# 13 Rangeland Soil Erosion and Soil Quality: Role of Soil Resistance, Resilience, and Disturbance Regime

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## INTRODUCTION

The relationships between rangeland soil quality, soil resilience, and soil erosion depend on several interacting factors: (1) landscape and climate characteristics, (2) current disturbance regime, and (3) recent and evolutionary disturbance history. These factors tend to be more variable across rangelands than across agricultural lands. There are at least four specific relationships between soil quality and soil erosion which involve soil resistance or soil resilience. The first is the *historical resistance* of the soil to past disturbances, which can serve as an indicator of soil quality. Second, the *current resistance* of the soil to disturbance is related to soil erosion potential. The third relationship is the *current resilience* of the system following soil erosion. Finally, soil erosion is a *driver* in the system which determines soil quality. This final relationship illustrates the need to view the system dynamically: soil erosion both reflects and affects soil quality. These dynamic relationships depend, in turn, on the characteristics of historic and current disturbance regimes. Both ecosystems and species tend to evolve in response to dominant disturbance regimes, such as fire, drought, and grazing. The resistance and/or resilience of the system will tend to be higher for disturbance regimes which share key characteristics with historic and evolutionary patterns.

Over 30% of the U.S. land surface and 34% of the global land surface, exclusive of Antarctica, is classified as rangeland (World Resources Institute, 1992; National

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Research Council, 1994). Rangelands are arguably the most diverse of any class of productive land and are associated with infertile lowland soils throughout the humid tropics and with arid, semiarid, and steepland soils on nearly every continent. A common characteristic of most rangelands, however, is that they have some edaphic and/or climatic limitations which have prevented them from being used for intensive crop production. This functional definition of rangeland is implicit in the USDA land capability classification system (Dent and Young, 1981).

Soil quality can be generally defined as the long-term capacity of a soil to perform functions which sustain biological productivity and maintain environmental quality. This definition, similar to many of those listed in recent reviews by Doran and Parkin (1994) and the National Research Council (1993), explicitly does not favor one land use over another. Rangelands are valued for a wide variety of uses including food and fiber production, watershed protection, wildlife conservation, and recreation (National Research Council, 1994). A high-quality rangeland soil is one which will maintain its functional integrity and therefore sustain its many possible uses into the future. Consequently, the conservation of soil and water resources, or minimization of runoff and soil erosion, has emerged as a potentially key indicator of rangeland health, as well as soil quality. The relationships between soil erosion and soil quality, however, are not fully understood.

The objectives of this chapter are to define specific relationships between soil erosion and soil quality, to identify and describe several factors which determine the nature of these relationships for specific ecosystems, and to illustrate these relationships with examples from south-central New Mexico, southeastern Arizona, and northeastern Colorado. A brief discussion of the contribution of soil quality to rangeland health is also included. This chapter is designed to generate discussion relevant to assessing rangeland soil quality and its relationship to soil erosion. As such, it is not intended to serve as a review of the literature on rangeland soil erosion, for which the reader is referred to two edited volumes on the subject (Blackburn et al., 1994; Spaeth et al., 1996a).

## RELATIONSHIPS BETWEEN SOIL EROSION AND SOIL QUALITY

### Definitions

Disturbance, resistance, and resilience are interrelated terms which are critical to understanding relationships between soil erosion and soil quality. Definitions vary widely and frequently depend on both the author and the context in which the word is used. A disturbance is generally defined as any event which causes a significant change from the normal pattern in an ecosystem (Forman and Godron, 1986), where pattern includes both spatial and temporal distributions of plants, microtopographic features, and soil and plant community properties, processes, and functions. Changes caused by disturbances may be positive or negative. Whether or not an event is classified as a disturbance depends in part on the spatial and temporal scales of interest. The creation of a macropore by an earthworm at the base of a grass clump may be viewed as a disturbance at the individual plant scale during the course of a season. However, it would have little impact on hydrology at the watershed scale or at the plant scale over a period of several decades.

Whether or not an event is classified as a disturbance also depends on how resistant the system is to the particular event. Resistance is defined as the capacity of a system to continue to function without change through a disturbance (Pimm, 1984). The resistance of a system depends both on effects on individual system elements and the relationships between those elements and on the extent to which there is redundancy or overlap in ecosystem function. The recognition of functional redundancy among species has led ecologists to increasingly focus on groups of species which perform "keystone functions" rather than on individual keystone species (Mills et al., 1993). This paradigm can be broadened to include physical and chemical processes, such as macropore formation by soil biota versus shrinking and swelling.

The third term, resilience, has been defined in at least three very different ways. The most common definition is that resilience is proportional to the recovery of the functional integrity of a system following a disturbance (Pimm, 1984). Others have argued that the term is more useful if it is defined as a capacity of the system to recover following catastrophic disturbances (Holling and Meffe, 1996) or following several simultaneous and/or repeated catastrophic disturbances or stressors. While this third definition makes an already complex concept even more difficult to assess, it may serve to better identify the key periods when ecosystem thresholds are likely to be exceeded. All three definitions are useful. The first, most common definition will be applied in this chapter except where specified.

## General Relationships

There are at least four specific relationships between soil erosion and rangeland soil quality (Figure 13.1). The first three include (1) the historic resistance of the soil to erosion, (2) the current resistance of the soil to erosion, and (3) the current resilience

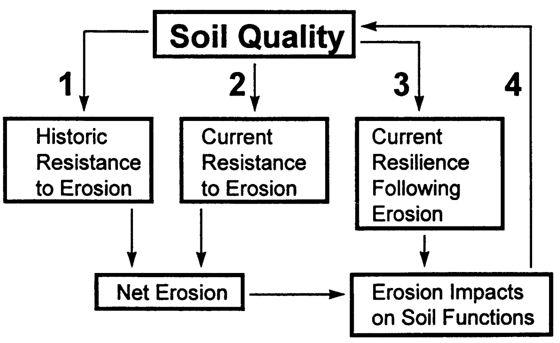



**FIGURE 13.1** Conceptual framework illustrating the four relationships between soil erosion and rangeland soil quality.

of the system following erosion. These relationships reflect past and potential future responses of a system to erosion and can serve as indicators of soil quality. The fourth relationship is related to the third: changes in soil properties following erosional events not only serve as indicators of soil quality but also reflect the capacity of soil erosion to modify soil quality. Soil erosion, then, is a determinant of (#4), as well as a response to (#1–3), soil quality (Figure 13.1).

## Historic Resistance

Soil properties which reflect the resistance of a system to soil erosion during the past several months or years are frequently suggested as indicators of soil quality (Arshad and Coen, 1992; Romig et al., 1995). The degree of pedestalling, heterogeneity of soil surface texture at the plant-interspace scale, the presence of rills, and signs of recent soil redeposition all indicate that some redistribution of soil resources has occurred. In linking soil loss to soil quality, the assumption is made that this loss results in a decline in the capacity of the ecosystem to fulfill one or more functions, such as water storage and nutrient supply. This assumption is frequently met in rangelands, which often occur on shallow soils.

While the relationship between the historic resistance of a system and past soil quality is relatively straightforward, the relationship between historic resistance and the indicators used to quantify it are not. The observed degree of pedestalling and the presence and characteristics of rills, for example, depend on a variety of factors. These factors include soil properties such as texture, the characteristics of the most recent storm(s), the time elapsed since the last storm, and the type and intensity of subsequent surface disturbances which could degrade or obscure the pedestals, rills, and depositional areas. Furthermore, identification of many of these features can be difficult without background knowledge of other processes occurring in the system. For example, soil accumulation around the bases of bunchgrasses in the Chihuahuan Desert is often attributed to pedestalling and/or deposition of material eroded from bare interspaces. While both processes can and do occur, much of the accumulation is frequently created by the activity of termites which bring soil to the surface and deposit it around standing dead vegetation. Consequently, indicators of historic resistance to soil erosion must be interpreted in the context of additional information and a knowledge of processes which may or may not be available for a specific site.

## Current Resistance

The current resistance of a soil to erosion depends on both soil properties and vegetation characteristics. Soil properties such as aggregate stability, hydraulic conductivity, and ground cover can be directly related to soil quality. Karlen and Stott (1994) selected aggregate structure, surface sealing, and porosity as key soil quality indicators related to soil erosion by water. These properties have been measured and correlated with data from natural runoff and rainfall simulation plots for storms of different intensities and durations (e.g., Benkobi et al., 1993), and the results have been used to generate empirical relationships between specific soil properties and resistance to soil erosion (Weltz et al., 1996).

Vegetation cover is frequently the most important factor affecting site resistance to interrill erosion on arid and semiarid rangelands (Wood et al., 1987; Blackburn et al., 1992). Spaeth et al. (1996b,c) point out that while hydrologists have typically focused on quantitative soil factors, vegetation parameters such as cover, above- and belowground growth form, phenology, and spatial distribution can and should be used to enhance predictions of soil resistance to erosion and susceptibility to runoff. Plant community composition may be used as a surrogate for many of these attributes (Spaeth et al., 1996b,c). Attempts have been made to establish vegetation cover guidelines for resistance to soil erosion at the site level. Studies have identified minimum cover values ranging from 20% in Kenya (Moore et al., 1979) to near 100% in Australia, while typical recommendations for the United States range from 50 to 75% (Packer, 1951; Nobel, 1965; Orr, 1970; Gifford, 1984). This variability in cover values is a function of the interaction between the applied stress (rainfall intensity and duration) and the resistance of the soil (soil erodibility).

Correlations between vegetation characteristics and soil erosion are related to the effects of the vegetation on soil properties and raindrop impact and overland flow patterns. In the case of soil properties, vegetation measures can be simply viewed as surrogate indicators for soil quality: the soil affects, and is affected by, vegetation growing in it. The relationship is less clear for raindrop impact: Is a soil of higher quality because it supports higher plant cover and therefore is better protected from raindrops? Similarly, is a soil of higher quality if it supports a plant spatial distribution which increases the residence time of water on a slope by increasing the tortuosity of flow paths? For agronomic crops which are removed every year, this would not be true. According to this perspective, the assessment of soil quality should be based on soil characteristics alone or on crop productivity where production serves to reflect differences in soil quality. With the exception of this caveat, the crop is largely viewed as a secondary, independent factor. In rangelands, however, it is more difficult to separate the vegetation from the soil. Many rangelands are dominated by perennial vegetation, and the vegetative community which exists on a soil has, in many cases, developed with that soil (Blackburn et al., 1992). This also applies to annual grasslands which effectively reseed themselves with a similar suite of species year after year. In light of the overwhelming impact of vegetation on resistance to rangeland soil erosion and the posited key relationship between soil erosion and soil quality, we argue that vegetation characteristics should, at a minimum, be used to interpret and apply soil quality assessments.

Indicators of current resistance or its inverse, soil erodibility, have been widely employed in soil erosion models such as the Universal Soil Loss Equation (USLE), Revised Universal Soil Loss Equation (RUSLE), and Water Erosion Prediction Project (WEPP) (Weltz et al., 1996). Process-based erosion simulation models such as WEPP provide the potential to study the many interactive effects of management practices as they affect soil erosion.

## Current Resilience

The resilience of a soil following erosion can be broadly defined in terms of the recovery of specific soil functions, such as infiltration and storage of water in the plant rooting zone (Figure 13.1, #3). More narrowly, however, it may be thought of in terms of the recovery of the resistance of the soil to future erosional events. In other words, to what extent does a single disturbance lead to increased susceptibility to future disturbances?

Resilience is one of the most critical issues to be addressed when assessing soil quality, yet it is also the most difficult to predict. Most studies have been designed to identify factors contributing to the resistance of systems to degradation. There are, however, some potentially useful, albeit largely untested, indicators of resilience following erosion. Soil depth is an obvious indicator of potential resilience, although the mere presence of a deep soil profile does not guarantee that surface soil structure will be regenerated following erosion, as illustrated by lateritic soils. Recent studies on the rate and extent of formation and reformation of soil aggregates in different soils (e.g., Chaney and Swift, 1986; Tisdall, 1996) and on soil organic matter regeneration (Reeder et al., in press) also provide some information on the relative importance of different soil constituents. However, vegetation and soil biotic indicators may be even more sensitive than any single soil chemical or physical parameter (Linden et al., 1994; National Research Council, 1994). This is particularly true for rangeland soils in which at least some of the vegetation remains intact or quickly regenerates following disturbance and where soil biotic communities are generally well established.

In addition to total plant cover, vegetation indicators of potential resilience might include the functional species composition and spatial distribution of the existing plant community, the soil seed bank, and the reservoir of seeds, including exotics, which are likely to disperse into a site following a catastrophic erosional event (National Research Council, 1994). Relevant questions include which species are likely to dominate following disturbance and what their likely impacts are on the regeneration of soil functional integrity and resistance to future disturbances. Although the vegetation recovery per se is not a direct indicator of soil resilience, the regeneration of soil functional integrity is inextricably linked to vegetation in most rangeland systems: the soil both affects and depends on the reestablishment of vegetative cover (DePuit and Redente, 1988).

Similar questions can be asked of the soil biotic community. In light of logistical difficulties in describing these communities, however, it is necessary to identify specific components which contribute directly to recovery processes, are relatively easy to census, and, most importantly, reflect the overall status of the living component of the soil. Recent attempts to isolate one or more key soil biotic indicators have yielded mixed results. In the case of ants, for example, studies in recovering Australian minelands on both basic biology (Majer, 1983) and community composition (Andersen, 1993) suggested that they should reflect soil biotic, if not physical, integrity. However, an extensive study recently completed in the Chihuahuan Desert yielded no significant relationships between ant community composition and site condition (W. Whitford, personal communication).

## Soil Erosion as a Driver

Soil erosion can also be viewed as a *driver* in the system which determines soil quality (Figure 13.1, #4). Soil erosion is generally considered to be detrimental to soil quality at the field scale. With few exceptions, soil loss is associated with a reduction in the capacity of soils to perform ecosystem functions. The impacts of soil erosion are generally greater than the proportion of the soil profile removed would suggest due to the concentration of soil organic matter and nutrients near the soil surface. This generally negative view of soil erosion is less applicable at the watershed scale, at which processes of both erosion and deposition must be considered. Virtually the entire country of Bangladesh, for example, owes its relatively high-quality soils to deposition of sediment eroded from the mountains outside of its borders. This perspective can also be usefully applied at the microcatchment scale to predict potential future changes in soil quality based on current depositional patterns (Watters et al., 1996). Thus, soil quality assessments may be enhanced by including multiple scales and by quantifying the impacts of changes at one scale, or on one part of the landscape, on soil quality at other scales and in connected landscape units.

# FACTORS INFLUENCING THE RELATIONSHIPS BETWEEN SOIL QUALITY AND SOIL EROSION

The nature of the four relationships between soil quality and soil erosion in each rangeland ecosystem depends on interactions among at least three sets of factors: (1) the characteristics of the disturbance regime associated with the current land use, (2) the characteristics of the disturbance regime(s) under which the current soil–landscape and soil–vegetation patterns developed, and (3) climate and landscape attributes, including degree of soil development and parent material. These three sets of factors largely determine the resistance of the soil to erosion, its resilience following erosion, and, ultimately, the impact of soil erosion on soil quality.

## Disturbance Regime

The disturbance regime for an ecosystem can be defined by five attributes: disturbance type or types, spatial scale, intensity, frequency, and predictability. The type of disturbance can be simply defined in terms of the event which causes it, such as fire, logging, grazing, or vehicle traffic. In order to compare different disturbances, however, it is more useful to break each event down into individual components which affect soil processes using a disturbance matrix (Table 13.1). For example, fires, logging, and grazing all remove aboveground biomass, thereby affecting litter and vegetation cover and organic matter supply. Logging, grazing, and vehicle traffic compact the soil, which affects runoff, water availability, and aeration. Grazing and logging are differentiated by the component of the biomass removed, the nature of the compaction, and the form and distribution of nutrients returned to the system.

**TABLE 13.1**
**Disturbance Matrix Illustrating Classification of Disturbance Events Based on Individual Components Which Affect Soil Processes**

| Disturbance event | Biomass removal | Soil compaction | Nutrient return | |
|---|---|---|---|---|
| | | | Form | Distribution |
| Fire | All | Diffuse[a] | Mineral | Follows vegetation |
| Logging | Woody | Linear | Unprocessed organic | Depends on practices |
| Grazing | Herbaceous | Linear and single point | Mineral and processed organic | Discrete, concentrated (dung); diffuse (trampled vegetation) |
| Vehicle traffic | All within wheel tracks | Linear | | |

[a] The effect here is indirect: fire increases the susceptibility of soil to crusting and compaction by removing the protective vegetation and litter layer.

The matrix in Table 13.1 can be expanded to include the scale of the disturbance. The impacts of a fire on runoff and soil erosion are much lower for a fire covering 10 $m^2$ than for one covering a section (square mile) or more. Similarly, the resilience of a system should be higher for the area covered by the smaller fire: this area would be immediately recolonized by organisms at the periphery. It would also be protected from future runoff events by the intact vegetation surrounding the affected area.

Disturbance frequency and intensity could also be added to the matrix in Table 13.1. The capacity of systems to recover from frequent, intensive compactive disturbances (such as a road or cattle path) is much lower than their capacity to recover from an occasional perturbation. In addition to the frequency, the timing of the disturbance can be very important. In the case of compaction, timing affects both resistance (which varies as a function of soil moisture) and resilience (which varies temporally with both biotic activity and physical processes such as frost heave).

Disturbance predictability affects plant and soil community composition. Long-lived perennial plants are often adapted to predictable disturbance. Annuals, conversely, tend to have large seed banks that allow them to quickly recover from a wide variety of disturbances (Barbour et al., 1987). Similarly, much of the soil biota in ecosystems dominated by predictable seasonal drought survive with well-adapted systems of timed reproduction. Populations of these same species, however, can be significantly reduced by a wet season drought (Steinberger and Whitford, 1984; Steinberger et al., 1984).

## Disturbance History

The disturbance history of an ecosystem rarely can be used to predict the potential impact of a new disturbance regime. For example, the North American tallgrass prairie evolved under a disturbance regime which included high-intensity grazing by bison. Consequently, it should be both more resistant and more resilient under grazing than much of the Colorado Plateau. The current plant and soil communities of the Colorado Plateau may have never been impacted by large groups of large herbivores (Belnap, 1995). Identification of mechanisms of resistance to erosion can yield additional information on the potential resistance to new types of disturbances. Soils on the Colorado Plateau are relatively resistant to erosion, even after prolonged droughts, due to the stabilizing effects of soil surface cryptogams (primarily lichens). When the physical integrity of the soil surface is disrupted by new disturbances such as trampling, off-road vehicles, or mountain bikes, the resistance is lost (Anderson et al., 1982a). However, the resilience (rate of recovery) can be increased by carefully timing disturbances, such as grazing, to promote regeneration of the biologically active soil surface (Anderson et al., 1982b). This idea of timing disturbances to increase resilience is similar to more common recommendations to time grazing to coincide with periods of high resistance, such as when the soil is drier (Warren et al., 1986).

A combination of this historical perspective and a consideration of the attributes of each disturbance regime may be useful in resolving debates over grazing on public lands. Many of these debates are based on different perceptions of the impacts of grazing on soil quality and soil erosion. While the majority of studies have concluded that grazing generally increases runoff and erosion (Spaeth et al., 1996c), significant improvements under grazing in soil properties that are related to soil hydrology, such as organic matter content, have been recorded. Positive responses, such as that reported by Manley et al. (1995), are more frequently found for ecosystems with a history of large herbivore grazing. Contemporary attempts to improve soil quality and rangeland health under grazing in ecosystems in the southwestern United States (where historic grazing was probably intermediate between the Great Plains and the Colorado Plateau) are based on careful management of the intensity and frequency of fires and grazing with respect to precipitation.

## Climate and Landscape

The ultimate impact of any disturbance regime on soil erosion and soil quality depends on interactions with inherent climate, soil, and landscape characteristics. Timing and characteristics of storms, slope, landscape position, topographic complexity, soil depth, and parent material affect resistance and resilience. Measurements of a soil's resistance to detachment by overland flow are more relevant if they are made during the season when precipitation events are likely to exceed infiltration capacity.

## Factor-Based Approaches to Assessment

The above discussion suggests that rangeland soil quality assessments can be enhanced by considering the climate and landscape context together with the anticipated disturbance regime. An erosion-based assessment of soil quality should be completed in three interdependent stages. In the first stage, point and field-scale measurements of soil properties are scored using standard scoring functions and combined to generate a standard index (e.g., Karlen and Stott, 1994; Yakowitz et al., 1993). A typical disturbance regime and climate for the region are assumed for this stage. In the second stage, the scoring functions are modified according to local climate and landscape conditions, while the disturbance regime is held constant. In the third stage, the scoring functions are further modified to reflect different disturbance regime scenarios. At this stage, multiple assessments are possible, depending on the scenario selected. For example, a soil in an arid region which is protected from wind erosion by physical crusts may have high resistance to erosion under a disturbance regime of rainy season grazing (when physical crusts rapidly reform following rainstorms) but low resistance under a dry season or continuous grazing regime. Conversely, its long-term resilience may be lower under rainy season grazing due to the potentially negative impacts of growing season grazing on biomass production and subsequent soil carbon input.

## CASE STUDIES

The Jornada Experimental Range, Walnut Gulch Experimental Watershed, and Central Plains Experimental Range (CPER) represent three distinct rangeland ecosystems (Figure 13.2). When compared with the global diversity of rangelands, these three sites appear relatively similar. They all lie within 1000 km of each other and

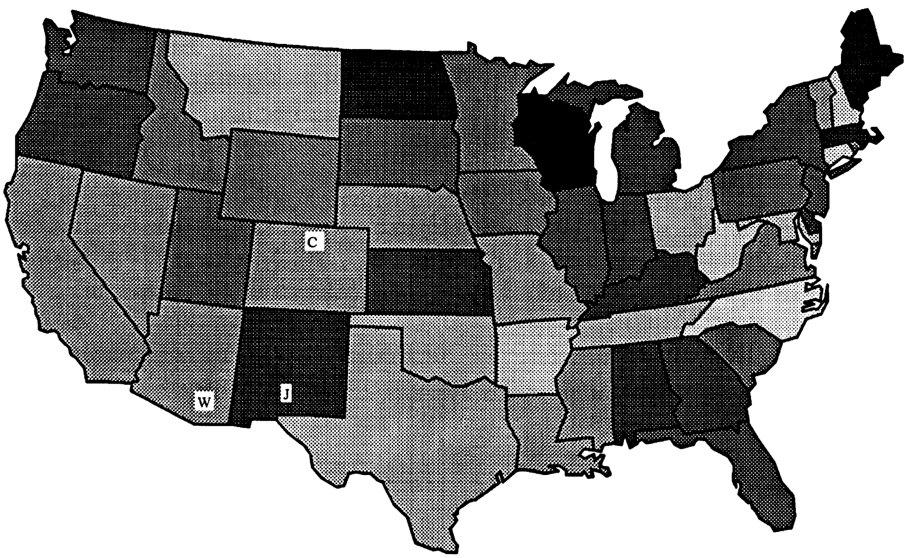



**FIGURE 13.2** Locations of the three case studies: Central Plains Experimental Range (C), Jornada Experimental Range (J), and Walnut Gulch Experimental Watershed (W).

are classified as semiarid, with average annual precipitation ranging from 240 mm at the Jornada to 325 mm at the CPER. However, unique relationships between soil quality and soil erosion exist at each site. Post-European colonization changes in vegetation structure are believed to be associated with increased erosion and declines in soil quality at both the Jornada and Walnut Gulch. A change in disturbance regime is thought to be at least partially responsible in both cases. Water erosion has played a major role at the two locations, but much of the Jornada has also been severely altered by wind erosion. Unlike the Jornada or Walnut Gulch, the CPER continues to be dominated by grasses, although there is evidence that some changes in species composition have occurred with heavy grazing. With the exception of areas subjected to prolonged overgrazing coupled with drought and areas converted to croplands, the post-European colonization disturbance regime does not appear to have had a negative impact on soil quality or led to sharp increases in soil erosion on rangelands at the CPER.

## Jornada Experimental Range

The Jornada Experimental Range is located in a closed basin in the northern Chihuahuan Desert. The elevation varies from 1190 to 1372 m. During the past 150 years, the plant community covering much of the basin, including most of the soils with a high sand content, has shifted from black grama- (*Bouteloua eriopoda*) dominated grassland (Figure 13.3a) to mesquite- (*Prosopis glandulosa*) dominated shrubland (Figure 13.3b). This shift reflects reductions in soil quality which have both led to and been reinforced by increased wind erosion (Buffington and Herbel, 1965; Gibbens et al., 1983). In many areas, eolian dunes up to several meters in height have developed around individual mesquite shrubs (Gould, 1982), leaving the wind-scalped interdunal areas largely devoid of perennial vegetation (Hennessy et al., 1983).

This transition is believed to be the result of an interaction between climate (drought) and a change in the disturbance regime (Nelson, 1934; Schlesinger et al., 1990). The historic disturbance regime consisted of unpredictable, large-scale droughts and more regular small-scale animal-induced soil surface and grazing disturbances. Antelope, deer, rabbits, prairie dogs, kangaroo rats, and a variety of other rodents all generated soil surface disturbances. All of these species are still present, with the exception of prairie dogs. Unlike the CPER (see below), there is little evidence to suggest that bison played a significant role in this system. With the introduction of cattle and water development, the intensity of grazing disturbances increased. This, together with the simultaneous invasion by mesquite (dispersed by the cattle) of the sandy grassland soils, is hypothesized to have caused the breakdown of the system during severe droughts (Nelson, 1934; Herbel et al., 1972). The system was not resistant to the new disturbance regime, nor was it resilient: grasses have not recovered even where cattle have been excluded for over 50 years (R.P. Gibbens, personal communication). In this case, the third definition of resilience discussed in the introduction is perhaps the most relevant insofar as it focuses on the capacity of the system to recover from multiple, simultaneous assaults.

(a)

(b)

**FIGURE 13.3** Chihuahuan Desert grassland on the Jornada Experimental Range with soaptree yucca (*Yucca elata*) and invading mesquite (*Prosopis glandulosa*) in background (a) and former grassland now dominated by mesquite (b).

**TABLE 13.2**
**Average Soil Surface (0–5 mm) Structural Stability at Three Sites as Measured by a Field Wet Aggregate Stability Test in Which Air-Dry Soil Fragments Are Gently Sieved in Distilled Water 5 Minutes After Submersion**

| Site | Grass cover (%) | Bare[a] | Grass[a] | Shrub[a] | Average (weighted by vegetative cover) |
|---|---|---|---|---|---|
| Mesquite dune | 3.9 | 1.3 | 4.2 | 4.4 | 1.7 |
| Grassland | 23.4 | 1.3 | 5.1 | 4.8 | 2.2 |
| Grassland — exclosure | 25.3 | 2.4 | 5.0 | 5.6 | 3.4 |

*Note*: A value of 1 indicates that the visible structure of the fragment disintegrated within 5 sec of submersion, while a value of 6 was assigned to fragments which remained at least 75% intact after sieving. All sites are located within 750 m of each other and all contain 81–84% sand in the top 10 cm.

[a] de Soyza et al., in press.

From a soil quality perspective, three particularly significant changes have occurred. The first is that there has been a net loss of soil resources from the mesquite dune areas. Gibbens et al. (1983) recorded a net loss of 4.6 cm of soil from a 259-ha pasture over a period of 45 years. In this exclosure, the existing mesquite was killed and grasses were planted in the mid-1930s. By 1980, the shrubs had become reestablished and dunes had reformed. The second change is in soil texture. Based on soil samples taken from the same site, Hennessy et al. (1986) concluded that the material lost was confined to the silt and clay fractions. Many of these soils have sand contents well in excess of 80%, further magnifying the impacts of the loss of the fine fractions on soil aggregate stability and water and nutrient retention. Preliminary studies of soil aggregate stability on these sandy soils suggest that a reduction in soil surface structural stability occurs relatively early in the transition from grassland to shrubland (Table 13.2). This would suggest that soil resistance to disturbance in the grassland system is quite low and that it is only the protective grass cover which prevents catastrophic wind and water erosion from occurring (Figure 13.3b).

This net reduction in soil quality based on average site characteristics is reinforced by a third change: a redistribution of remaining resources within the site. Organic matter and associated nutrients tend to accumulate beneath shrubs (Schlesinger et al., 1990, 1996; Virginia et al., 1992). This tends to stratify the system into relatively high-quality microsites associated with shrubs and lower quality microsites in the interspaces. This is reflected in increased heterogeneity in soil structural stability (Table 13.2) and infiltration capacity (Table 13.3), with the highest stability and infiltration occurring beneath shrub canopies.

As a result of this heterogeneous distribution of resources, the interspace soil is both less resistant to erosion (Figure 13.1, #2) and less resilient after erosion has

**TABLE 13.3**
**Relative Infiltration Capacity as Indicated by Time Required for 2.5 cm of Water to Infiltrate Saturated Soil from a 15-cm Ring Inserted to a Depth of 1.5 cm for Sites Listed in Table 13.2**

| Site | Min:sec | | | Average (weighted by vegetative cover) |
|---|---|---|---|---|
| | Bare | Grass | Shrub | |
| Mesquite dune | 8:38 (3:13) | 2:48 (0:38) | 2:12 (0:40) | 7:18 |
| Grassland | 13:05 (3:14) | 1:15 (0:33) | 0:46 (0:10) | 9:27 |
| Grassland — exclosure | 6:02 (1:04) | 2:05 (0:38) | 1:20 (0:28) | 4:25 |

*Note*: Mean and standard deviation for $n = 3$.

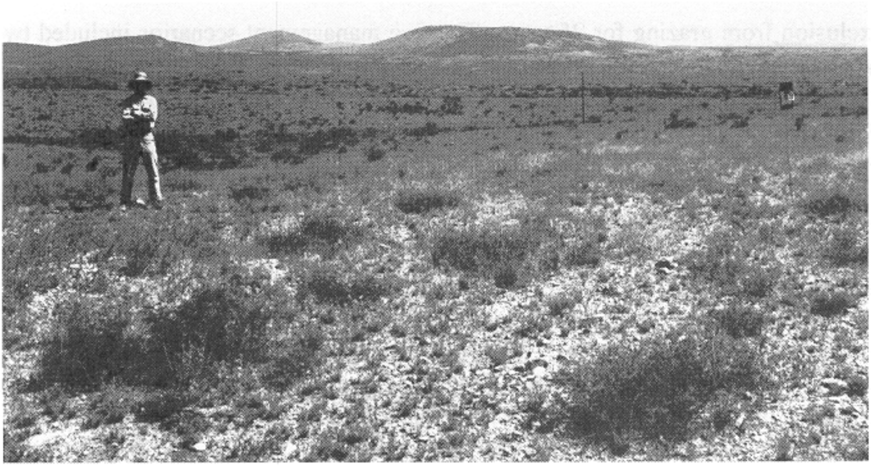
occurred (Figure 13.1, #3). This then leads to a reduction in soil quality (Figure 13.1, #4) and results in a negative feedback loop of increasingly depleted interspaces and enriched shrub microsites (Schlesinger et al., 1990).

## Walnut Gulch

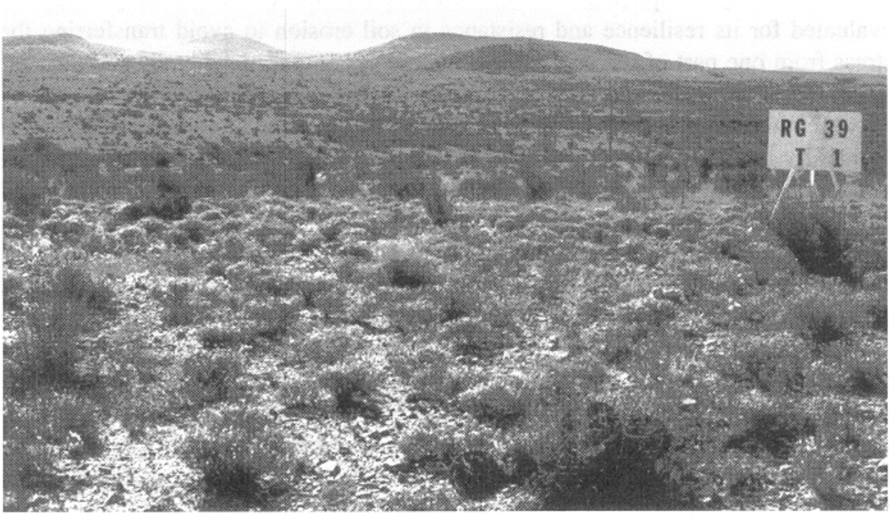
The Walnut Gulch Experimental Watershed encompasses an area of 150 km$^2$ that surrounds Tombstone, in southeastern Arizona. Elevation of the watershed ranges from 1250 to 1585 m. The watershed is located primarily in a high foothill alluvial fan portion of the San Pedro River watershed (Renard et al., 1993; Weltz et al., 1996). Soils of the watershed reflect the parent material, with limestone-influenced alluvial fill as the dominant source. These soils are generally well-drained, calcareous, gravelly loams with a large percentage of rock and gravel on the soil surface (Breckenfield, 1996). Erosion pavement can exceed 70% on steep eroded hillslopes and typically ranges from 35 to 50%.

The watershed is in a transition zone between the Chihuahuan and Sonoran Desert plant communities (Figure 13.4). Historical records on plant community composition are limited but indicate that, like the Jornada, a larger percentage of the watershed was grass prior to European settlement in the late nineteenth century. Currently, the lower two-thirds of the watershed is dominated by shrubs that include creosote bush (*Larrea tridentata*), whitethorn (*Acacia constricta*), tarbush (*Fluorensia cernua*), burroweed (*Haplopappus tenuisectus*), and snakeweed (*Gutierrezia sarothrae*). The upper third of the watershed is dominated by desert grassland plant communities. Dominant grasses are black grama (*Bouteloua eriopoda*), blue grama (*B. gracilis*), and bush muhly (*Muhlenbergia porteri*).

The climate of Walnut Gulch watershed is classified as semiarid with frequent local droughts. Precipitation varies considerably both seasonally and annually. Average annual precipitation for the period 1956–90 was 312 mm with a standard deviation of 79 mm. Approximately two-thirds of the annual precipitation occurs as high-intensity, convective thunderstorms of limited areal extent during the summer monsoon period of June through September. Runoff and soil erosion on Walnut Gulch result almost exclusively from these summer convective storms.



(a)




(b)

**FIGURE 13.4** Transition zone between Chihuahuan and Sonoran deserts at the Walnut Gulch Experimental Watershed. Photo point comparison shows an increase in burroweed and desert zinnia from 1967 (a) to 1994 (b).

The WEPP model was used to predict the impact of five management actions on soil erosion at the Lucky Hills watershed, a small instrumented catchment on Walnut Gulch (Renard et al., 1993). Lucky Hills is dominated by creosote bush and whitethorn, with an average canopy cover of 28% and ground cover of 56% (primarily rock and gravel cover). Little or no herbaceous vegetation exists in the watershed despite exclusion from grazing for 25 years. The five management scenarios included two types of disturbance and a range of disturbance intensities. Existing vegetation was maintained in two of the scenarios. In one of these, moderate grazing was imposed. In the other scenarios, three levels of grazing intensity (none, moderate, and heavy) were imposed on areas following herbicide-based shrub removal and grass reseeding.

The model simulation results show that grazing intensity increased water yield, sediment yield, and the magnitude of the 2-year frequency peak discharge over the 15-year simulation period. The most important effect of management was on hillslope sediment yield. Converting from shrub to grass with no grazing decreased hillslope sediment yield by 91%. However, this decrease translated into a much smaller (25%) decrease in reduction of sediment yield from the watershed. The sediment yield entering the channel decreased significantly, but runoff amounts and peak discharge rates did not decrease. This resulted in an increase in channel scour and a net decrease in watershed sediment yield of only 25%.

If on-site evaluation is limited to upland areas, then shifts in system stability related to hillslope, riparian corridor, and stream channel areas may go unrecognized. The major implication of this work is that the entire landscape must be evaluated for its resilience and resistance to soil erosion to avoid transferring the stress from one part of the landscape (hillslope) to another (channels) and destabilizing the entire landscape through complex feedback interactions.

A second study conducted at Walnut Gulch illustrates the importance of temporal variability in this ecosystem. Monthly evaluations of erosion rates using a rotating boom rainfall simulator (Swanson, 1965) were made on two treatments (natural and clipped) at three soil moisture contents on a Haplargid soil (Simanton et al., 1991) and compared to the soil erodibility factor (K) of the RUSLE model (Figure 13.5). The K factor of the RUSLE model is varied throughout the year and is a function of frost-free period and average annual erosivity (R: MJ*mm/ha*h*yr).

Measured erosion rates were lowest between May and July and highest in November. This is in complete contradiction to RUSLE model estimates (Figure 13.5). The discrepancy in the cycle of soil erodibility extremes may be due to the lack of freeze–thaw intensity in this Haplargid soil as compared to the soils from which the RUSLE K algorithm was developed (i.e., cropland soils from the east and midwestern United States) and needs to be modified to address rangeland conditions. Time-related changes in erosion rates associated with rangeland treatment need to be evaluated over a multiyear period using multiplot studies. Biotic factors, both flora and fauna, appear to significantly influence the temporal variability of soil quality and need to be considered before we can adequately define the interactions between soil quality and soil erosion in this ecosystem.

## Central Plains Experimental Range

The CPER, established in 1937 to evaluate and develop improved management practices for fragile grasslands, is located in northeastern Colorado on the shortgrass steppe of the western Great Plains (Figure 13.6). The region is characterized by low but highly variable rainfall, frequent droughts, high evapotranspiration, and a short

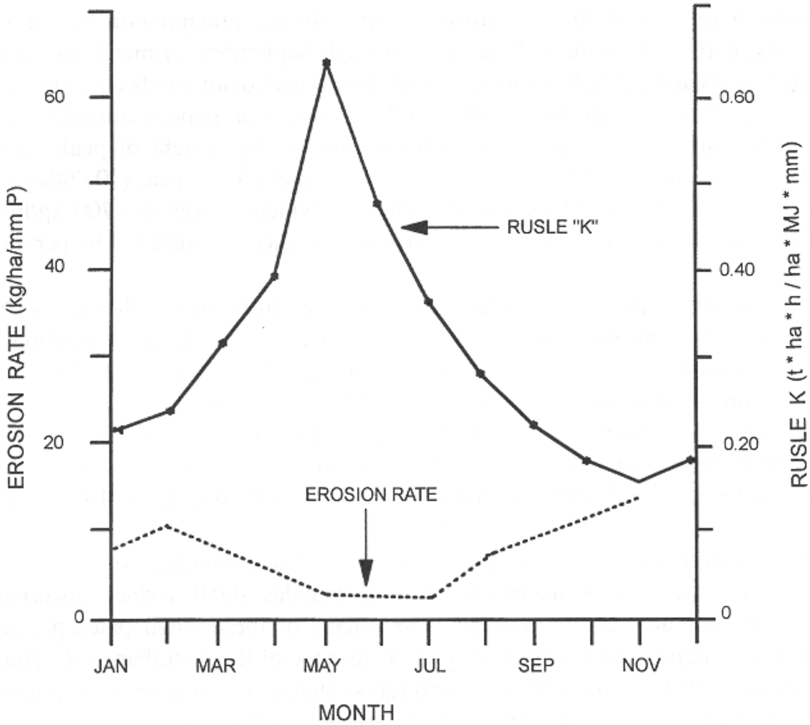



**FIGURE 13.5** Monthly measured erosion rate versus RUSLE estimated monthly K from the Walnut Gulch Experimental Watershed.

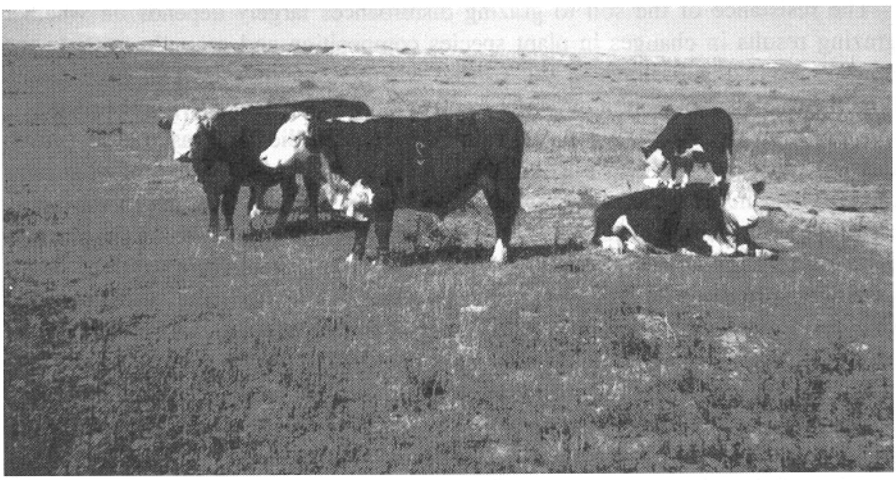


**FIGURE 13.6** Semiarid shortgrass rangeland at the Central Plains Experimental Range.

growing season (average 133 frost-free days). Annual precipitation averages 325 mm (range 109–580 mm, 80% in April through September, primarily as thunderstorms). Soils at the CPER are mostly sandy loams and loamy sands on a topography of rolling hills at elevations of 1600–1700 m. Vegetation is predominantly grasses (48–70% warm season and 8–10% cool season, as dry weight of peak standing crop), with shrubs (1–11%), forbs (5–9%), and plains prickly pear (10–24%) (Sims et al., 1978; ARS, unpublished data). Annual production averages 700 kg/ha, but varies from 300 to 1700 kg/ha, while carrying capacity averages 1.7 ha per animal unit month.

Grasses and forbs have been the dominant vegetation on the shortgrass steppe since the early Cenozoic, and herbivores have coevolved with the vegetation. The plant community has therefore experienced a long history of relatively heavy grazing pressure (Laurenroth and Milchunas, 1992). Fire has been a dominant force in maintaining the integrity of the prairie plant community (Reichman, 1987), and periodic extreme climatic events, such as floods or dust storms, have played a significant role in pedologic additions and translocations over time (Blecker et al., in press).

In addition to grazing by cattle and wildlife, extreme climatic events, and occasional fires, the current disturbance regime includes small surface disturbances caused by ants and small mammals. The effects of these small patch-producing disturbances depend on the frequency of occurrence of the disturbance (Coffin and Laurenroth, 1988). Tolerance of the shortgrass steppe to disturbance by grazing is well documented and is due in large part to the tolerance of the dominant grass species, blue grama, to heavy grazing (Klipple and Costello, 1960). A 12-year study at the CPER by Hyder et al. (1975) to evaluate repeated heavy grazing and N fertilization strategies suggested that, unlike other rangelands, the standing biomass of perennial and annual species of the shortgrass steppe tends to increase or decrease more in response to weather conditions than to heavy grazing.

The resistance of the soil to grazing disturbances largely depends on whether grazing results in changes in plant species composition and ground cover. In the long-term grazing pastures at the CPER, where a healthy stand of blue grama dominates the plant community irrespective of grazing intensity, soil erosion is minimal from normal high-intensity rainfall events. Studies by Frasier et al. (1995) indicated that soil loss was negligible when simulated rainfall was applied at rates of 55–110 mm/hr (a range of intensities common to thunderstorms in the area), although total runoff quantities and rates were higher from heavily grazed pastures than from lightly grazed pastures. The lower infiltration rates as a result of heavy grazing improved within 2 years after removing cattle from the heavily grazed pastures. Lower infiltration rates with heavy stocking rates also have been reported on mixed grass rangeland (Abdel-Magid et al., 1987).

Studies are currently under way at the CPER to evaluate effects of long-term grazing on other soil quality parameters such as organic matter content. The quantity and distribution of organic matter in a rangeland soil depend on the rooting characteristics of the plant community. Changes in plant species composition due to grazing pressure and consequent changes in total root biomass and distribution within the soil profile can alter soil organic matter concentration, composition, and distribution in the soil profile. Increases in surface soil organic carbon (C) have been reported for a mixed grass prairie as the result of grazing-induced reductions in needle-and-thread, a deep-rooted species, and increases in blue grama, a shallow-rooted species (Smoliak et al., 1972). Increases in mixed grass prairie soil C have also been attributed to grazing-enhanced decomposition of standing dead and surface litter (Manley et al., 1995). On the shortgrass prairie at the CPER, where blue grama has remained the dominant plant species irrespective of grazing intensity and where average annual production is low, preliminary investigations have revealed no significant differences in A-horizon organic C and N concentrations in 55-year-grazed versus 55-year-ungrazed pastures. However, because of higher soil bulk density with heavy grazing, the total organic C content of the A horizon in the heavily grazed pastures is significantly higher than the C content of nongrazed exclosures (G. Schuman and J. Reeder, ARS, unpublished data). Other studies conducted at the CPER have demonstrated that variability in soil organic matter content can be as high within a grazing treatment as between grazing treatments due to natural differences between bare soil and soil under plants (Hook et al., 1991; Burke et al., 1995).

Although the shortgrass prairie displays a high degree of resistance to grazing, it is not highly resilient where prolonged overgrazing coupled with drought conditions has degraded the plant community (Shoop et al., 1989). The soils of the region are highly erodible when not protected by vegetation, and attempts at renovation of damaged grasslands by seeding native species are largely unsuccessful because seed production by blue grama is low and variable (Coffin and Laurenroth, 1992), and the seedling morphology of blue grama is not well adapted to the low and sporadic precipitation common to the area (Hyder et al., 1975).

## RANGELAND HEALTH AND SOIL QUALITY

While soil quality and soil health have emerged as new paradigms for assessing ecosystem condition in cultivated systems around the world, the term rangeland health has been proposed to refer to evolving approaches to the assessment and monitoring of noncultivated, nonforested lands. The National Research Council's Committee on Rangeland Classification (National Research Council, 1994) proposed three criteria for determining whether a rangeland is healthy, at risk, or unhealthy: "degree of soil stability and watershed function, integrity of nutrient cycles and energy flow, and presence of functioning recovery mechanisms." The first criterion is directly related to soil quality, while the latter two ultimately depend on soil stability and watershed function.

Tongway (1994) has proposed an approach which relies heavily on soil surface characteristics and other indicators of soil quality. This approach is being used to evaluate rangeland health in Australia. One characteristic of a soil quality–based approach to rangeland health assessment is that it depends on plant community structure rather than species composition. This approach has the potential to resolve contentious issues related to defining the preferred or ideal plant community for a site. In many areas, including most of the Chihuahuan Desert, soil quality is highly correlated with the desired potential natural community (PNC). However, exceptions do occur. For example, while buffalograss (*Buchloe dactyloides*) is a component of the native plant community in the Great Plains, and therefore is associated with good range condition, infiltration tends to be lower in buffalograss stands even when texture and organic matter are constant (Spaeth et al., 1996b). Where soil quality and comparisons with the PNC yield different conclusions, either one can be assigned precedence or the two can be combined using the weighting system discussed above (Karlen and Stott, 1994; Herrick and Whitford, 1995).

## CONCLUSIONS

The link between soil quality and soil erosion has been clearly established in the popular press. In a recent *Los Angeles Times* article detailing the impacts of the mid-1990s drought in the Southwest, a social scientist with the National Center for Atmospheric Research was quoted as stating that "We shouldn't blame nature for destroying a lot of property. People keep moving in harm's way by building homes in fire-prone mountains, and farming and running cattle in regions of poor soil quality."

This intuitive link between soil erosion and soil quality is supported by four specific relationships. Soil quality is reflected in the historic and current resistance to erosion and in the current resilience following erosion. Erosion is also a key determinant of soil quality in many systems. The nature of these relationships is determined by the climate and the landscape together with the current disturbance regime and its similarity to the historic disturbance regime(s) under which the soil and biotic communities evolved.

These relationships between soil erosion and soil quality lead to a number of implications for the assessment of soil quality. Soil quality must be evaluated at a variety of spatial scales in order to incorporate the wide range of scales at which soil erosion processes occur. Ideally, the measurements at these scales should be linked to each other in parallel with hydrologic and eolian linkages in the landscape. Another implication of these relationships is that the current resistance to, and resilience following, erosion must be interpreted in the context of the current and historic disturbance regime. A soil which is highly resistant to erosion following drought may lose this resistance when the drought is combined with fire or intensive grazing. A third implication follows from the first two: the resistance of a soil to erosion ultimately depends on the spatial scale at which erosion is measured and on the intensity of the disturbance event which is assumed. Redeposition of soil within a landscape can disguise the magnitude of the potential sediment yield from a watershed until a 100-year storm occurs.

A final implication of the relationships between soil erosion and soil quality is that the resilience, as well as the resistance, of the soil must be incorporated into assessments if the long-term sustainability of the system is to be evaluated. While most models currently focus on the resistance of the system to soil erosion, many systems, such as soils on recent deep volcanic materials and alluvium, have such a high resilience that resistance is relatively less important. Conversely, many soils which are shallow, have lost most of their biota, or are in areas with severe climatic limitations have virtually no resilience. This last point illustrates the need to refine and improve definitions of "soil loss tolerance" as part of the effort to define soil quality and its relationship to soil erosion.

Future research on relationships between disturbance regimes, soil quality, and soil erosion should be conducted at three time scales: (1) immediately postdisturbance, for direct impacts on soil properties; (2) medium term, for impacts on soil biota and vegetation (including growth, growth form, and biomass allocation [root:shoot ratios]) which affect soil resistance and/or resilience; and (3) long term, for impacts on plant and soil biota community composition. Very few long-term disturbance studies exist, with the exception of some on grazing, and most of these do not include a hydrological component.

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