Application of Soil Quality to Monitoring and Management: Paradigms from Rangeland Ecology

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ABSTRACT

Recent interest in soil quality and rangeland health, and the large areas set aside under the USDA Conservation Reserve Program, have contributed to a gradual convergence of assessment, monitoring, and management approaches in croplands and rangelands. The objective of this paper is to describe a basis for integrating soils and soil quality into rangeland monitoring, and through monitoring, into management. Previous attempts to integrate soil indicators into rangeland monitoring programs have often failed due to a lack of understanding of how to apply those indicators to ecosystem function and management. We discuss four guidelines that we have used to select and interpret soil and soil quality indicators in rangelands and illustrate them using a recently developed rangeland monitoring system. The guidelines include (i) identifying a suite of indicators that are consistently correlated with the functional status of one or more critical ecosystem processes, including those related to soil stability, soil water infiltration, and the capacity of the ecosystem to recover following disturbance; (ii) basing indicator selection on inherent soil and site characteristics and on site- or project-specific resource concerns, such as erosion or species invasion; (iii) using spatial variability in developing and interpreting indicators to make them more representative of ecological processes; and (iv) interpreting indicators in the context of an understanding of dynamic, nonlinear ecological processes defined by thresholds. The approach defined by these guidelines may serve as a paradigm for applying the soil quality concept in other ecosystems, including forests and ecosystems managed for annual and perennial crop production.

While farmers often characterize long-term trends in their land in terms of soil productivity, ranchers are more likely to evaluate changes in the dominant vegetation. These different perspectives reflect the different approaches to assessing and monitoring croplands and rangelands. Recent interest in soil quality and rangeland health, and the large previously cropped areas set aside under the USDA Conservation Reserve Program, have contributed to a gradual convergence of assessment, monitoring, and management approaches in croplands and rangelands. Many farmers enrolled in the Conservation Reserve Program, who have traditionally managed annual monocultures, are now managing perennial polycultures. The objective of this paper is to describe some of the ways in which soils and soil quality are being integrated into rangeland monitoring, and through monitoring, into management. This integration may serve as a paradigm for applying the soil quality concept in other areas.

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The concept of soil quality was developed in response to public demand for an increased emphasis on sustainability and to a recognition by many in the scientific community that soil management could be improved by taking a more holistic, integrative approach to soils. These concerns are reflected in SSSA's definition of soil quality: "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation" (SSSA, 1997).

The concept of rangeland health was developed in response to similar concerns. Rangeland health is defined as, "the degree to which the integrity of the soil and the ecological processes of rangeland ecosystems are sustained" (Natl. Res. Counc., 1994). Rangeland monitoring and assessment systems have traditionally focused heavily on plant community composition and productivity. Recent interest in rangeland health and a growing recognition of the importance of soil–vegetation feedbacks in structuring rangelands (Schlesinger et al., 1990; Tongway and Ludwig, 1994) have led to a renewed interest in integrating soil information into rangeland monitoring and management.

We have found the following guidelines to be useful in developing integrated soil–vegetation monitoring and management systems for rangelands:

1. Identify a suite of indicators that are consistently correlated with the functional status of one or more critical ecosystem processes.
2. Base indicator selection on site- or project-specific resource concerns and inherent soil and site characteristics.
3. Use spatial variability in developing and interpreting indicators to make them more representative of ecological processes.
4. Interpret indicators in the context of an understanding of dynamic, nonlinear ecological processes.

In addition to these guidelines, measurements included in monitoring and assessment systems need to be rapid, simple, inexpensive, and repeatable. To the extent possible, indicators should be predictive: They should reflect early changes in ecological processes and indicate that a more significant change is likely to occur. Each of the four guidelines above is illustrated below using a monitoring system that was recently developed through an informal interagency collaborative effort led by USDA-ARS (Herrick and Whitford, 1999). This monitoring system is designed to detect long-term trends in three attributes: soil and site stability, hydrologic function, and the biotic integrity of the system.

Abbreviations: SOM, soil organic matter.

Biotic integrity is defined as the capacity of the system to resist and recover from catastrophic disturbance.

IDENTIFY INDICATORS THAT ARE CONSISTENTLY CORRELATED WITH FUNCTION

In order to be cost-effective, indicators must add value to a monitoring program by providing information about the functioning of the system that cannot be derived directly from knowledge of the management system (Brown et al., 1998). If an indicator is consistently correlated with both management and a critical ecosystem function, then simple knowledge of the management practice or system can be used to replace the measured indicator at much lower cost. If the indicator is correlated with management, but not consistently correlated with a critical function, then the indicator may be erroneously used to support preconceptions about the superiority of one management system over another. Indicators need to be consistently correlated with some ecosystem function (e.g., plant productivity or retention of soil and water resources or biodiversity conservation). In the case of erosion, the function is erosion resistance, and all parameters used in erosion models are either directly or indirectly compared to measured erosion rates. An indicator is of little value for management if it lags behind the process of interest. However, the indicators should also be reflective of actual changes in the system, rather than changes that are assumed to follow from changes in management.

We have identified a suite of indicators for use on rangelands based on previously published studies, new research, and expert knowledge about the variability in relationships across diverse rangeland ecosystems. An ongoing research program includes testing and calibrating these indicators directly to ecosystem processes and functions, developing complementary landscape-level indicators, and generating more effective interpretation tools. The indicators are calculated from three core measurements and a number of supplementary measurements (Table 1). Each of the core measurements can be used to generate a suite of indicators, which previous studies have shown are related to each of three attributes or criteria: soil and site stability, watershed function, and biotic integrity (Table 1). The supplementary measurements are applied depending on resource concerns and site characteristics (see below) and are used to generate indicators that generally apply to just one or two of the three attributes.

Table 1. Selected quantitative measurements and their relevance to each of three landscape attributes.

Measurement	Soil and site stability	Hydrologic function	Biotic integrity
CORE			
1. Plant cover and composition using line-point intercept	X	X	X
2. Canopy gaps using continuous line intercept (minimum 20 cm between canopy elements)	X	X	X
3. Soil stability test	X	X	X
SUPPLEMENTARY			
4. Belt transects for woody and invasive plants	X	X	X
5. Species richness			X
6. Plant production by species (double sampling)		X	X
7. Impact penetrometer		X	X
8. Single-ring infiltration		X	X
9. Riparian channel vegetation survey	X	X	X
10. Riparian channel profile	X	X	X
11. Tree density	X	X	X

The core measurements include line-point intercept, continuous line intercept, and an aggregate stability test. The line-point intercept is used to quantify plant cover and composition and soil surface characteristics (Bonham, 1989). This measurement is used to generate a number of indicators, including bare ground, which is highly correlated with both runoff and susceptibility to water erosion (Smith and Wischmeier, 1962; Blackburn and Pierson, Jr., 1994), and basal cover, which is related to overland flow path length and to the capacity of the system to recover following overgrazing (Herbel et al., 1972; Gutierrez and Hernandez, 1996). Increasing overland flow path length increases the amount of time available for infiltration to occur. Microbiotic crust cover can be calculated separately from the line-point intercept data for systems in which these organisms play an important role in stabilizing the soil surface (Eldridge and Kinnell, 1997; Belnap and Gillette, 1998). Noncanopy patches larger than a minimum diameter (e.g., 20 cm) are recorded along a continuous line intercept. These patches cannot be detected using the line-point intercept method and are highly correlated with susceptibility to wind and water erosion and to the invasion of some species that change vegetation structure (Gould, 1982; Musick and Gillette, 1990). The size of canopy gaps is also an indicator of the relative uniformity of soil resource distribution (Schlesinger et al., 1990). The third core method is a field aggregate stability test (Herrick et al., 2001). This test is used to rate water-stable aggregation on a scale of 1 (slakes immediately) to 6 (75% remains on 1.5-mm screen following sieving) for soil surface fragments that are 6 to 8 mm in diameter. The method is highly correlated with laboratory measurements of aggregate stability (Herrick et al., 2001), which in turn, have been negatively correlated with interrill soil erosion in the field (Blackburn and Pierson, Jr., 1994). We have found that aggregate stability, as determined by this method, is relatively insensitive to intensive short-term disturbances, such as trampling by horses, humans, and vehicles, but reflects longer-term changes in soil structure. Insensitivity to single disturbance events is critical to ensure that monitoring results do not simply reflect normal variability in the system. At the same time, the use of a larger sieve size (1.5 mm) increases the probability of detecting change at an early stage because aggregation at this scale is generally controlled by rapidly cycling organic matter (Tisdall and Oades, 1982). Soil aggregate stability at the surface is particularly important in plant canopy interspaces when rock and litter cover are minimal because there is no protection from raindrop impact. We recommend calculating aggregate stability, as well as rock, microbiotic crust, and litter cover, separately for plant canopy and intercanopy spaces.

The eight supplementary methods (Table 1) have also been correlated with ecosystem function. The belt transect is used for early detection of invasive species, both native and exotic. Many of these species, including mesquite [*Prosopis glandulosa* Torr.], juniper [*Juniperus* spp.], Lehmann's lovegrass [*Eragrostis lehmanniana* Nees], and cheatgrass [*Bromus tectorum* L.], have been shown to be associated with dramatic changes in soil quality, as reflected in changes in C- and nutrient-cycling processes (Barth and Klemmedson, 1982; Schlesinger et al., 1996; Connin et al., 1997; Arredondo and Johnson, 1999), soil erosion (Davenport et al., 1998), and infiltration capacity (Reid et al., 1999). However, the establishment of these species is often simply an indication that a change has already occurred (Brown and Archer, 1999), significantly reducing their value as indicators. Species richness is a direct measure of the number of species present on a site, calculated using a species area curve based on counts in plots of different sizes (Stohlgren et al., 1995). Plant production using double sampling is normally used for assessment only as it is relatively imprecise and varies dramatically among years.

Two supplementary measurements are specifically designed to measure soil properties that are often closely related to function. The impact penetrometer (Herrick and Jones, 2002) is similar to a standard Corps of Engineers strain gauge penetrometer (Bradford, 1986), except that repeated blows of a 2-kg mass dropped from a standard height replace human force. The use of strain gauge penetrometers is limited by relatively high cost, repeatability problems associated with the need to maintain a constant rate of insertion, and difficulties in comparing data from penetrometers designed for different ranges of soil strength (Fritton, 1990; Vyn and Raimbault, 1993). Furthermore, human strength is often insufficient to use strain gauge penetrometers in dry, uncultivated soils. The impact penetrometer overcomes these limitations. Impact penetrometers can be fabricated by a machine shop for $100, energy is consistently applied, and direct comparisons can be made between measurements made using different drop heights. Drop height is increased in dry, uncultivated soils to reduce the number of strikes and, therefore, the amount of time required. The impact penetrometer does, of course, share several limitations with all penetrometers: Measurements depend on moisture content and cannot be directly related to bulk density. However, it can be easily used to reliably monitor relative changes in compaction over time, provided that moisture content is constant.

The single-ring infiltrometer is similar to that described in Bouwer (1986), except that a constant water depth is maintained by an inverted bottle with an air supply tube inserted to the desired water depth. One of the reasons that the penetrometer and infiltrometer are listed as supplementary methods is that the relationships to ecosystem functions appear to be inconsistent in some plant communities, such as blue grama [*Bouteloua gracilis* (H.B.K.) Lag] grasslands. For example, penetrometer resistance often increases under blue grama grass canopy due to the high root density near the soil surface (Table 2). The final set of supplementary measurements apply specifically to riparian and woodland systems: riparian channel vegetation survey and channel survey and tree density.

Additional measurements are listed, with references to appropriate procedures, to address site-specific resource problems, including salinization and high alkalinity or acidity. These problems, when they occur in rangelands, are often a function of parent material rather than management, and consequently are not included in monitoring. They can, however, be extremely important in the case of abandoned agricultural land and in areas that have been heavily disturbed by mining activities.

BASE INDICATOR SELECTION ON RESOURCE CONCERNS AND SITE CHARACTERISTICS

The core measurements listed above can be used to generate management-relevant indicators in many situations at both local and regional scales. However, monitoring efficiency at the ranch or small-watershed level can often be increased by selecting only those supplementary indicators that are most sensitive to site-specific changes in ecosystem function and that are relevant given the soil and site characteristics. For example, penetrometer resistance is of little value on coarse-textured upland sites with little potential for intensive animal or vehicular impact.

Given that societal values, as well as scientific understanding, change over time, monitoring programs should be designed to quantify the potential of the system to (i) function in support of a range of societal values rather than to support any individual value, (ii) resist degradation, and (iii) recover following degradation. The premise that the capacity of an individual site to function depends on a core set of processes is common to most definitions of both soil quality and rangeland health. The core measurements were selected to generate indicators of these processes. Indicators that address specific values or land uses, such as livestock forage or wildlife habitat, can often be calculated from the core measurements (Table 1), and additional measurements can be included. In rangelands, however, we have found that it is quite useful to maintain a distinction between

Table 2. Penetrometer resistance under blue grama canopy and in intercanopy spaces for randomly selected points inside and outside of a 4-ha livestock exclosure† in southern Otero County, NM, USA. Data reflect number of strikes required to drive a 2-kg mass dropped from a height of 60 cm to drive a standard Corps of Engineers penetrometer 5 cm into dry soil at each of three depths.

Depth	Bare zone (intercanopy)	Grass zone (intracanopy)	*P*-value
cm	Replications		
	66	54	
	Mean (±SE)		
0–5	6.6 (0.3)	7.3 (0.3)	0.091
5–10	9.0 (0.3)	11.3 (0.4)	<0.001
10–15	9.3 (0.4)	11.5 (0.6)	0.001

† Data for both treatments are combined due to lack of exclosure effect ($P \geq 0.1$ for all depths).

core and supplementary measurements to minimize unnecessary cost increases. The primary criterion for indicator selection must be the strength and consistency of its relationship to a critical process while recognizing that the relative importance of different ecological processes, and the strength of the relationship between indicator and process, varies among soils, landscape positions, and regions. In some cases, relationships between indicators and processes can be inferred from the literature while in others, they must be quantified with new studies.

Termites illustrate how the relationship between an indicator and a property can vary at both the landscape and regional scale due to differences in the relative importance of processes that the termites affect and the relative effects of termites on those processes. Termites are often proposed by progressive land managers as a valuable biological indicator in the southwestern USA, particularly after the managers learn of termites' contribution to dung decomposition, nutrient cycling, and macropore formation.

Removal of termites from plots on a Chihuahuan Desert bajada resulted in a 42% reduction in saturated infiltration rate in plant interspaces (88.4–51.3 mm h^{-1}) 4 yr after termite exclusion (Elkins et al., 1986). A similar experiment completed in West Texas generated opposite results. The depth of water infiltrating the soil over a period of 40 min was 23% higher in termite removal plots (18.5 mm) than in controls (15 mm) 2 to 3 yr after termite exclusion (Spears et al., 1975). The explanation provided by the authors was that in the Chihuahuan Desert study, macropore formation was the dominant process affecting infiltration. In West Texas, litter cover was more important. Termites are responsible for removing large amounts of litter and reducing soil organic matter (SOM) in both systems (Nash and Whitford, 1995). In the Texas study, litter increased from 35 g m^{-2} in the control plots to 63 g m^{-2} in the termite removal plots. Soil organic C in the surface 1 cm correspondingly increased from 1.2 to 1.8%. This illustrates that the use of an individual organism as an indicator of a property, such as infiltration capacity, can be complicated if the organism affects more than one process related to the property. This also illustrates how relative differences in the importance of different processes can confound indicator interpretation across regions.

Termites are also difficult to use as indicators because their populations vary with soil type and landscape position. In the Chihuahuan Desert, the relationship between recent termite activity and infiltration capacity is potentially high in the uplands but nonexistent in the playas, from which termites are largely absent (Nash and Whitford, 1995). The contribution of termites to infiltration and other ecosystem processes also declines with increases in latitude and elevation due to the changes in species composition (Weesner, 1965; Herrick, 1999).

USE SPATIAL VARIABILITY IN DEVELOPING AND INTERPRETING INDICATORS

Patterns and scales of spatial variability present tremendous opportunities to develop robust indicators because the spatial structure of ecosystems often reflects how they are functioning. The spatial structure also often reflects the status of key processes. Landscape ecology is largely based on the inference of process from pattern (Turner et al., 1993). Thus, detecting change in ecological processes and responding with management actions will in most cases involve detecting subtle changes in pattern. At the most basic level, a change in spatial variability at any scale indicates that the distribution of resources has changed. Correspondingly, it reflects a change in the processes that both control, and are affected by, the spatial distribution of resources at that scale. Changes in spatial variability may differentially affect ecosystem processes across a landscape. Figure 1 shows how an increase in spatial variability of infiltration capacity has cascading effects throughout the system, leading to interacting feedback loops in plant production and community composition and belowground processes. Many of these feedbacks have been described (e.g., Davenport et al., 1998; Reid et al., 1999) although there are few long-term studies that have effectively documented their development over time.

The importance of water redistribution at the plant-interspace and patch-interspace scale, and its relationship to differences in soil properties, has been documented for a number of arid and semiarid ecosystems throughout the world. Bromley et al. (1997) calculated that the amount of water received by grassy open areas is up to 3.2 times actual rainfall due to runoff from surrounding nonvegetated areas. This number is based on the higher hydraulic conductivity of the grassy open areas (0.3–0.6 vs. 1.5×10^{-6} m s^{-1}) and on the relative area of each area. It was validated using soil moisture measurements. The effects on plant production included higher rates of survival and longer persistence of green leaves during the dry season. Similar data have been reported for piñon (*Pinus edulis*)–juniper woodlands in the USA (Reid et al., 1999) and for Australian mulga woodlands (Tongway and Ludwig, 1997; Table 3).

Careful indicator selection is necessary to effectively interpret spatial variability. Figure 1 illustrates that changes in the spatial variability of a number of other soil and plant community properties could also be used. Indicator selection should be based on a comprehensive analysis, potentially aided by modeling, of two interrelated criteria: a high rate of change in spatial variability early in the degradation or recovery process and an ability to measure the property at a level of precision that is sufficient to detect that change. Spatial patterns of some properties change relatively quickly while others, such as the distribution of long-lived plant species that are primarily establishment limited, can lag years or decades. The placement of infiltration capacity at the top of Fig. 1 was arbitrarily based on the functional importance of this process in many arid and semiarid rangeland ecosystems. Precise measurements of infiltration capacity are expensive. We have included single-ring infiltration as a supplementary method for systems, such as irrigated pastures, wet meadows, and soils dominated by lichen crusts, in which large changes in infiltration capacity can occur relatively quickly. Spatial variability in soil aggregation or in a related SOM fraction

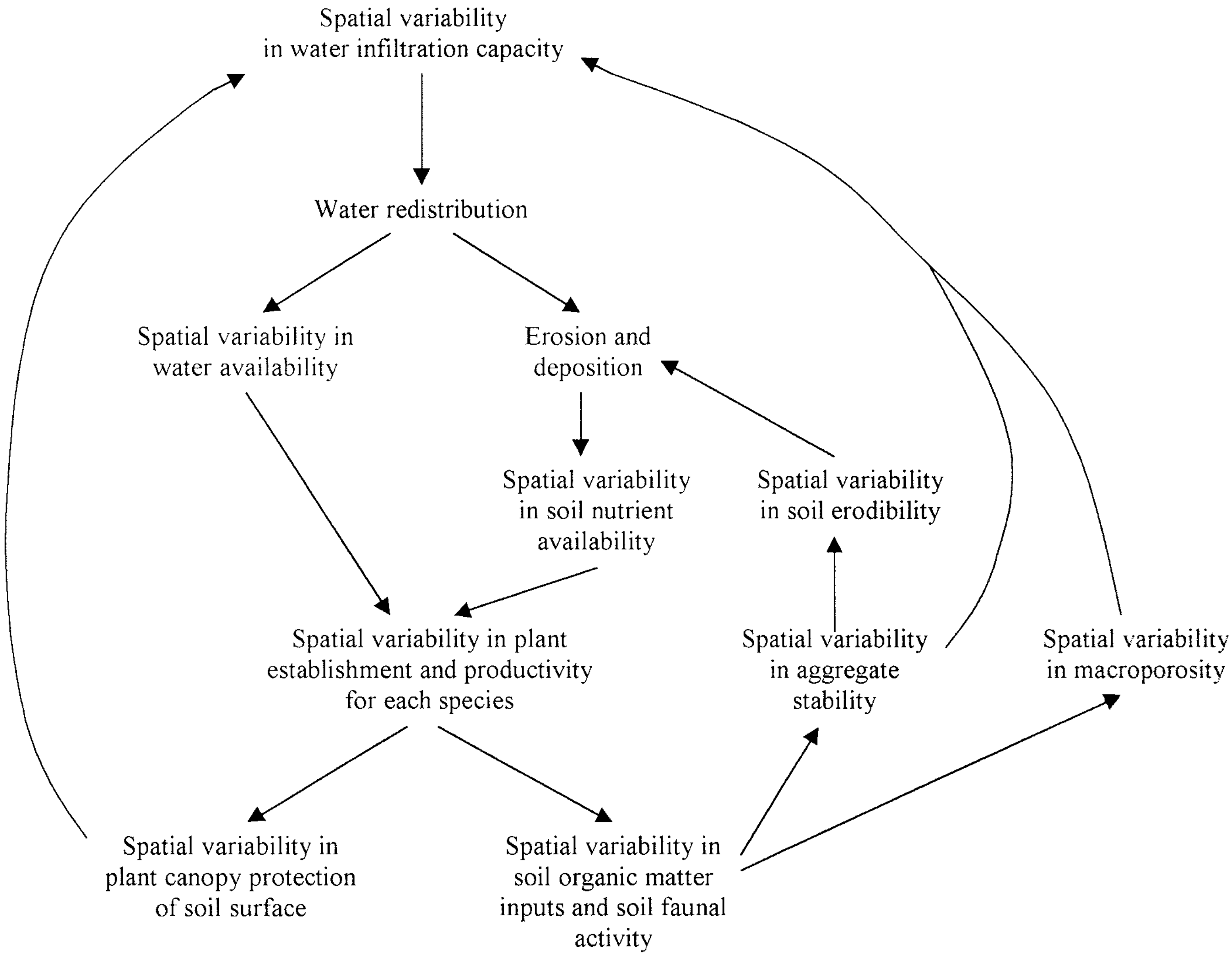

Fig. 1. Example of how spatial variability in a soil property can serve as an early warning indicator of change in ecosystem function due to cascading effects and positive feedback loops.

may be a cost-effective and sensitive surrogate indicator in many cases (Fig. 1; Herrick and Wander, 1998). Studies are currently underway to evaluate both spatial variability of aggregate stability and various SOM fractions.

Changes in spatial variability that reflect changes in process and function can occur at any one of a number of scales. The example illustrated by Fig. 1 is commonly used because its relationship to function has been relatively well described for a number of systems (Schlesinger et al., 1990; Pierson, Jr., et al., 1994; Reid et al., 1999). Spatial variability at finer scales, such as surface roughness, can be a useful indicator of susceptibility to erosion (Saleh and Fryrear, 1999). At coarser scales, changes in the relative size of vegetation patches, erosional areas, and depositional areas and shifts in ecotonal boundaries can be sensitive indicators that change is occurring throughout the landscape (Coffin and Lauenroth, 1990; Ludwig and Tongway, 1995).

Because they are related to processes, spatial indicators are often particularly useful for making management decisions. Management is often designed to effect a short-term change in a property of a system, such as bulk density or infiltration capacity, but it is the long-term change in a process, such as redistribution of water, that has the greatest effect on the system.

INTERPRET INDICATORS BASED ON AN UNDERSTANDING OF ECOLOGICAL PROCESSES

Significant progress has been made in the identification of suitable indicators for both cropland and rangeland ecosystems (Doran and Parkin, 1994; Natl. Res. Counc., 1994; Doran and Parkin, 1996; Brown et al., 1998). The integration and interpretation of these indicators has been more difficult. A number of approaches have been suggested and successfully applied to some

Table 3. Water redistribution associated with soil–vegetation patches along a topographic gradient in an Australian mulga woodland (modified from Tongway and Ludwig, 1997).

Measurement	Bare zone† (upper)	Grass zone (middle)	Tree zone (lower)
Soil water infiltration, mm	15.7	33.7	51.6
Percentage of 37.5 mm of rainfall	42	90	138

† No vascular vegetation present.

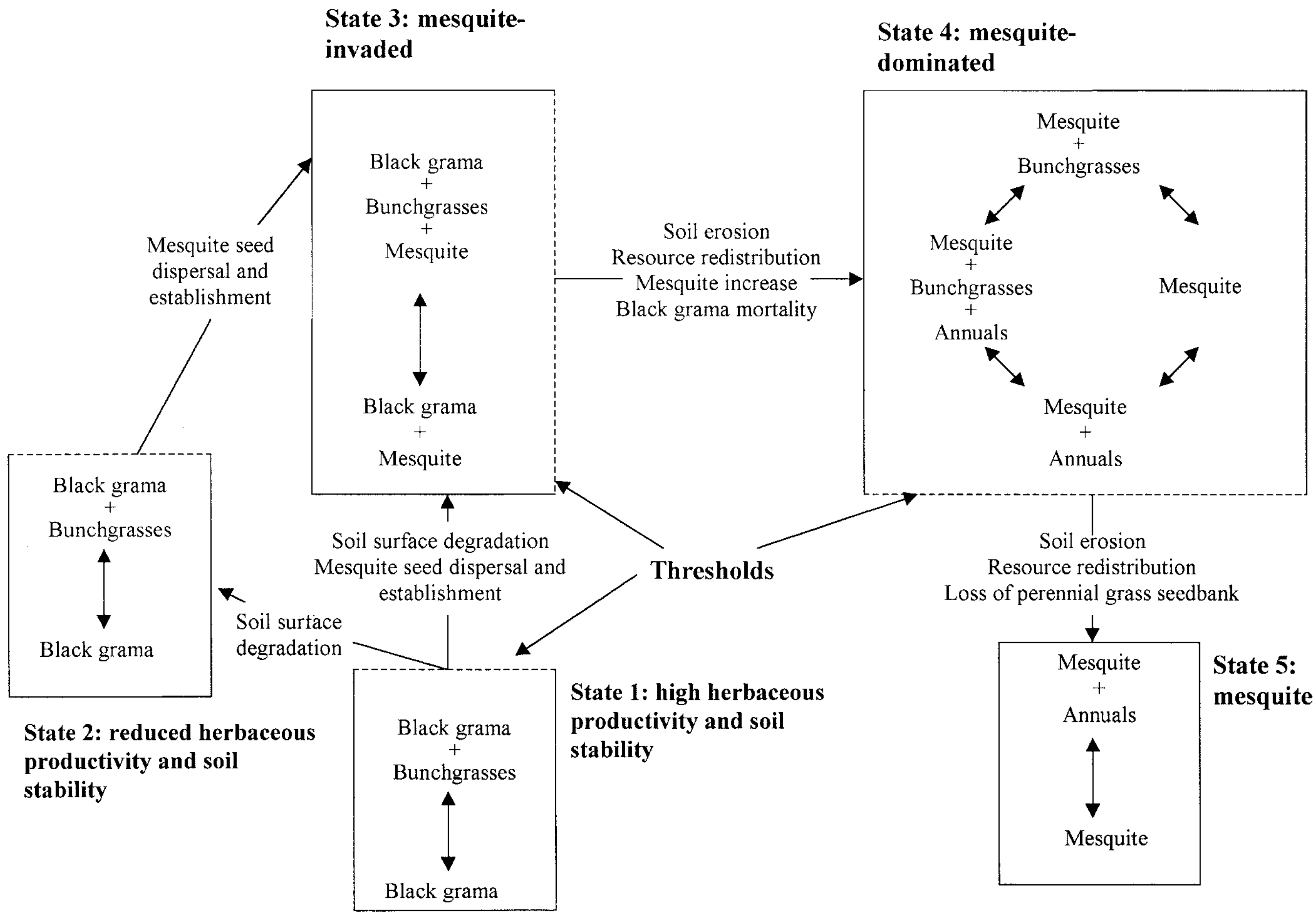

Fig. 2. Example of a state and transition diagram for a Simona soil in the northern Chihuahuan Desert. The structure of the model is based on Stringham et al. (2001). Arrows within boxes (states) represent easily reversible plant community composition and associated soil changes. Unidirectional arrows between states reflect that the transitions are nonreversible without external inputs. Text between states describes some of the processes associated with each transition. The diagram has been simplified and does not include all possible plant communities in States 3 and 4.

systems. Linear combinations of indicators have been used to develop indices (e.g., Doran and Parkin, 1994). These approaches are extremely valuable for documenting change in systems that are gradually evolving. However, ecological theory suggests that a more dynamic model may be more appropriate in systems that are structured by relatively infrequent catastrophic disturbances or in which cumulative effects are not expressed until a threshold is reached (Holling, 1973).

The threshold concept is widely applied in agronomy and land management, particularly in the areas of integrated pest management (Kogan, 1998; Hoffman et al., 1999), landscape stability, and soil erosion (Davenport et al., 1998; Weltz et al., 1998). The concept has been applied in rangeland ecosystems in the form of state and transition models (Westoby et al., 1989; Friedel, 1991). These models are based on the assumption that relationships between different properties and processes become increasingly nonlinear as a threshold is approached. The application of these models is a relatively recent development and represents a significant departure from the linear plant succession–regression based paradigm that guided range management through most of this century (Soc. for Range Manage., 1995). The fact that the process–property relationships become increasingly nonlinear near thresholds means that linear combinations of indicators that effectively reflect changes in ecosystem function may become completely ineffective at these critical periods.

A unique state and transition model can be described for each soil or suite of similar soils. State and transition models consist of states, transitions, and thresholds. A state can be defined by a single plant community or multiple plant communities together with characteristic dynamic soil properties, such as organic matter content and erodibility. Although the dominant species in each of the plant communities are used to conveniently describe the states, the state is defined by soil and vegetation properties and processes and by soil × vegetation interactions mediated by the animal community. Soil and vegetation changes within a state are easily reversible, and the states themselves are relatively stable. Transitions between states occur after crossing a soil- or vegetation-defined threshold that is not easily reversed without significant inputs of resources (Friedel, 1991; Committee on Rangeland Classification, 1994).

The state and transition diagram illustrated in Fig. 2 is based on current understanding of the ecological

Table 4. Measured values for quantitative indicators for sites representing States 2 and 3 in Fig. 2.

Measurement	State 3	State 4
LINE-POINT INTERCEPT (300 points, each 30 cm apart)	%	
Total cover (live canopy, gravel, dead canopy, soil lichen)	81.6	49.3
Total canopy cover	75.6	32
Mesquite cover	0.0	17.0
Total basal cover	9.6	1.0
CONTINUOUS LINE INTERCEPT (two 50-m lines)		
Proportion of line covered by noncanopy intercepts longer than 50 cm in length	21.1	66.5
Proportion of line covered by noncanopy intercepts longer than 100 cm in length	7.2	59.7
SOIL STABILITY IN WATER (n = 18)		
Proportion of values = 6 (highly stable)	37.5	12.5
	Mean ± SE	
Mean value (n = 18)	4.1 (0.4)	1.9 (0.5)
Mean value for bare (no canopy)	2.6 (0.9)	1.1 (0.4)
Mean value for under grass	4.7 (0.4)	n/a
Mean value for under shrub	n/a	5.3 (0.7)

dynamics on a northern Chihuahuan Desert site with soils that are shallow (<60 cm deep), sandy, and have a petrocalcic horizon, as represented by the Simona soil (loamy, mixed, thermic, shallow Typic Paleorthids). States 1 and 2 are dominated by black grama [*B. eriopoda* Torr.], a C4 stoloniferous grass, and have few large canopy interspaces, except those generated by small-scale disturbances such as banner-tailed kangaroo rat [*Dipodomys spectabilis*] mounds. Use of these mounds by the rats prevents the establishment of perennial plants. Except for the rodent mounds, the soil is relatively stable in State 1, and there is a clearly defined A horizon evenly distributed throughout the site, reflecting relatively uniform resource distribution. Microbiotic crusts dominated by cyanobacteria are common. A black grama–bunchgrass community can also exist in this state.

The transition to State 2 occurs when the soil and plant community become degraded and plant production declines. It is characterized by loss or degradation of the soil surface and an increase in the size of canopy interspaces. State 2 is more likely than State 1 to be dominated by bunchgrasses.

The transition from State 1 or 2 to State 3 is defined by mesquite seed dispersal and establishment. Mesquite is a leguminous shrub. The relative importance of soil surface degradation and mesquite invasion in defining the transition to State 3 has never been clearly defined because the two processes have historically occurred simultaneously in many areas: A combination of overgrazing and drought have exposed the soil surface to erosion, and livestock have dispersed large quantities of mesquite seed into grasslands. Evidence from a more mesic system in Texas suggests that seed dispersal may be the limiting factor (Brown and Archer, 1999). Reversal of this transition requires elimination of the mesquite, its seedbank, and dispersal agents and possibly restabilization of the soil surface.

The third state includes at least two communities. In addition to the presence of mesquite, the state is characterized by larger intercanopy gaps. The core measurements listed in Table 1 were completed at sites that are representative of States 3 and 4. The quantitative indicators are summarized in Table 4. Both sites are located on the Simona soil in southern New Mexico on the USDA-ARS Jornada Experimental Range where precipitation from 1961 to 1994 averaged 261 mm. As the transition from State 3 to State 4 begins, soil aggregate stability in the plant interspaces continues to decline, SOM distribution becomes increasingly concentrated under plants, and canopy gap diameters increase (Tiedemann and Klemmedson, 1986; Wright and Honea, 1986; Schlesinger et al., 1990) (Tables 1 and 4).

The transition from State 3 to State 4 is defined by the dominance of mesquite over the grasses and the associated increase in gap diameter and soil susceptibility to both wind and water erosion within the gaps. Once this threshold is crossed, re-establishment of black grama is nearly impossible. The shorter-lived perennial bunchgrasses may persist and expand during wet years, declining or disappearing again during droughts. The transition to the fifth state occurs when bunchgrass production is insufficient to maintain a viable seedbank for re-establishment during wet periods, or when soil surface conditions become so degraded that establishment is impossible. In this state, the system is dominated by mesquite coppice dunes, with annual forbs sometimes occurring in the interspaces during wet years.

This diagram (Fig. 2) is relatively simple: Community pathways branch within states, but there are at most two potential transitions for each state. The structure is based on that by Stringham et al. (2001). The number of potential states could easily increase in response to climate change or species invasions. Earlier state and transition models often recognized individual communities as separate states, whether or not they were separated by thresholds, making it difficult to consistently define states.

The key points illustrated by this example are that transitions are defined by nonlinear changes in the function of the system and that mean values of a suite of indicators may provide relatively little information about the status of the system when it is near threshold. The transition from State 1 to State 3 can be precipitated by the mere establishment of mesquite in the system. An average of all other indicators may show little or no change. Similarly, the fact that the site described under State 3 in Table 4 is near, if not at, threshold would not be predicted from an average value of the quantitative indicators, nor from a casual visual examination of the site or a more formal qualitative evaluation. With the exception of small, scattered mesquite shrubs, the site appears to be similar to a grassland site in State 1, and only 2 of 17 qualitative vegetation and soil surface indicators used to evaluate the site (Pellant et al., 2000) were rated by an interdisciplinary team of experts as being more than moderately different from what would be expected for a site in State 1. This qualitative evaluation system uses five categories to describe the relative difference between the site of interest and a real or (in this case) reconstructed reference site: none–slight, slight–moderate, moderate, moderate–extreme, and extreme. The two qualitative indicators that were rated as being more than moderately different from what would be expected for the site in State 1 were the degree of pedestalling and soil surface resistance to erosion. The former indicated a large amount of historic soil redistribution; the latter indicated that the soil is currently highly erodible. A third indicator, plant functional and structural groups, was rated by the interdisciplinary team as being only slightly to moderately different from what would be expected for State 1 because species representing most of the original functional and structural groups were present on the site and there was only one new group: that represented by mesquite. Although mesquite was not recorded on the two 50-m line transects (Table 4), there were a number of plants on the site that were too large to be effectively killed by fire or rodent activity. This, together with a highly eroded and erodible surface and the incipient development of large gaps in the canopy, indicate that this site is at or near threshold, in spite of the fact that the majority of both qualitative and quantitative indicators suggest that it is in relatively good condition compared with a site in State 4 (Table 4).

The precise definition, quantification, and recognition of site-specific thresholds are some of the most important and difficult tasks faced by the range management community. Economic criteria similar to those applied in integrated weed-management systems can be used to more clearly, albeit artificially, define thresholds: the point at which external inputs are no longer recovered by increased production of the former state. This is qualitatively different than the earlier threshold most commonly defined in integrated weed-management programs: the point at which pest populations reach a level beyond which the cost of additional reductions in crop production in the absence of control exceeds the cost of the of the control measure. This type of threshold can be identically applied to weed management programs in rangelands and, if applied, may prevent the system from reaching a second threshold beyond which control is no longer economically viable and the system moves into a new state. In general, however, ecologically defined thresholds are preferable to economically defined thresholds for describing ecosystems because the definitions are less likely to change over time with economic conditions.

There are a number of other studies in the literature that also demonstrate the importance of thresholds, and the most effective early warning indicators are not necessarily the most obvious indicators. For example, herbaceous species composition and productivity are commonly considered to be good indicators of condition. However, Northrup and Brown (1999) illustrated that while a tussock grass–dominated community appeared to be stable based on these indicators, loss of biological integrity in response to overgrazing was occurring via the fine-scale redistribution of soil resources. In another example, Brown and Ash (1996) showed that monitoring species composition and plant production data to detect shrub invasion can result in delaying important management decisions until well after woody invaders have become established. Instead, indicators that indicate susceptibility to invasion need to be identified. Finally, Davenport et al. (1998) clearly demonstrated the role of erosion thresholds in defining different states for grasslands invaded by piñon and juniper in the Southwest and documented that the probability of crossing different thresholds strongly depends on site characteristics.

CONCLUSIONS

The four guidelines described above can be used to develop effective rangeland monitoring systems that are relevant to management. While the paradigm described here is not necessarily one that can be directly applied to croplands, many of the elements are already being applied through soil quality. Applying the concept of thresholds to soils under different management systems, including cropping, and developing state and transition models for these systems may more accurately reflect critical dynamics. State and transition models could be used to help focus conservation resources on those areas at highest risk of degradation or with the greatest potential for recovery.

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