with different minimum cover values, the regression slopes of the Grass sites remained relatively constant (Figure 4). However, regression slopes of the Shrub and None classes changed substantially with changing minimum cover values (Figure 4). Each series of regression lines appears to have a pivotal point: approximately 60% bare and 50 cm

Table I

Results of simple linear regressions on percent bare soil versus mean bare patch size for 129 Chihuahuan Desert sites. Each site was assigned to a grass, shrub, grass + shrub or none category based on a minimum cover of 5%. This was repeated for all 129 sites at 10%, 15%, 20%, 25%, and 30% minimum covers respectively. Significant relationships ($p < 0.05$) were due to positive relationships between percent bare soil and mean bare patch size. n.a. indicates regressions that were not applicable due to small sample size.

Cover	Statistics	Grass	Shrub	Grass + Shrub	None
5%	df	25	60	41	n.a.
	r	0.70	0.11	0.48	
	p	< 0.0001	0.40	0.001	
10%	df	41	71	9	4
	r	0.63	0.04	0.51	0.55
	p	0.0001	0.76	0.13	0.33
15%	df	33	70	4	20
	r	0.51	0.03	0.76	0.63
	p	0.002	0.79	0.24	0.003
20%	df	29	59	n.a.	39
	r	0.41	0.29		0.56
	p	0.028	0.025		0.0001
25%	df	21	42	n.a.	63
	r	0.28	0.51		0.52
	p	0.20	0.0005		<0.0001
30%	df	15	21	n.a.	91
	r	0.30	0.58		0.34
	p	0.26	0.004		0.001

mean bare patch size for Grass sites; 55% bare and 125 cm bare patch size for Shrub sites (Figure 4). Thus changing the minimum cover requirements for site categorization affects the three site types differently. Shrub site mean bare patch size mostly increases as shrub cover increases while, to a lesser extent, the mean bare patch size of Grass sites decreases as grass cover increases (Figure 4).

The results of applying the appropriate regressions (based on minimum cover class) to the grazing gradient sites and to the grazed and ungrazed sites in the Chihuahuan Desert described by DeSoyza et al. (1998), and their percent error are shown in Table II. Even though several predicted mean bare patch sizes were similar to the actual measured values, the percent error ranged from 225.1 to -142.4% (Table II). Sites with high grass cover (site numbers 15 and 17) had some of the least predictable mean bare patch sizes (Table II). With the exception of three sites

Table II

Percent error of mean bare patch sizes predicted for 20 Chihuahuan Desert sites using simple linear regression models generated from the six cover-based (5% to 30%) site classifications (See Table I, Figure 3). Predicted mean bare patch size was not calculated for classes that did not have statistically significant regression models (= n.a.).

	Mean	Percent			Predicted mean bare patch sizes						Precent error					
Site	bare	Bare	Grass	Shrub	5%	10%	15%	20%	25%	30%	5%	10%	15%	20%	25%	30%
1	292.4	93.6	0.47	4.29	n.a.	n.a.	n.a.	216.1	202.1	200.5	n.a.	n.a.	n.a.	26.1	30.9	31.4
2	140.9	88.8	2.44	4.35	n.a.	n.a.	n.a.	183.8	176.9	182.0	n.a.	n.a.	n.a.	-30.4	-25.6	-29.2
3	127.0	89.8	4.58	1.97	n.a.	n.a.	n.a.	190.8	182.4	186.0	n.a.	n.a.	n.a.	-50.3	-43.7	-46.5
4	68.5	74.9	20.88	0.83	86.8	72.4	60.3	52.8	103.9	128.4	-26.6	-5.6	12.1	23.0	-51.5	-87.3
5	147.6	92.1	1.23	3.0	n.a.	n.a.	n.a.	206.0	194.2	194.7	n.a.	n.a.	n.a.	-39.5	-31.6	-31.9
6	102.2	79.5	4.12	9.62	n.a.	n.a.	n.a.	121.2	128.3	146.3	n.a.	n.a.	n.a.	-18.6	-25.6	-43.2
7	90.2	83.5	5.73	2.17	111.9	n.a.	n.a.	148.2	149.3	161.7	-24.1	n.a	n.a	-64.4	-65.6	-79.4
8	62.6	70.8	23.75	2.64	74.9	64.5	55.5	49.7	82.4	112.6	-19.8	-3.13	11.3	20.4	-31.7	-80.0
9	162.4	83.1	0.88	8.98	n.a.	n.a.	n.a.	145.5	147.2	160.2	n.a.	n.a.	n.a.	10.4	9.4	1.4
10	102.2	75.9	3.32	17.91	n.a.	n.a.	n.a.	97.3	109.7	132.7	n.a.	n.a.	n.a.	4.7	-7.4	-29.9
11	133.1	77.3	8.9	12.42	86.9	n.a.	n.a.	106.4	116.7	137.8	34.73	n.a.	n.a.	20.1	12.3	-3.6
12	50.9	67.9	27.64	3.69	66.5	58.9	52.1	47.6	40.7	101.4	-30.6	-15.7	-2.4	6.6	20.1	-99.2
13	49.5	53.3	41.61	2.23	24.3	31.0	35.3	36.7	35.7	36.0	50.8	37.4	28.8	25.9	27.9	27.3
14	171.1	93.1	2.22	4.64	n.a.	n.a.	n.a.	213.0	199.6	198.7	n.a.	n.a.	n.a.	-24.5	-16.7	-16.1
15	50.2	23.2	53.37	4.04	-62.9	-26.8	0.4	14.2	25.4	24.2	225.1	153.4	99.3	71.8	49.4	51.9
16	90.2	83.5	5.73	2.17	111.9	n.a.	n.a.	148.2	149.3	161.7	-24.1	n.a.	n.a.	-64.4	-65.6	-79.4
17	125.1	17.6	47.89	0.66	-79.1	-37.6	-6.1	10.0	23.5	22.0	163.2	130.1	104.9	92.0	81.2	82.4
18	54.1	75.6	6.72	17.45	84.1	n.a.	n.a.	94.4	107.5	131.0	-55.6	n.a.	n.a.	-74.7	-98.8	-142.4
19	311.9	73.5	1.76	24.77	n.a.	n.a.	n.a.	186.8	96.6	123.0	n.a.	n.a.	n.a.	40.1	69.0	60.6
20	313.7	62.3	1.64	36.06	n.a.	n.a.	n.a.	149.2	162.7	187.5	n.a.	n.a.	n.a.	52.5	48.1	40.2

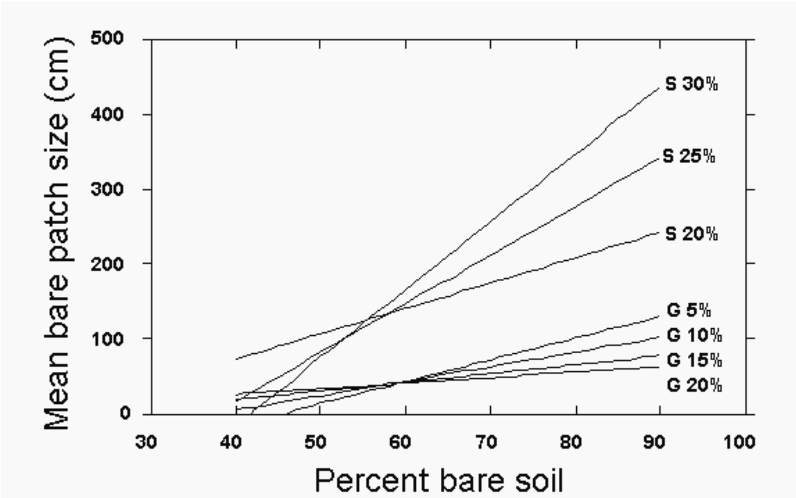

Figure 4. *The regression lines for mean bare patch size as a function of percent bare soil for minimum cover values of grass and shrubs as indicated on the figure. S indicates shrub sites and G indicates grass sites*

(numbers 15, 17, and 18) mean bare patch size could be predicted to within ± 50% error or less for the remaining 17 sites; and within ± 25% for 10 sites (Table II).

Regression analysis of other indicators derived from the data on plant cover and species composition on mean bare patch size and on percent bare cover showed that several other indicators could reasonably be predicted from mean bare patch size but not from percent bare ground (Figure 5). Three of the indicators predicted from mean bare patch size were significant (species richness, percent cover of increasers, and percent cover of palatable species) and showed that bare patch size accounted for between 25% and 36% of the variability in those indicators. There was no significant relationship between cover of toxic plant species and mean bare patch size or percent bare ground. There were no non-native (exotic) weed species that occurred at sufficiently high densities to provide data for examining relationships between bare patch size and cover of exotics.

4. Discussion

The concept of ecosystem health as a single descriptive value has proved to be an elusive goal. More attainable are evaluations of the functional components of ecosystem health, such as biodiversity, productivity, etc. Of these functions, the ability of an area to retain soil and water resources is paramount to its ability to fulfill any other ecosystem function. For example, there is a feedback between sufficient plant cover to retain water and nutrients *in situ* and primary production (Ludwig 1987). Over the past few years several ground-based indicators that describe the

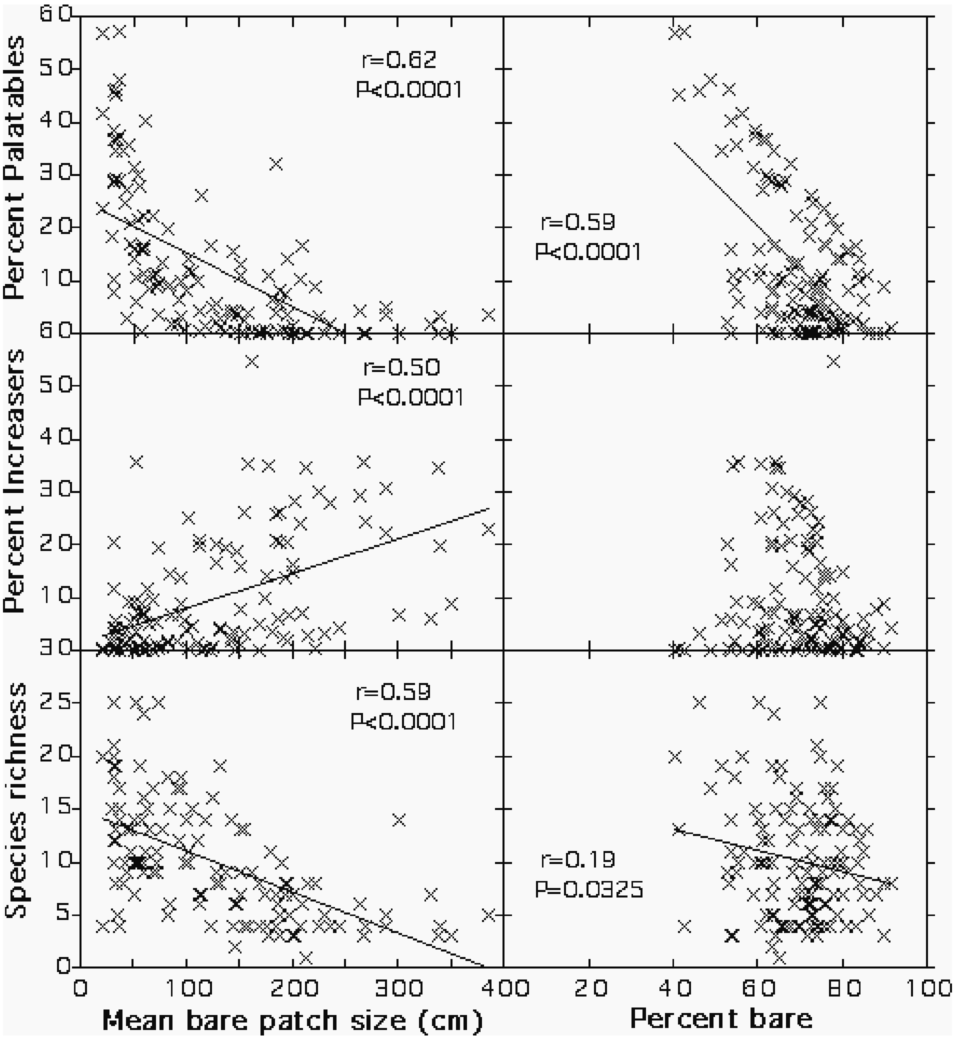

Figure 5. *The relationship between mean bare patch size (cm) and total percent bare ground with three indicators of rangeland health. Percent cover of palatables refers to those species that are used as forage by domestic livestock; percent cover of increasers refers to those species of shrubs that established in areas of grassland following the development of the commercial livestock industry in the Chihuahuan Desert.*

ability of a site to retain soil and water resources have been developed. These include vegetation indicators such as the cover of long-lived perennial grasses present at a site (DeSoyza et al. 1997), a soil stability test (Herrick et al. 1999), and others (e.g., see Whitford et al. 1998, DeSoyza et al. 1998). However, while several effective ground-based indicators have been identified, there are time and human resources constraints on the use of ground-based measurements for assessing or monitoring the health of rangeland watersheds.

Western U.S. rangeland landscapes are very heterogeneous. A large number of ground-based site assessments are necessary for a statistically valid assessment of the condition or health of the ecosystems within a region. Data collection for calculation of indicators for the 129 sites used in this study required 60 days for a team of 4 scientists. Most of the time commitment was for travel from the base location to and between sites. Data entry, quality control checks of the data, and initial data processing required an additional 60 person-days. This example documents the time and human resource commitment necessary to conduct an assessment in a relatively small region in the western U. S. Because of the potentially large amount of time and resources needed to conduct these assessments, it may be difficult for State and Federal agencies, that have been mandated to make assessments of ecosystem health, to commit the resources necessary for a reliable on-the-ground assessment.

Remotely sensed satellite or aerial imagery may offer a more cost effective means to quantify the vegetation and structural composition of landscapes. However, current methods of remote sensing are unable to resolve site characteristics such as bare patch size which are most closely linked to the soil nutrient and water resource function. In order to obtain data on the health of watersheds using remote sensing, it will be necessary to derive information on size and distribution of bare patches from the imagery.

Our results suggest that using empirically derived simple regression models to predict mean bare patch size from percent bare soil is unreliable, and it is likely to result in an erroneous conclusion about the soil and water resource function of a site. The initial analysis for the 129 test sites yielded a significant regression model, but since the model explained only 11% of the variability in bare patch size, it would not be suitable for precise predictions of bare patch size.

The wide variation in mean bare patch size of sites that were classified as irreversibly degraded by Eve et al. (1999) serves as an accuracy assessment of that imagery classification. Thirty-five percent of the sites classified as irreversibly degraded based on vegetation would not be classified as degraded based on mean size of bare patches. Forty–one percent of the sites classified as not irreversibly degraded were characterized by mean bare patch sizes greater than 50 cm [higher than the emperically determined threshold fetch length for soil erosion (Herrick, unpublished data)]. Sites having mean bare patch sizes greater than 50 cm are degraded and will continue to lose soil by wind and water erosion. The data from this study demonstrate that even limiting classification of sites to irreversibly degraded by using irreversibly degraded sites as training sites for classifying AVHRR imagery, is not feasible.

Although the on-the-ground sampling in this study was stratified by vegetation cover based on classified imagery (Eve et al. 1999), the mean bare patch size was 25 cm or less at only 4% of the sites and was 50 cm or less at only 22.5 % of the sites. Measurements of fetch distances for initiating saltatory movement of sand at wind velocities of 40 km h^{-1} in desert grassland indicated that 25 cm was the critical threshold (Herrick J. E. unpublished data). Bare patches with fetch distances greater than 50 cm would be subject to several episodes of wind erosion every year and would also be subject to soil loss by water erosion. Thus, based on a stratified random sampling scheme used in this study, seventy eight percent of the rangeland watershed areas in the central Rio Grande drainage would warrant classification as unstable or not healthy.

We used vegetation life-form cover to further classify the sites because this could be done from remotely sensed data. All perennial grass species detected at the test sites were of the C4 physiological type, while all the perennial shrubs were of the C3 type, and cover by perennial forbs and succulents was insignificant. Due to the different phenological and spectral characteristics of C4 grasses and C3 shrubs, the cover of these life-forms could be detected relatively easily from remotely sensed imagery (Peters et al. 1997). The life-form classification of sites yielded some interesting, although not unexpected, insights into the vegetational dynamics of Chihuahuan Desert rangelands. For example, other than at the low, 5% cover level (5% grass and 5% shrub), few sites supported grasses and shrubs at higher levels of cover. However, many sites had grass or shrub cover over 25%. The absence of sites with high cover of both grass and shrubs supports the hypothesis that shrubs and grasses cannot coexist at high densities, and that shrub encroachment into desert grasslands often results in the displacement of grasses (Buffington and Herbel 1965).

Perhaps most surprising was the lack of a significant relationship between percent bare and mean bare patch size for sites with greater than 25% grass cover. It appears that once a threshold of grass cover (approximately 25%) has been reached, further increases in grass cover result in little change in mean bare patch size. This can occur through a change in the size distribution of bare patches, where large bare patches are replaced by several smaller bare patches. While further increases in grass cover would eventually cause mean bare patch size to decrease, systems with very high grass cover may not exist in desert rangelands. Thus, in these Chihuahuan Desert ecosystems, 25% grass cover may represent a threshold between sites that are able to retain soil nutrient and water resources, and those with less grass cover and larger bare patches that are unable to do so.

The correlation between mean bare patch size and indicators such as cover of plant species useable by livestock, and plant species richness, is important in evaluating the potential for remote sensing of rangeland health. These indicators provide information on the biodiversity, potential for further degradation, and economic health of rangelands. This type of information is frequently required by land managers and policy makers. If it were possible to obtain information on bare patch sizes from remote sensing platforms, this study shows that these other indicators can be derived from this parameter with a reasonable level of accuracy.

This study has provided a rich data set for testing the accuracy of indicators derived from remote sensing platforms. This study and previous studies have shown that the most valuable indicator of rangeland health is size of bare patches. Since this parameter is related to many other important attributes of rangeland ecosystems, it is the most useful and reliable indicator of rangeland health. If we are to make accurate assessments of rangeland health based on remote sensing in the future, it will be necessary to derive information on bare patch sizes and not just percent of total area that has no vegetation cover.

Acknowledgements

The environmental commands at White Sands Missle Range and Fort Bliss provided assistance and access to research sites on these military reservations. Tom Waddell (Amendaris Ranch) and Steve Dubrott (Ladder Ranch) permitted access to the lands they manage. Anne Neale (U.S. EPA Las Vegas) assisted with field work. The research described herein was partially funded by the United States Environmental Protection Agency through its Office of Research and Development. The research described herein as been subjected to the Agency's peer and administrative review and approved as an EPA publication.

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