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Desert rangelands, degradation and nutrients

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ABSTRACT It is well recognized that rangeland environments, especially degraded environments, are relatively infertile. Though arid lands are moisture limited, it is often stated that they are nutrient regulated. A discussion of rangeland soil nutrients relative to landscape degradation should include: (1) dynamics of nutrient distributions; (2) soil properties affecting nutrient fluxes; (3) indicators of soil quality; and (4) nutrient based strategies for remediation. Nutrient distribution is strongly influenced by vegetation structure, and spatial nutrient patterns are closely linked to aspects of primary productivity and species composition. Effects of vegetation on nutrient spatial patterns can persist for decades, even if vegetation is altered by natural or anthropogenic disturbances. Though abiotic processes strongly shape physical features of arid landscapes, biotic activities directly affect soil nutrients. Unfortunately, assessment and monitoring technologies based on soil properties which quantify integrity of nutrient related processes have not been fully developed. Approaches for remediating degraded conditions need to exploit nutrient spatial and temporal heterogeneities. Remediation should be targeted to the most fertile sites, even if these are only relatively small parts of the overall landscape. Understanding nutrient dynamics is a key to triggering autogenic remediation of degraded rangelands.

Key words: desertification, desert grassland, nitrogen, nutrient dynamics, phosphorus.

1. INTRODUCTION

Rangeland degradation frequently occurs in desert environments throughout the world (Ludwig and Tongway, 1995; Arnalds, 1987). Although productivity in desert rangelands is one to three orders of magnitude lower than in forest ecosystems (Ludwig, 1987), desert rangelands cannot be viewed as either simple or unproductive systems. Productivity is strongly coupled to precipitation, but nutrients and their availability strongly regulate both primary and secondary production (Noy Meir, 1979/80).

Degradation of desert grasslands is often characterized by replacement of perennial herbaceous species by long-lived woody shrubs and by an associated reduction in the capacity of the ecosystem to perform some functions. These reductions, such as the capacity to capture and retain water from high-

intensity precipitation events (Ludwig and Tongway, this volume; Thurow, this volume), may be irreversible on human timescales.

Numerous causes have been proposed for this grassland-to-shrubland transition including overgrazing (Conley et al., 1992), exclusion of fire (Brown and Minnich, 1986), dispersal of seeds of woody species by herbivores (Buffington and Herbel, 1965) and a combination of natural and anthropogenic stressors (Hastings and Turner, 1965; Archer et al., 1995). It is not likely that the historical effects of any one factor (stressor) can be clearly and singularly associated with desert grassland degradation. We can not necessarily differentiate between cases in which certain stressor(s) are directly responsible for changes and cases in which they simply facilitate the transition from a community that evolved under a previous climatic regime to one that is better adapted to current conditions (Tranquillini, 1979; Neilson, 1986). However, the effects of degradation are widely recognized. In addition, it is now understood that these changes can be long-lived, non-linear and resistant to remediation (Herrick et al., 1997; Archer and Stokes, this volume; Tongway and Hindley, this volume).

The dynamics of desert rangeland degradation have been conceptualized for the Chihuahuan Desert of North America (Fig. 1). Allogenic forces contributed to reductions in herbaceous cover and to increased dispersal of seeds of competitive woody species, which rapidly established under favorable conditions including reduced competition from the perennial grasses (Grover and Musick, 1990). Autogenic forces then reinforced spatial redistribution of limiting resources into larger, shrub-associated patches (Schlesinger et al., 1996). These dynamics parallel degradation processes described for other grazed terrestrial systems (van de Koppel et al., 1997; Ludwig and Tongway, this volume).

2. NUTRIENT DYNAMICS

The conceptual model of degradation in Fig. 1 focuses on nutrient responses to vegetation dynamics. It is well recognized that even desert grasslands in excellent condition are nutrient-poor environments and that several important soil nutrients including N, P, K and Zn are commonly deficient (Dregne, 1976). These deficiencies are compounded by spatial (both vertical and horizontal) and temporal variability, which further limit nutrient availability to plants during periods when water is relatively non-limiting (Virginia and Jarrell, 1983). The relative importance of these deficiencies depends on both inter- and intra-specific differences, and on the degree and effectiveness of mycorrhizal infection.

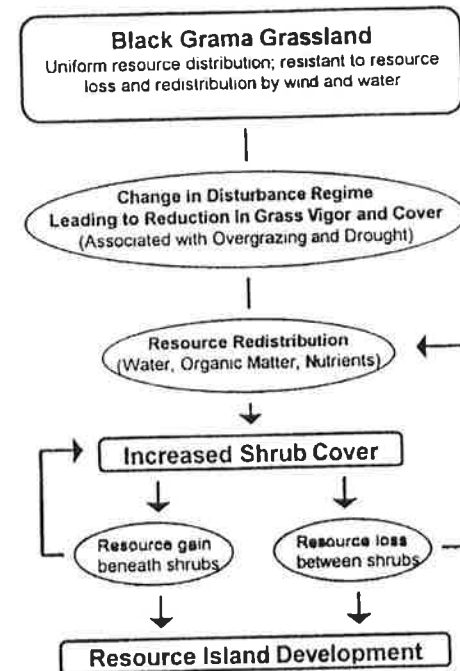


Figure 1. A central hypothesis of the research at the Jornada Long-term Ecological Research site in southern New Mexico is that changes in vegetation are accompanied by a redistribution of water and soil nutrient resources on the landscape, and this redistribution acts as a positive feedback mechanism to further promote degradation processes.

Nutrient-based processes in deserts are poorly understood (Zaady et al., 1996) due to both the diversity and complexity of these ecosystems, and the limited resources available to study them. Existing information, however, suggests that the distribution and dynamics of organic matter may be used as a general framework. Organic matter serves as an important source and sink for nitrogen and other nutrients in nearly all soils (Broadbent, 1986; Mengel and Kirkby, 1987), and is particularly important in degraded and sandy soils in which exchange surfaces and inorganic sources of nutrients are limited (Herrick and Wander, 1998). In non- or little-fertilized systems, plant N, P, S and microelement nutrition can depend heavily on mineralization of organic matter and residues of plants and animals (Chen and Stevenson, 1986; Broadbent, 1986). The most dynamic or rapidly cycled soil organic matter fraction is the biologically active fraction, which is tied to mineralization and therefore soil nutrient supply (Greenland and Ford, 1964).

A conceptual model (Fig. 2) describing soil organic carbon (Sollins et al., 1996) may be used to extend the grassland degradation model (Fig. 1). Degradation is associated with a change in spatial distribution of soil OM inputs.

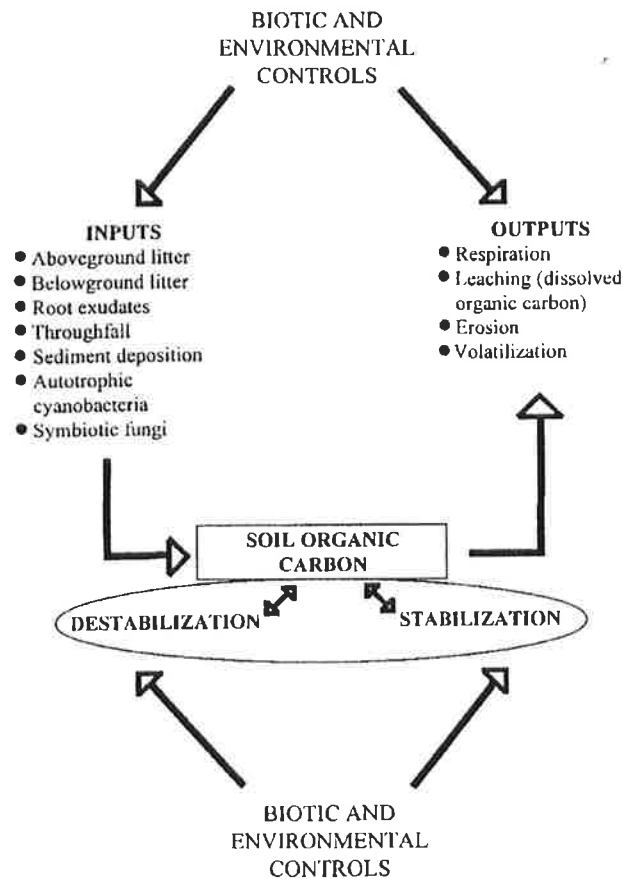


Figure 2. Organic C in soils results from long-term processes that control various C inputs and outputs. Destabilization processes increase the outputs. Adapted from Sollins et al., 1996.

In the Chihuahuan Desert, Connin et al. (1997) demonstrated a relatively rapid change in the form and vertical distribution of organic matter inputs to the soil profile but little change in total organic carbon following conversion of a grassland to a mesquite (*Prosopis glandulosa*) shrubland. For example, root diameters of >3 mm were absent from the grassland, and few roots were

encountered below a depth of 40 cm, whereas the degraded shrub-dominated systems had significant quantities of large diameter roots throughout the profile. The quality of the organic matter inputs also changed. A C:N ratio of 38 was reported for grassland litter, while in the shrubland the ratio dropped to 16. Litter is recognized as an important regulator of the rate and fate of mineralized N in deserts (Zaady et al., 1996). These changes could dramatically affect C storage, N mineralization and OM turnover rates (Connin et al., 1997). These effects on C:N ratios and litter inputs are similar to those reported for grazing responses in more mesic grassland environments (Frank et al., 1995; Shariff et al., 1994).

The distribution of N, P and K has been shown to be strongly associated with the presence of shrubs in deserts (Schlesinger et al., 1996; Schlesinger and Pilmanis, 1998). Abrams and Jarrell (unpubl. data) found that the intensity of spatial variability of soil nutrients in the top 30 cm (Table 1), and the degree of association with the organic matter-rich soils under shrubs, increased with time along a shrub invasion chronosequence. Other studies have demonstrated similar spatial patterns in the activity of soil fauna which contribute to nutrient cycling (e.g., Santos et al., 1978; Kieft et al., 1998).

Table 1. Mean (x) soil nutrient concentrations in top 30 cm and coefficients of variation (CV) for three nutrients across a degradation chronosequence. The chronosequence occurred over a linear distance of 400 m where a gradual gradation from desert grassland to mesquite duneland was documented in 1937. The original grassland is intact at the start of the gradation, but mesquite dunelands have advanced in the ensuing 60 years, and this encroachment has been documented (Abrams and Jarrell, unpublished data).

Degradation chronosequence (yrs)	P (mg kg ⁻¹)		K (cmol kg ⁻¹)		Zn (mg kg ⁻¹)	
	x	CV	x	CV	x	CV
0 (Undegraded desert grassland)	1.58b*	39.91	0.44c	24.60	0.34a	17.74
10	4.07a	50.19	0.47b	18.97	0.30b	23.34
35	1.02c	39.92	0.41d	18.04	0.22c	36.61
45	1.65b	69.24	0.49b	21.70	0.23c	37.43
60 (Degraded mesquite duneland)	1.57b	54.19	0.58a	27.89	0.18d	40.50

*Means in the same column followed by the same letter do not differ ($P < 0.05$).

Based on measurements of erosion rates in these grassland-mesquite transitions (Table 2; Gibbens et al., 1983), this increase in variability should be associated with a net reduction in both organic matter and nutrients. A textural analysis of soils (Table 2) further indicated that silt-sized particles were preferentially removed from mesquite-invaded parts of the "grid stakes" site (Hennessey et al., 1986). However, the measurements of Abrams and Jarrell, made at the "transect stakes" site (Table 2) provided little indication of nutrient loss from the system as a whole with degradation. The one exception was a significant decrease in Zn availability with degradation.

Table 2. Deposition and deflation of soil in 1935 and 1980 at 105 grid stakes on the 259-ha natural revegetation enclosure where soil levels were marked in 1933, and 1980 soil levels at 113 transect stakes on the enclosure on which soil levels were marked in 1935. From Gibbens et al., 1983.

Database	Year of measurement	Soil movement category	Number of points	Maximum (cm)	Minimum (cm)	Mean (cm)	Net loss (-) or gain (+) (cm)
Grid stakes	1935	No change	9				
		Deposition	36	6.0	0	1.1	
	1980	Deposition	33	78.3	1.8	23.8	
		Deflation	72	61.9*	0.9	17.4	-4.6*
Transect stakes	1980	Deposition	43	78.6	0.6	2.5	
		Deflation	70	45.1*	0.9	2.1	-3.5*

*Represent minimum values because one stake was completely excavated by wind erosion.

Net primary productivity data from the Chihuahuan Desert provides some further indirect evidence for an increasing nutrient spatial heterogeneity in response to degradation. Huenneke, Anderson, Muldavin and Schlesinger (unpublished data) reported on aboveground biomass production for grassland and shrubland systems measured during three seasons for each of four years. Overall mean values of net primary production did not vary among these systems even though shrubland systems represent severely degraded conditions in this environment (Schlesinger et al., 1990). Systems did significantly vary in seasons of peak production and in the spatial variability of aboveground production. Shrublands displayed significantly greater spatial heterogeneity of production.

3. NUTRIENT DYNAMICS AND REMEDIATION

The overall importance of spatial patterning in deserts is well recognized and increasingly well described (Stafford-Smith and Morton, 1990; Tongway and Hindley, this volume). Production is highly patterned (and boundaries can be very abrupt), water is intermittently available, and nutrients are concentrated within fertile pockets in many systems. Remediation of degraded landscapes needs to recognize and exploit the opportunities created by spatial patterns (Whisenant and Tongway, 1996) and the episodic conditions that created these patterns (Winkel et al., 1991). Remediation technologies developed for average and homogenous environments will be ineffective, unsuccessful, costly and unadopted. Whisenant and Tongway (1996) argued that effective mitigation of degradation requires an intense attention

to ecological processes. Call and Roundy (1991) have suggested that "safe" sites where germination and establishment can occur must be identified and exploited for arid and semiarid land reclamation. Herrick et al. (1997) have proposed a conceptual strategy for remediation that extends this concept of "safe" sites to recognize relatively fertile sites that occur within degraded arid landscapes. A basic premise of this model is that initial sites for remediation not only have greater ecological potential than surrounding sites, but will inherently facilitate migration from the site (Fig. 3). Conceptual models which may be used to help identify and select these fertile "trigger" sites include the Desertification Response Unit approach (Imeson et al., 1996; Imeson and Cammeraat, this volume) and the Australian pulse-reserve (Ludwig and Tongway, this volume). Both have nutrient dynamics embedded within them.

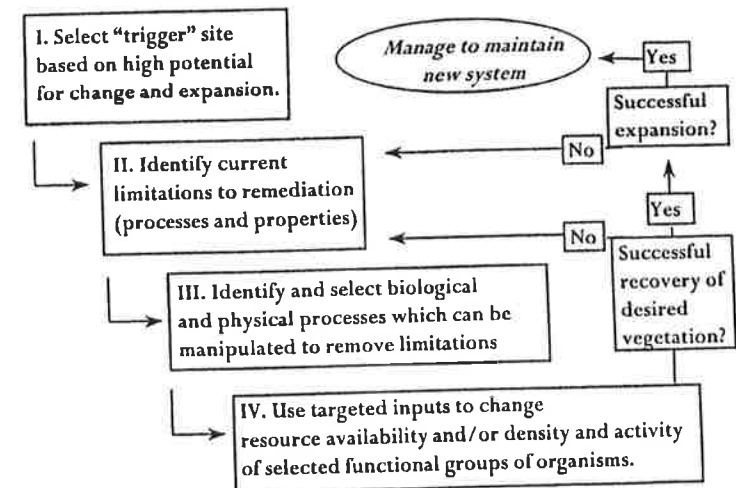


Figure 3. Conceptual model for development and application of an ecologically-based approach to rangeland remediation. Adapted from Herrick et al., 1997.

4. NUTRIENT PATTERNS: INDICATORS AND/OR DRIVERS OF DEGRADATION

The complexity of ecosystem processes, their interactions and their dynamics in response to degradation may confound attempts to identify simple

nutrient-vegetation associations across vegetation states. However, the connection between changes in the spatial distribution of nutrients and the degradation of desert grasslands is well documented, and it is clear that most arid and semi-arid ecosystems are nutrient-limited when water is abundant and temperatures are appropriate for plant growth. The spatial distribution of nutrient resources may, in fact, serve as a valuable indicator of soil quality (Schlesinger et al., 1996; Herrick and Wander, 1997; Tongway and Hindley, this volume). Some parameters or indices may even serve as early warning indicators of degradation risk. For example, an increase in the coefficient of variation or ratio of soil organic matter concentrations in bare spaces vs. below plant canopies may be an indicator of reduced resilience of a site to recover from degradation.

While the correlation between changes in nutrient patterns and degradation makes this a potentially valuable indicator, the causal relationship between nutrient redistribution and the maintenance or further degradation of altered systems remains largely untested. Spatial and temporal patterns of nutrient availability can control plant community composition by limiting plant establishment and/or growth and reproduction. So can water (Thurow, this volume). The distribution of both is highly altered by shrub invasion. An understanding of the relative importance of water and nutrients at establishment, growth and reproduction is essential to future remediation attempts, in which inputs must be carefully targeted for maximum benefit.

5. FUTURE RESEARCH DIRECTIONS

One of the key barriers to developing effective, ecologically-based remediation technologies is our relatively limited understanding of how nutrient cycling processes are linked at different scales in the context of constantly changing moisture and disturbance regimes. Changes in nutrient availability at one scale frequently depend on ecosystem processes which occur at completely different scales. The spatially and temporally dynamic nature of nutrient cycling in deserts should provide tremendous opportunities to effect change at larger scales by understanding and targeting smaller scale processes. These opportunities will remain unexploited, however, unless significant advances are made in our understanding of processes occurring at the level of the rhizosphere and even the "hyphal sphere", how these processes are affected by moisture and disturbance, and how they affect, and are affected by water and nutrient resource transfers at scales from plant to the landscape. Historically, most studies have been completed at the plot level, or in the laboratory using disturbed, homogenized samples under relatively controlled moisture regimes. Future studies must build on current knowledge

of spatial patterns in arid landscapes by quantifying inputs and losses at the patch scale which functions in both patch- and landscape-level processes (see also Ludwig and Tongway, this volume; Tongway and Hindley, this volume).

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