

Land degradation in drylands: Interactions among hydrologic–aeolian erosion and vegetation dynamics

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ARTICLE INFO

Article history:

Received 2 May 2009

Received in revised form 28 September 2009

Accepted 18 November 2009

Available online 26 November 2009

Keywords:

Drylands

Ecogeomorphology

Land degradation

Sediment transport

Vegetation patterns

ABSTRACT

Land degradation in drylands is one of the major environmental issues of the 21st century particularly due to its impact on world food security and environmental quality. Climate change, shifts in vegetation composition, accelerated soil erosion processes, and disturbances have rendered these landscapes susceptible to rapid degradation that has important feedbacks on regional climate and desertification. Even though the role of hydrologic–aeolian erosion and vegetation dynamic processes in accelerating land degradation is well recognized, most studies have concentrated only on the role of one or two of these components, and not on the interactions among all three. Drawing on relevant published studies, here we review recent contributions to the study of biotic and abiotic drivers of dryland degradation and we propose a more holistic perspective of the interactions between wind and water erosion processes in dryland systems, how these processes affect vegetation patterns and how vegetation patterns, in turn, affect these processes. Notably, changing climate and land use have resulted in rapid vegetation shifts, which alter the rates and patterns of soil erosion in dryland systems. With the predicted increase in aridity and an increase in the frequency of droughts in drylands around the world, there could be an increasing dominance of abiotic controls of land degradation, in particular hydrologic and aeolian soil erosion processes. Further, changes in climate may alter the relative importance of wind versus water erosion in dryland ecosystems. Therefore acquiring a more holistic perspective of the interactions among hydrologic–aeolian erosion and vegetation dynamic processes is fundamental to quantifying and modeling land degradation processes in drylands in changing climate, disturbance regimes and management scenarios.

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1. Introduction: dryland degradation

Drylands of the world are of critical concern as they cover over 41% of the surface of the earth with over 2 billion inhabitants, mostly in the developing world (MEA, 2005). Land degradation in drylands is one of the major environmental issues of the 21st century because of its impact on food security and environmental quality (MEA, 2005). The United Nations Convention to Combat Desertification (UNCCD) defined desertification as “land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities”. The causes and consequences of dryland degradation, sometimes arguably referred to as “desertification”, remain controversial and poorly understood (Hutchinson, 1996; Thomas, 1997; Veron et al., 2006). Some authors even consider desertification as a potential but not necessarily an outcome of land degradation processes. Desertification results in environmental and

socio-economic-political changes through a complex interplay of biophysical and anthropogenic factors that act at different scales (Geist and Lambin, 2004). Because desertification is considered to be a cause and a consequence of poverty, the mitigation of desertification could aid in reducing poverty in dryland areas (MEA, 2005). In the case of many dryland systems, severe climatic conditions (e.g., the series of droughts such as those that affected sub-Saharan Africa since the late 1960s) combined with weak economies and unsustainable use of marginal resources, have increased the levels of stress in dryland ecosystems, which were often unable to sustain the pressure of the increasing human population (Darkoh, 1998). Collectively, these factors caused famine and large-scale human migrations that have had important regional-scale socio-economic and political consequences (Mabbutt and Wilson, 1980; Darkoh, 1998).

Human activities have a profound influence on the degradation trends and patterns in drylands (Reynolds et al., 2007). A classic example of anthropogenic degradation of drylands is the “Dust Bowl” period that occurred in the Great Plains of the United States during the 1930s, when dramatic soil loss and dust emissions were triggered by poor land management practices and concurrent dry climatic conditions

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(Worster, 1979). Anthropogenic disturbances of dryland soils after the European colonization and consequent shifts in vegetation patterns have also been reported in the southwestern United States, Australia, Southern Africa, and South America (Pickup, 1998; Van Auken, 2000; Ravi et al., 2009a). Anthropogenic pressures include the overgrazing of rangelands, which comprise around 70% of drylands, and conversions to cropland. These pressures are exacerbated by climatic changes, urbanization and management factors. A notable example is the Sahel region of Africa (Nicholson et al., 1998; Taylor et al., 2002), where a feedback between vegetation and climate can induce the existence of two alternative stable states: a stable dry (desertified) and a stable moister (vegetated) climate regime (Charney, 1975; Xue and Shukla, 1993; Zeng and Neelin, 2000). In this system a decrease in vegetation cover may result into a highly irreversible shift to a dry climate state, in which rainfall would be insufficient to allow for the recovery of vegetation (Brovkin et al., 1998).

Degradation of soil and vegetation can lead to substantial reduction in ecosystem functions and services, perhaps in as much as 70% of drylands (Dregne and Chou, 1992). Regardless of the actual amount, land degradation is clearly occurring at an alarming pace, threatening the food security and environmental quality in drylands (UNCCD, 1994; MEA, 2005). Notably, soil erosion is the most widespread form of land degradation in these landscapes. Recent studies indicate that triggering factors of land degradation, such as global climate change, have resulted in drier conditions in arid and semi-arid regions (Held et al., 2005; Burke et al., 2006; Seager et al., 2007). An increase in aridity can result in an increase in relative importance of abiotic factors that propagate land degradation, such as aeolian and hydrological transport processes. Indeed, wind and water erosion are considered to have contributed to 87% of the degraded land (Middleton and Thomas, 1997). Grazing pressure, loss of vegetation cover, and the lack of adequate soil conservation practices render soils in these regions more susceptible to processes of soil erosion, which in turn can have important impacts on regional climate and desertification (Nicholson et al., 1998). Even though the role of hydrologic–aeolian erosion and changing vegetation dynamics in accelerating land degradation is well recognized, our understanding of how the interactions among different forms of soil erosion and vegetation dynamic processes contribute to desertification is generally still uncertain. Many studies have concentrated only on the role of one or two of these components, and not on the interactions among these processes (Fig. 1). Based on a more holistic perspective of relevant studies, here we review recent studies on the interactions between wind and water erosion processes in dryland systems, and discuss how these processes affect vegetation patterns and how vegetation patterns, in turn, affect these abiotic processes. We focus

on (1) interrelationships between wind and water erosion; (2) hydrologic–aeolian erosion and changing vegetation dynamic processes; and (3) applications, including the increasing need to estimate long-term changes in food and fodder supply, to design and evaluate soil conservation and land reclamation programs, assessment of the rate of entrainment of dust into the atmosphere and its contribution to global climate change, and the effect of climate change and land management scenarios on arid and semi-arid regions.

2. Wind and water erosion in drylands

Soil erosion can be defined as the detachment and transport of soil particles and subsequent redeposition in near or distant areas mainly by the action of wind and water. Soil erosion is a natural land surface process, which can be accelerated and exacerbated by anthropogenic and biophysical factors with adverse effects on soil resources, crop productivity, environmental quality and climate (Lal, 1994).

Archeological evidences indicate that accelerated soil erosion emerged as a serious environmental issue as early as 8000 years ago. Since then, it has often threatened the existence of civilizations (e.g., Redman, 1999; Rolett and Diamond, 2004; Montgomery, 2007). On a global scale, 1094 million ha are affected by water erosion and 550 million ha by wind erosion (Middleton and Thomas, 1997). Even though, globally, water is the major contributor to soil erosion, in many arid and semi-arid systems, erosion by wind can be substantial and even the dominant (Breshears et al., 2003; Field et al., 2009). For example, the cultivated soils in the Great Plains of North America are particularly prone to the action of winds, and rates of wind erosion may exceed those of water erosion. In this region, dramatic soil losses and dust emissions were induced in the 1930s (the “dust bowl” period) by poor land management and drought conditions (Worster, 1979).

Soil erosion is often considered as a cause and an effect of desertification (Nicholson et al. 1998; Lal, 2001; MEA, 2005) and important feedbacks have been shown to exist among erosion, biodiversity loss and climate change (Fig. 2). An increase in rates of soil erosion because of climatic changes that increase aridity could result in enhanced loss of soil resources and a loss in biodiversity, which can further increase rates of soil erosion and result in loss of vital services from drylands, including the possible reduction in primary production and carbon sequestration (e.g., Chapin et al., 1997). Moreover, biodiversity loss has been related to a decrease in ecosystem resilience—the ability of the ecosystem to recover from disturbances (e.g., Elmqvist et al., 2003). Therefore, less diverse ecosystems are more prone to highly irreversible shifts to a desertified state induced by anthropogenic factors or climate fluctuations (Fig. 2). Soil erosion affects the productivity and spatial pattern of dryland vegetation

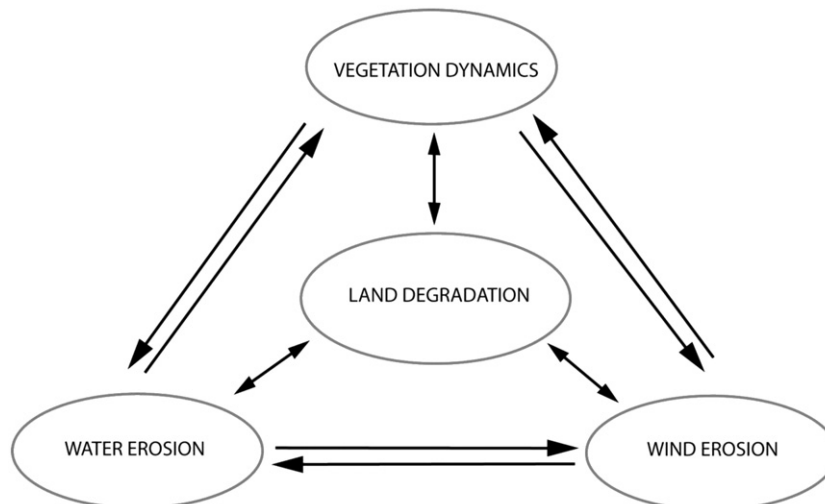


Fig. 1. A holistic perspective of land degradation dynamics by considering the interactions among wind–water erosion and vegetation dynamic processes.

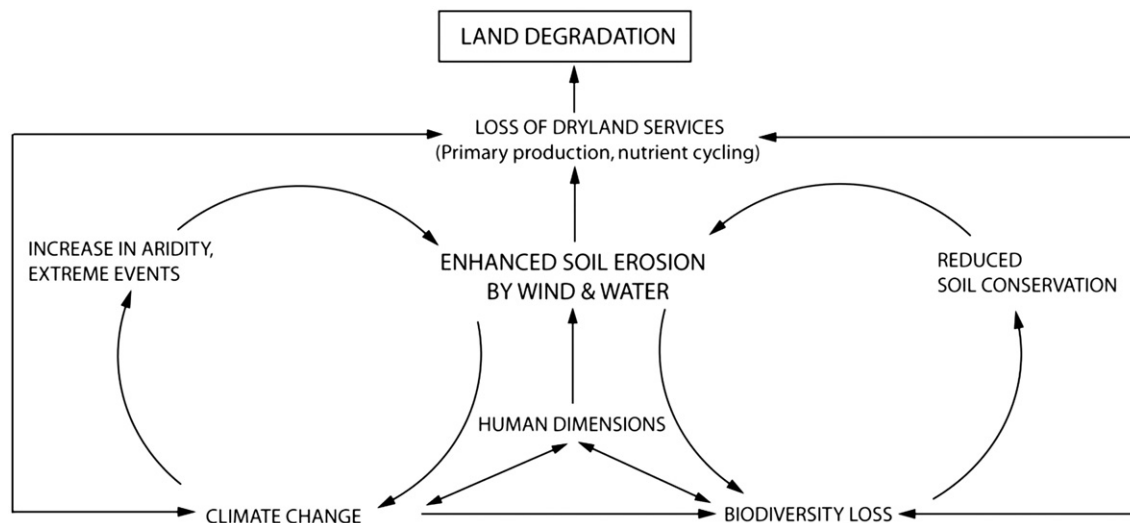


Fig. 2. Conceptual diagram showing the interrelations among soil erosion, land degradation, climate change and biodiversity loss (modified from MEA, 2005).

and soil resources (Schlesinger et al., 1990; Puigdefabregas, 2005), and is recognized as a threat to sustainable agricultural production in arid and semi-arid landscapes (Lal, 2001). In agricultural lands (including crop and rangelands), accelerated soil erosion is responsible for the loss of fertile topsoil, depletion of soil fertility and subsequent decrease in crop productivity (Lal, 2001, 2003; Li et al., 2008). The detachment, transport and deposition of fine soil, which holds most of the nutrients, all affect the soil organic carbon pool and can influence the global carbon budget (Lal, 2003; Li et al., 2008). Therefore, erosion results in loss of soil nutrients and the capacity of soil to hold water, and in an overall decrease in the ability of land to sustain vegetation. In addition, these erosion processes result in the movement of sediments and agricultural pollutants into water bodies, thereby affecting scarce fresh water resources in these dryland landscapes (Lal, 2001). Dust emissions resulting from wind erosion contribute to dust aerosols that are transported long distances and deposited over continents and oceans (Duce and Tindale, 1991; Swap et al., 1996). Desert dust is a major contributor of tropospheric aerosols, which affect global climate, air quality and hydrological–biogeochemical cycles (Ramanathan et al., 2001; Hui et al., 2008; Field et al., in press). Consequently, the impacts of soil erosion processes can extend beyond the geographic boundaries of dryland regions.

The susceptibility of soils to erosion depends on several interrelated factors such as climate, moisture availability, soil properties, topography, land cover and management (Lal, 1994). Among biophysical controls, two factors are perhaps most important in dryland ecosystems: soil moisture and vegetation cover. First, soil moisture, even in arid soils, can strongly affect soil susceptibility to erosion by wind through its effect on the bonding forces between soil particles (Ravi et al., 2004, 2006). Soil moisture dynamics are in turn affected by soil physical and biological properties. Second, vegetation shelters the soil from erosive action of wind and water (Raupach, 1992). Cover by physical or biological soil surface crusts also affects erosion by wind and water (Belnap and Gillette, 1998; Singer and Shainberg, 2004). In addition, anthropogenic factors such as agriculture, grazing, soil management, fire, and urbanization further affect soil erosion (Doerr et al., 2000; Neff et al., 2005; Ravi et al., 2009b).

3. Interactions among hydrologic–aeolian erosion and vegetation dynamic processes

3.1. Interactions between hydrological and aeolian transport processes

The physical process of soil erosion by wind and water involves three stages: detachment, transport and deposition. Significant

differences exist between soil erosion process by wind and water (Toy et al., 2002; Lal, 1994). Wind erosion is generally thought to be more dominant in more arid environments, particularly where intense winds occur in a dry season, whereas water erosion is thought to be dominant in more humid environments (Marshall, 1973; Kirkby, 1978; Bullard and Livingstone, 2002; Field et al., 2009; Fig. 3). Water erosion is directly related to the hydrological cycle, in that processes such as rainsplash detachment, rill, and gully erosion require rainfall and runoff. For either type of erosion, the detachment process begins with breaking of soil aggregates, which can be due to raindrop impacts for water erosion or saltation bombardment for wind erosion. High intensity rainfall can dislodge soil particles directly if the impacting raindrops have sufficient kinetic energy. Rainfall can also indirectly detach and transport soil by generating runoff in rills and gullies (Lal, 1994). Notably, water erosion can be spatially restricted in contrast to wind erosion, which can transport large quantity of sediments over spatially extensive areas (Bullard and Livingstone, 2002; Field et al., 2009). Further, fluvial transport represents an event based, unidirectional and somewhat irreversible transport of sediments depending on slope of the landscape (Dingman, 1994; Field et al., 2009). In contrast, wind erosion can be more frequent compared to water erosion, in that it can happen in response to less frequent high velocity winds and to more frequent short wind gusts (Stout and Zobeck, 1997; Field et al., 2009). Wind erosion processes can be two-dimensional, with vertical and horizontal transport components, the latter of which are less constrained by landscape topography than in the case of water erosion (Breshears et al., 2003). Sediment transport by wind is not much affected by the slope and is somewhat reversible, because a possibility exists that the eroded sediment may be redeposited back to the place of origin (Field et al., 2009). The nature of sediment supply (size, amount, sorting, and availability) has significant impact on rates of aeolian erosion (Chepil, 1945). For example, aeolian erosion is more effective when sediments have been pre-sorted. In addition, a sediment-laden air requires a lower wind speed to entrain soil particles compared to a non-sediment carrying wind (Chepil, 1945).

Although wind and water erosion and associated transport processes operate at different spatial scales, they nonetheless both impact the stability of the soil surface at any given location and it is, therefore, important to investigate these impacts collectively rather than separately (Breshears et al., 2003; Visser et al., 2004; Field et al., 2009). One approach for comparing rates of wind and water driven transport is to quantify the amount of transported material of each that crosses per unit length of a line that is oriented perpendicular to

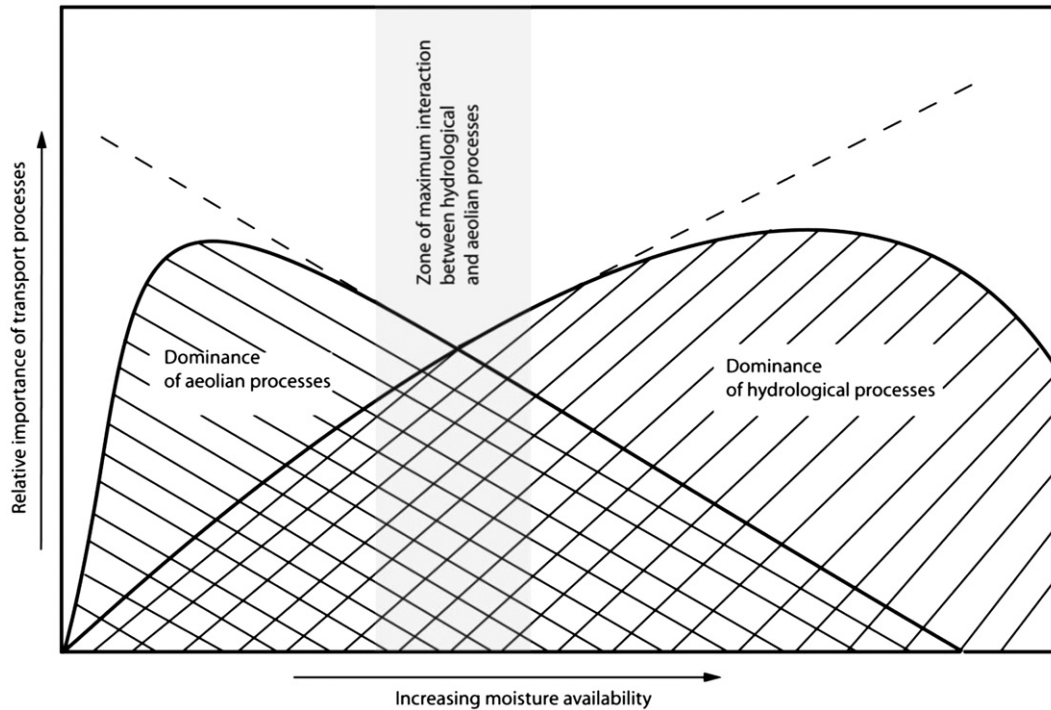


Fig. 3. Illustration of the process intensity domain concept (redrawn from Kirkby, 1978) applied for hydrological and aeolian transport processes in drylands, showing the region of maximum interaction between these processes.

the erosion force (Breshears et al., 2003). For water erosion, the erosional force is parallel to the slope. In contrast, for wind erosion, the erosional force can be omni-directional and is parallel to the wind direction. Consequently, wind transported material can move in one direction at one time and then subsequently move back in the opposite direction; the degree to which this occurs depends on the degree to which the prevailing wind direction dominates other wind directions. Some initial estimates of wind and water erosion, based on time-series measures of wind transported material and extrapolations from rainfall simulation of water transported material, indicate that wind transport can often dominate water transport in arid and semi-arid landscapes (Breshears et al., 2003). Because wind transported material is a small fraction of net erosion loss at larger scales, wind and water erosion in different semi-arid ecosystems can both be large enough that neither is negligible enough to ignore (Breshears et al., 2003). Needed are studies that simultaneously quantify co-located rates of wind and water driven transport (Field et al., 2009).

As noted, the relative importance of wind and water erosion depends on topography, land cover conditions, and climate, with the dominance of aeolian and hydrological processes being typical of the drier and the wetter climates, respectively (Fig. 3). The presence of aeolian and fluvial landforms may provide useful indications on the alternations of relatively arid or relatively humid phases in the history of regions of Earth. In some cases interactions between aeolian and hydrologic processes may be separated in time, with aeolian processes being affected by the legacy of hydrologic processes from previous wetter periods and vice versa. This separation can involve geological times. For example, the major sources of atmospheric dusts on Earth are from sediment deposits that accumulated in the lake beds that were flooded in the Pleistocene (Goudie and Middleton, 2006). Indeed the drying of lake beds is activating sediment removal by aeolian processes even in modern times as in the case of Lake Chad or the Aral Sea (e.g., Goudie and Middleton, 2006). Similarly, during dry spells, sand dune may encroach into the bed of ephemeral rivers, resulting in a complex interaction between aeolian and fluvial landforms (Nanson et al., 2002). In some cases the topography of aeolian landforms sets the stage for hydrological processes taking place during wet periods.

For example, the coppice dunes in the Chihuahuan desert (New Mexico, USA), discussed in the following sections, are generated by aeolian processes and provide a relief that determines the spatial patterns for surface runoff. Similarly, the sand dunes in the Badain Jaran “sand sea”, some of the biggest dunes on Earth, exhibit ephemeral lakes confined in the interdune areas. The existence on the dune slopes of palaeo-shorelines at higher elevations than the current ones provides interesting information on the climatic history of the region and the existence of a wetter epoch in the early Holocene (e.g., Yang et al., 2004).

In many systems hydrologic and aeolian processes do not operate independently and the interactions between them are known to have implications on landscape evolution, vegetation pattern formation, and land degradation (e.g., Fearnough et al., 1998; Ravi et al., 2007). Interactions between aeolian and hydrological processes can occur at all moisture levels, but they are particularly significant in areas where neither wind nor water erosion dominate, as in the case of semi-arid regions (Fig.3) (Kirkby, 1978; Bullard and Livingstone, 2002).

3.2. Soil erosion and vegetation patterns

Wind and water erosion result in selective removal and redistribution of nutrient rich fine soil particles. Consequently, through the impact on the soil properties and, in turn, on soil moisture dynamics, hydrological and aeolian transport processes determine the conditions affecting the establishment and survival of different functional types of vegetation, which collectively affect structure and function of water-limited ecosystems (Bhark and Small, 2003; Puigdefabregas, 2005). Interactions between soil erosion processes and vegetation are thought to be the major factors contributing to the formation of vegetation patterns typically observed in resource-limited arid and semi-arid landscapes, as reflected in spatial patterns that can include bands, stripes, spots and rings (e.g., Lefever and Lejeune, 1997; Ravi et al., 2008; Borgogno et al., 2009). The spatial distribution, geometry and scale of these vegetated patches affect the spatial patterns of soil moisture, sediments and nutrients, which in turn determine plant growth and species composition in arid and semi-arid landscapes

(Puigdefabregas, 2005). Positive feedbacks between vegetation and surface soil moisture (D'Odorico et al., 2007) may further enhance heterogeneities in vegetation and soil distribution, contributing to the emergence of alternative stable states in dryland vegetation (Rietkerk and van de Koppel, 1997) and vegetation pattern formation (e.g., Lefever and Lejeune, 1997; van de Koppel et al., 2002; Borgogno et al., 2009). The wetter surface soils typically found under plant canopies may facilitate vegetation establishment, whereas drier soils in the barren interspaces may prevent vegetation growth. Therefore, vegetation patterns can be considered as biological indicators of abiotic processes (e.g., runoff and infiltration), and of source-sink areas for sediments in dryland landscapes (Puigdefabregas, 2005; Ludwig et al., 2005). Changes in vegetation patterns, induced by changes in sediment transport processes, are viewed as readily recognizable signs of environmental change (e.g., desertification) in degraded arid landscapes (van de Koppel et al., 2002). In short, information on the interactions between soil erosion processes and vegetation patterns can improve our current understanding of important processes underlying the dynamics of arid and semi-arid ecosystems and the response of these systems to climate change and management scenarios.

3.2.1. Grass growth patterns

Several theories have been proposed to explain the formation and growth of grass ring patterns such as external disturbances like fires, negative soil–plant feedbacks and changes in plant growth architecture (Danin and Orshan, 1995; Lewis et al., 2001; Bonanomi et al., 2005). The role of soil erosion processes in modifying grass growth patterns have been documented in many arid and semi-arid landscapes. Studies have shown that negative soil–plant feedbacks inducing the development of grass ring patterns can result from the interactions between hydrological–aeolian processes and vegetation dynamic processes (Ravi et al., 2008). Grass mortality and growth patterns induced by aeolian deposits have been reported as early as in the 1930s. In the shortgrass ecosystems in the Great Plains in the United States soil deposits from dust storms caused mortality of blue grama grass (Weaver and Albertson, 1936). The growth patterns of the grasses in this region appear to be strongly affected by the deposition of aeolian sediments (Robertson, 1939). Such deposited sediments can result in a negative soil–plant feedback at the center of the bunch grasses, leading to die-back of the buried grass and consequent modification of bunch grass patterns. These processes are exemplified by the formation of ring-shaped patterns in blue grama grasses in the Chihuahuan desert (Fig. 4 a). Field experiments have shown that the development of ring patterns in blue grama grass are best explained by the deposition of wind borne fine soil particles onto the bunch grass, which results in considerable changes in soil texture at the center of the grass clump (Ravi et al. 2008) (Fig. 4 b). Runoff contributes to redistribution of water and nutrients from the central areas of the ring—which have low rates of infiltration and high nutrient content—to the outer edges (Fig. 4 c). This explains the preferential establishment of new vegetative grass growth at the outer edges of the ring and the mortality of grass in the central areas of the ring. The same mechanism determines the growth of the ring over time. As the ring grows in size, it becomes less efficient at trapping windblown sediments (Ravi et al., 2008). At a certain point the grass ring breaks down into pieces, which become “separate” bunch grasses, thereby contributing to the vegetative regeneration of the community (Lewis et al., 2001). Each of these bunch grasses will then undergo a similar process of lateral growth, ring formation, and ring break-up.

3.2.2. Dynamics of shrub coppice dune

The dynamics of the formation and development of shrub coppice dunes in the Chihuahuan desert are an example of the interactions between wind/water erosion and shrubs (Fig. 4 d). Soil erosion

processes, in particular by wind, are responsible for the removal of nutrient-rich fine soil particles from the intercanopy areas and the deposition onto shrub patches (Ravi et al., 2007; Li et al., 2008). The deposition of fine soil particles causes considerable changes in the soil texture in the vegetated patch and results in a heterogeneous spatial distribution of soil infiltration capacity, runoff, and rates of erosion (Puigdefabregas, 2005; Ravi et al., 2007). Shrubs contribute to formation and augmentation of coppice dunes through the differential trapping of fine wind-borne sediments among areas located at the center, upwind and downwind of the shrub (Fig. 4 e). Hydrological processes do not contribute to the accumulation of sediments in the soil beneath the shrub canopies because the microtopography diverts water flow away from the higher elevation microsites existing in shrub-dominated soil patches. Therefore, with the exception of rain splash, no hydrologic mechanisms are able to contribute directly to the formation and development of soil mounds and coppice dunes beneath shrubs through deposition. However, runoff contributes to the redistribution of water and nutrient-rich sediments from the middle to the edges of the dune, and to the consequent preferential establishment/growth of mesquite shrubs at the edges. The differential rates of soil deposition and removal by aeolian processes result in differential rates of hydrological processes such as infiltration and runoff, thereby affecting the formation and expansion of these coppice dunes (Fig. 4 f). A similar growth pattern in dune vegetation was observed in stabilized desert dunes of Northern China, where deposition of finer soils resulted in considerable changes in soil texture (e.g., Fearnough et al., 1998). The retention of moisture at the surface by the finer soils deposited at the center of the mounds combined with the formation of physical and biological soil crusts resulted in the decline of shrub vegetation at the center of the dunes and the formation of a ring-shaped pattern of shrubs at the base of the dune (Ravi et al., 2007). The importance of interactions between vegetation and aeolian sediments in coppice dune formation is further supported by research from arid landscapes around the world, including the Arabic Desert (Khalaf et al., 1995), Southern Africa (Dougill and Thomas, 2002), and West Africa (Nickling and Wolfe, 1994).

3.2.3. Erosion processes and rapid shifts in vegetation

The most common form of land degradation in many dryland systems around the world is associated with the relatively rapid change in the composition of the plant community, with a shift between grasses and woody plants. For example, woody plants have been observed to encroach into arid grasslands (Archer, 1989; Van Auken, 2000), while exotic grasses have been found to invade stable shrublands (e.g., D'Antonio and Vitousek, 1992). These relatively rapid (<100 years) changes in vegetation can result from the complex interaction among several factors including climate change, increase in CO₂ concentration and anthropogenic disturbances (Archer, 1989; Schlesinger et al., 1990; van Auken, 2000). The interactions between hydrological–aeolian processes and vegetation play a major role in these rapid vegetation changes with important implications for land degradation. On the other hand, shifts in the composition of the plant community are typically associated with changes in surface roughness and vegetation cover conditions, which, in turn, affect the rates and patterns of soil erosion and sediment transport in these landscapes. In these systems, changes in major ecosystem functions can be explained and predicted in terms of changes in the spatial and temporal distribution of soil resources (e.g., nutrients, soil moisture), which are controlled by hydrological and aeolian transport processes. As noted in the previous section these soil erosion processes interact with one another and also with the dynamics of vegetation. Further, these processes are affected by disturbances (e.g., fires, grazing) and management practices.

Even though wind erosion is considered to be the dominant erosion process in many drylands, most studies of wind erosion focus on differences associated with soil texture, in part due to a focus on agricultural fields and dunes. However, recent studies from a wide

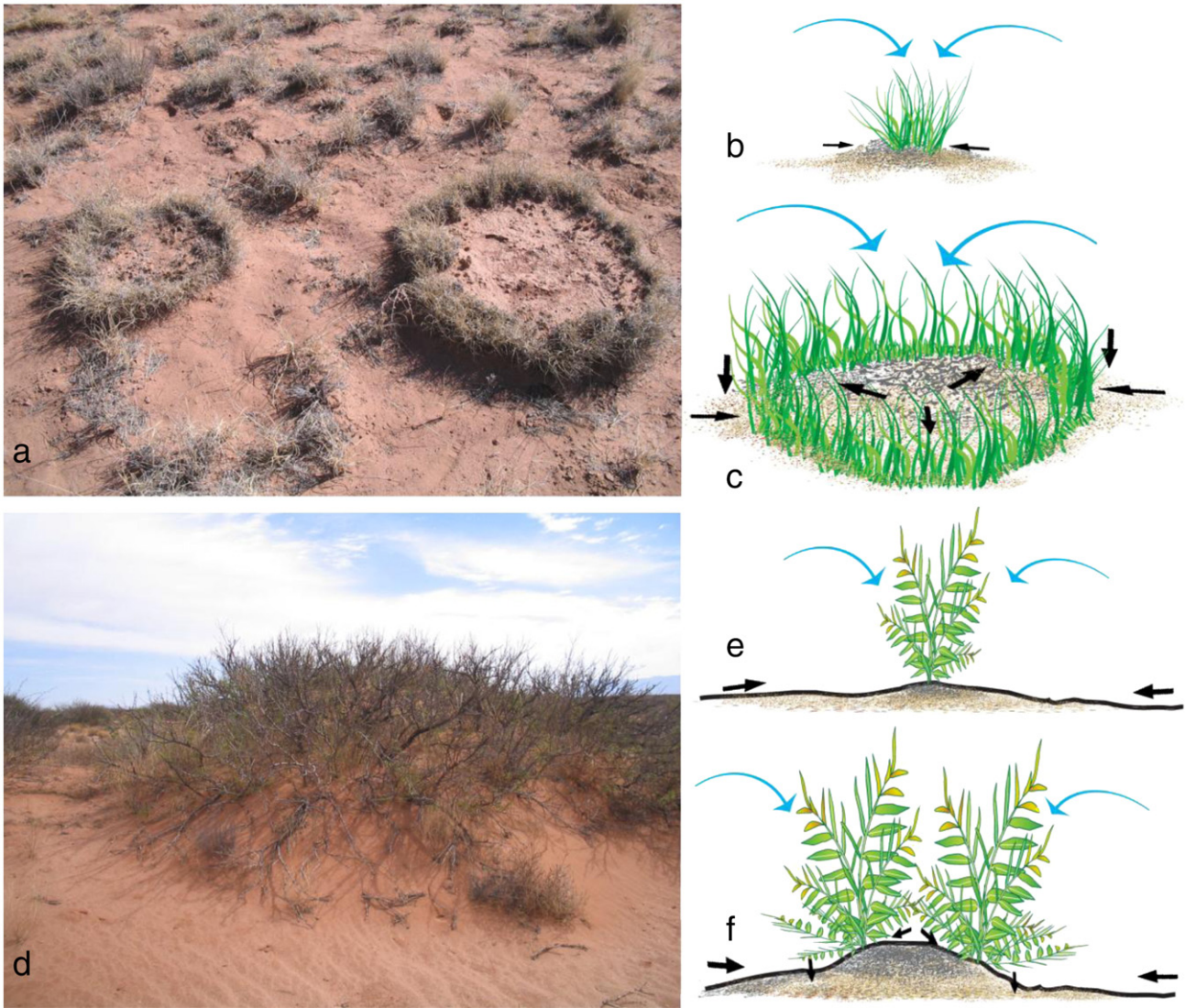


Fig. 4. (a) Ring growth pattern of *Bouteloua gracilis* and (d) a mesquite coppice dune in Chihuahuan desert, New Mexico, USA. (b,c e,f) Conceptual diagrams showing the interaction of hydrological and aeolian processes resulting in the formation and expansion of (b–c) grass rings and (e–f) coppice dunes. The straight black arrows indicate hydrological processes (infiltration and runoff) and curved blue arrows indicate aeolian processes (deposition and erosion) (modified from Ravi et al., 2007 and Ravi et al., 2008).

range of dryland ecosystem types that include grasslands, shrublands, woodlands, and forests, are providing new insights into erosional trends in drylands. More specifically, when drylands are considered in a broader context, the amount of wind erosion appears to be related to the amount of cover of woody plants (trees or shrubs) and the degree to which disturbance has reduced the amount of associated ground cover, particularly in intercanopy locations between woody plants (Fig. 5) (Breshears et al., 2009). New predictive equations of wind-driven transport that account for plant cover and gap sizes provide additional support for this perspective (Okin, 2008; Field et al., in press). This is in part because the amount of woody plant cover fundamentally influences air flow patterns. For solid objects, 0–14% cover produces isolated wake flow, 14–40% cover produces wake interference flow, and >40% cover produces skimming flow. Importantly, shrublands are associated with the range of 14–40% cover, and, therefore, are likely to produce wake interference flow, thereby maximizing potential for wind erosion (Wolfe and Nickling, 1993). In addition, the minimum amount of ground cover tends to increase with the amount of woody plant canopy cover because shrubs and trees usually have litter beneath them. Consequently, for relatively undisturbed ecosystems, shrublands have inherently greater aeolian sediment transport because of wake interference flow associated with

intermediate levels of density and spacing of woody plants (Fig. 5). For disturbed ecosystems, the upper bound for aeolian sediment transport decreases as a function of increasing amounts of woody plant cover because of the effects of the height and density of the canopy on airflow patterns and ground cover associated with woody plant cover. The net result of these apparent trends is that aeolian sediment transport following disturbance spans the largest range of rates in grasslands and associated systems with no woody plants (e.g., agricultural fields), an intermediate range in shrublands, and a relatively small range in woodlands and forests (Breshears et al., 2009). Consequently, management of wind erosion should account for combinations of woody and herbaceous ground cover, as increasing only woody plant cover may not effectively reduce wind erosion (Fig. 5).

In the case where shrub encroachment has developed in landscapes historically dominated by grasses (Fig. 6), the grass cover is typically replaced by scattered shrubs, with a substantial increase in bare soil fractions and the subsequent enhancement in the rates of soil erosion in barren soil patches (Schlesinger et al., 1990; Huenneke et al., 2002; Baez et al., 2006). A portion of the soil removed by erosion processes is trapped by the shrub canopies and deposited in shrub dominated soil patches (Charley and West, 1975; Schlesinger et al., 1990). This process leads to the concentration of soil resources (e.g.,

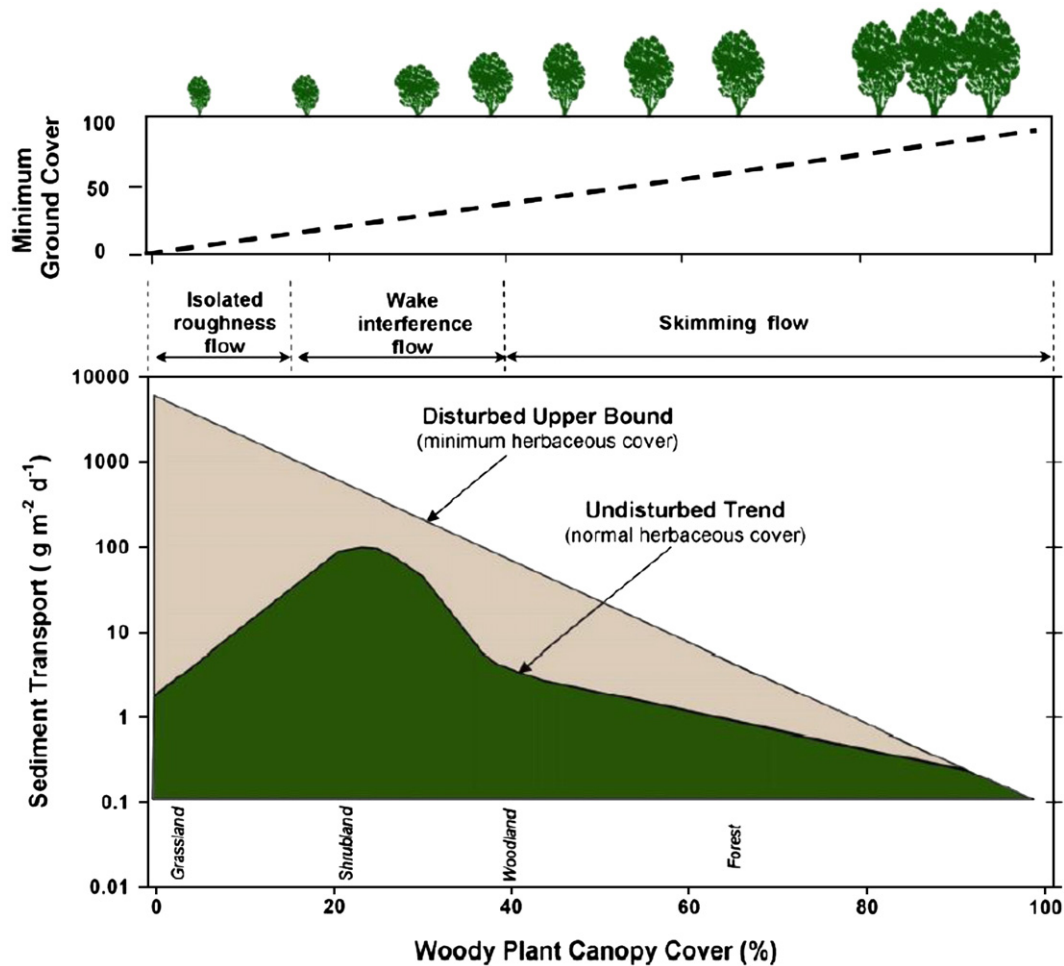


Fig. 5. Conceptual framework for aeolian sediment transport as sediment transport along the grassland–forest continuum for undisturbed and disturbed sites. Associated gradients are shown for minimum amount of ground cover (assumed here to equal woody plant cover) and main types of flow. Relatively undisturbed sites occur near the undisturbed trend line. Disturbances can increase sediment transport toward the indicated upper bound. Among relatively undisturbed ecosystems, shrublands inherently have the largest sediment transport. The range of values between the undisturbed trend line and the disturbed upper bound is greatest for grasslands. Variation in sediment transport between the undisturbed trend line and the disturbed upper bound is associated with changes in amount of ground cover in intercanopy areas, particularly from herbaceous plants but also including biological and non-biological soil crusts. Variation in soil texture may also influence variation within this interval. Unusually large amounts of intercanopy ground cover can yield sediment transport that are less than corresponding values on the undisturbed trend line (modified from [Breshears et al., 2009](#)).

nutrients absorbed onto fine soil particles) in the soil beneath shrub canopies and to the emergence of a heterogeneous distribution of soil properties with a mosaic of nutrient-rich soil patches (or “fertility islands”) supporting shrub vegetation bordered by nutrient-depleted bare soil. The erosion of bare soil patches and the deposition of aeolian sediments in the fertility islands lead to the formation of a heterogeneous microtopography, which mirrors the heterogeneities in the spatial distribution of soil resources and vegetation. Soil erosion processes by wind and water are responsible for creating these heterogeneities. Moreover, in shrub encroached grasslands the enhancement of aeolian erosion leads to an increase in the rate of dust production. The resulting increase in the levels of tropospheric dust aerosols has an important impact on human health and regional climate ([Ramanathan et al., 2001](#); [Hui et al., 2008](#)).

The relatively rapid and abrupt character of the process of shrub encroachment suggests that mechanisms of positive feedback might work to sustain this shift in plant community composition. A set of possible mechanisms has been invoked to explain the persistent and catastrophic character of land degradation dynamics as the result of a self-sustaining feedback loop ([Anderies et al., 2002](#); [Schlesinger et al., 1990](#); [Okin et al., 2009a](#)). For example, a self-sustained cycle of soil erosion, depletion of soil resources, and vegetation loss may contribute to the irreversible denudation of grass-dominated areas ([Archer et al.,](#)

[1995](#); [Schlesinger et al., 1990](#); [Okin et al., 2009a](#)). On the other hand, as noted, the encroachment of shrubs is favored by the deposition of nutrient-rich sediments transported by wind and water, and the consequent formation of fertile shrub patches ([Fig. 6](#)). At the same time, the loss in grass fuel—for example because of overgrazing—decreases the pressure of fires on shrub vegetation thereby further enhancing woody plant encroachment ([Anderies et al., 2002](#); [D’Odorico et al., 2006](#)). As a result, the ability of the landscape to produce grass fuel is further reduced. The feedback between soil erosion and vegetation dynamics, and the one between fire regime and vegetation composition involve some interesting interactions with wind and water erosion either directly or indirectly. In the case of fires, it has long been reported that fires reduce infiltration, increase runoff, and enhance water erosion as an effect of fire-induced water repellency ([DeBano, 1966, 2000](#); [Doerr et al., 2000](#)). More recently, it has been reported that fires also enhance the soil susceptibility to wind erosion ([Ravi et al., 2009b](#)) in arid grasslands and shrublands ([Ravi et al., 2006](#); [Whicker et al., 2002](#)). The post-fire increase in soil erodibility in areas affected by the burning of shrub biomass may favor the local redistribution of nutrient-rich soil from the fertility islands to the nutrient-poor interspaces, thereby reducing the heterogeneity of the landscape ([Ravi et al., 2009a](#)). Therefore, the interactions between erosion processes and fires may revert or limit the process of shrub encroachment at its early stages

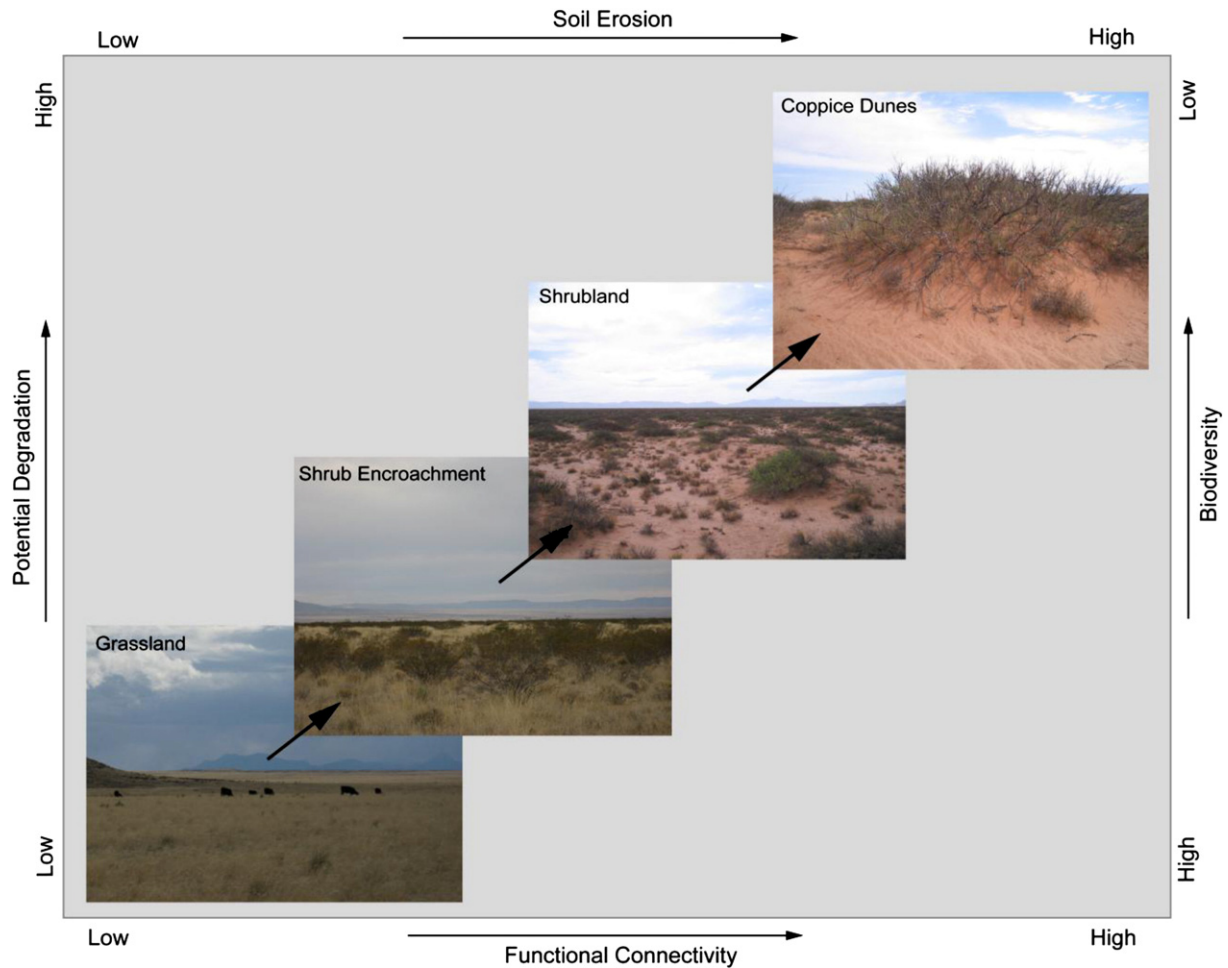


Fig. 6. Conceptual diagram showing the stages of grassland degradation in the Chihuahuan Desert along with changes in functional connectivity, soil erosion rates and biodiversity.

(Fig. 6), i.e., when a sufficiently continuous grass cover exists to carry the fire across the landscape. Once the grass cover is lost, however, fires decrease in frequency and intensity, and the landscape evolves towards a heterogeneous distribution of vegetation and soil resources with the formation of shrub-dominated fertility islands by wind and water (Fig. 6). In areas with sandy soils these erosion/deposition processes may lead to the formation of coppice dunes (Ravi et al., 2007) state (Fig. 4 d–f).

In many arid shrubland systems, invasion by exotic annual grasses is a major environmental issue leading to land degradation. These grasses are found to increase the connectivity between shrub patches, thereby triggering periodic fires (D'Antonio and Vitousek, 1992). Studies in the shrub-native grass ecotone in the Northern Chihuahuan desert have shown that fires can counteract the formation of heterogeneity, in that the interaction of fires with soil erosion favors a more homogeneous distribution of soil resources (Ravi et al., 2009a). Similarly, observations in Sonoran desert shrublands have shown that invasive annual grasses can enhance fire frequency, increase shrub mortality, and modify the rates of soil erosion thereby leading to a more uniform redistribution of soil resources (Ravi et al., 2009c). The increase in fire frequency in these shrub-dominated ecosystems that were not fire adapted may lead to the conversion of these landscapes into exotic (annual) grasslands. In the long term, however, the invasive grass cover is not sustainable, in that annuals do not survive recurrent droughts (Ravi et al., 2009c). Drought-induced loss of these invasive annual grasses is expected to be followed by higher rates of erosion and irreversible losses of soil resources from these landscapes. Thus, in these systems annual grass invasions can modify the rates and patterns of soil erosion processes with implications on land degradation.

A new framework explains the major forms of desertification in dryland ecosystems on the basis of landscape connectivity (Okin et al. 2009b). The degradation effects of erosion processes following vegetation shifts can be summarized in terms of changes in the length of connected pathways (interconnected bare soil patches) or “functional connectivity” in the landscape. These pathways existing in the landscape determine the rate of erosion processes and, hence, the dynamics of land degradation. Even though short pathways are known to be beneficial to the landscape, such as in increasing the availability of moisture and nutrients to vegetated patches (Ludwig et al 2005), once the path lengths exceed a certain threshold (e.g., induced by droughts, grazing, fires), soil erosion processes can lead to irreversible loss of resources (Turnbull et al., 2008; Ravi et al 2009c; Okin et al, 2009b). In contrast, in the case of invasion of shrublands by exotic grasses, the connectivity for wind and water erosion decreases. The grasses provide enough connectivity between the shrub patches to establish a fire-cycle in the system. As mentioned earlier, drought-induced loss of invasive grasslands may be followed by a dramatic increase in functional connectivity and consequently higher rates of erosion and losses of soil resources (Ravi et al., 2009c). This highlights the importance of understanding the role of connectivity, both structural and functional, in the landscape for the sustainable management of drylands under changing climate and natural and anthropogenic disturbances.

4. Conclusion

Drylands around the world are undergoing rapid land degradation and shifts in vegetation composition in response to climate change and anthropogenic disturbances. These changes in land cover and soil

properties affect the productivity of the landscape, with important environmental, socioeconomic, and political implications. Accelerated hydrological–aeolian erosion processes and rapid vegetation shifts are recognized as important drivers of land degradation. This review highlights some of the recent advances in the study of the interactions among these processes in dryland ecosystems and provides a synthetic perspective on the role of these interactions in the dynamics of land degradation (Fig. 1).

Global climate models have predicted the aridification of several dryland regions around the world. We argue that this increase in aridity could enhance the dominance of abiotic controls of land degradation, including the susceptibility of the landscape to wind and water erosion. Climatic changes and disturbances can lead to rapid large-scale vegetation changes in dryland systems, which can alter the rates and patterns of erosion by wind and water and the interactions between these processes. Human activities have profound influence on the hydrologic–aeolian processes contributing to land degradation, and the increase in erosion processes as a result of anthropogenic disturbances and management practices has dramatically increased over the past century. The altered soil erosion processes affect the emerging vegetation dynamics and in turn the fluxes of water, carbon and nutrients from these systems. Therefore, we suggest that a more integrated perspective of the interactions among hydrologic–aeolian and vegetation dynamic processes is fundamental to investigate the role played by climate change, disturbance regimes and management scenarios in the process of land degradation. Such a more holistic perspective can further aid land management by collaborative research among physical, biological and social scientists focused on the complex role of biophysical processes and anthropogenic factors affecting land degradation.

Acknowledgements

We acknowledge current and previous sponsors of this research: The Phileology Foundation of Fort Worth Texas (for UA Biosphere2), National Science Foundation (grants # DEB 0717360, EAR 0746228 and DEB-0620482 (to the University of New Mexico for Long-term Ecological Research), and DEB-0816162 (DDB).

References

- Anderies, J.M., Janssen, M.A., Walker, B.H., 2002. Grazing management, resilience, and the dynamics of a fire-driven rangeland system. *Ecosystems* 5 (1), 23–44.
- Archer, S., 1989. Have southern Texas savannas been converted to woodlands in recent history. *American Naturalist* 134 (4), 545–561.
- Archer, S., Schimel, D.S., Holland, E.A., 1995. Mechanisms of shrubland expansion—land-use, climate or CO₂. *Climatic Change* 29 (1), 91–99.
- Baez, S., Collins, S.L., Lightfoot, D., Koontz, T., 2006. Effects of rodent removal on community dynamics in desert grassland and shrubland vegetation. *Ecology* 87, 2746–2754.
- Belnap, J., Gillette, D.A., 1998. Vulnerability of desert biological soil crusts to wind erosion: the influences of crust development, soil texture, and disturbance. *Journal of Arid Environments* 39, 133–142.
- Bhark, E.W., Small, E.E., 2003. Association between plant canopies and the spatial patterns of infiltration in shrubland and grassland of the Chihuahuan Desert, New Mexico. *Ecosystems* 6 (2), 185–196.
- Bonanomi, G., Giannino, F., Mazzoleni, S., 2005. Negative plant–soil feedback and species coexistence. *Oikos* 111 (2), 311–321.
- Borgogno, F., D'Odorico, P., Laio, F., Ridolfi, L., 2009. Mathematical models of vegetation pattern formation in ecohydrology. *Reviews of Geophysics* 47, RG1005. doi:10.1029/2007RG000256.
- Breshears, D.D., Whicker, J.J., Johansen, M.P., Pinder, J.E., 2003. Wind and water erosion and transport in semi-arid shrubland, grassland and forest ecosystems: quantifying dominance of horizontal wind-driven transport. *Earth Surface Processes and Landforms* 28 (11), 1189–1209.
- Breshears, D.D., Whicker, J.J., Zou, C.B., Field, J.P., Allen, C.D., 2009. A conceptual framework for dryland aeolian sediment transport along the grass land-forest continuum: effects of woody plant canopy cover and disturbance. *Geomorphology* 105, 28–38.
- Brovkin, V., Claussen, M., Petoukhov, V., Ganopolski, A., 1998. On the stability of the atmosphere–vegetation system in the Sahara/Sahel region. *Journal of Geophysical Research* 103 (D24), 31,613–31,624.
- Bullard, J.E., Livingstone, I., 2002. Interactions between aeolian and fluvial systems in dryland environments. *Area* 34 (1), 8–16.
- Burke, E.J., Brown, S.J., Christidis, N., 2006. Modeling the recent evolution of global drought and projections for the twenty-first century with the Hadley Centre climate model. *Journal of Hydrometeorology* 7 (5), 1113–1125.
- Chapin III, F.S., Walker, B.H., Hobbs, R.J., Hooper, D.U., Lawton, J.H., Sala, O.E., Tilman, D., 1997. Biotic controls of the functioning of ecosystems. *Science* 277, 500–504.
- Charley, J.L., West, N.E., 1975. Plant-induced soil chemical patterns in some shrub-dominated semi-desert ecosystems of Utah. *Journal of Ecology* 63 (3), 945–963.
- Charney, J.G., 1975. Dynamics of deserts and drought in Sahel. *Quarterly Journal of the Royal Meteorological Society* 101 (428), 193–202.
- Chepil, W.S., 1945. Dynamics of wind erosion: 1. Nature of movement of soil by wind. *Soil Science* 60, 305–335.
- Danin, A., Orshan, G., 1995. Circular arrangement of *Stipagrostis ciliata* clumps in the Negev, Israel and near Gokaeb, Namibia. *Journal of Arid Environments* 30 (3), 307–313.
- D'Antonio, C.M., Vitousek, P.M., 1992. Biological invasion by exotic grasses, the grass–fire cycle, and global change. *Annual Review of Ecology and Systematics* 23, 63–87.
- Darkoh, M.B.K., 1998. The nature, causes and consequences of desertification in the drylands of Africa. *Land Degradation and Development* 9, 1–20.
- DeBano, L.F., 1966. Formation of non-wettable soils involves heat transfer mechanism. USDA Forest Service Research Note PSW-132. 8 pp.
- DeBano, L.F., 2000. The role of fire and soil heating on water repellency in wildland environments: a review. *Journal of Hydrology* 231, 195–206.
- Dingman, S.L., 1994. *Physical hydrology*. Prentice-Hall, Upper Saddle River.
- D'Odorico, P., Laio, F., Ridolfi, L., 2006. A probabilistic analysis of fire-induced tree–grass coexistence in savannas. *American Naturalist* 167 (3), E79–E87.
- D'Odorico, P., Caylor, K., Okin, G.S., Scanlon, T.M., 2007. On soil moisture–vegetation feedbacks and their possible effects on the dynamics of dryland ecosystems. *Journal of Geophysical Research* 112, G04010.
- Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 2000. Soil water repellency: its causes, characteristics and hydro-geomorphological significance. *Earth-Science Reviews* 51 (1–4), 33–65.
- Dougill, A.J., Thomas, A.D., 2002. Nebkha dunes in the Molopo Basin, South Africa and Botswana: formation, controls and their validity as indicators of soil degradation. *Journal of Arid Environments* 50, 413–428.
- Dregne, H.E., Chou, N.T., 1992. Global desertification dimensions and costs, in degradation and restoration of arid lands. Texas Tech University, Lubbock.
- Duce, R.A., Tindale, N.W., 1991. Atmospheric transport of iron and its deposition in the ocean. *Limnology and Oceanography* 36, 1715–1726.
- Elmqvist, T., Folke, C., Nystrom, M., Peterson, G., Bengtsson, L., Walker, B., Norberg, J., 2003. Response diversity and ecosystem resilience. *Frontiers in Ecology and the Environment* 1 (9), 488–494.
- Fearnough, W., Fullen, M.A., Mitchell, D.J., Trueman, I.C., Zhang, J., 1998. Aeolian deposition and its effect on soil and vegetation changes on stabilized desert dunes in northern China. *Geomorphology* 23, 171–182.
- Field, J.P., Belnap, J., Breshears, D.D., Neff, J.C., Okin, G.S., Whicker, J.J., Painter, T.H., Ravi, S., Reheis, M.C., Reynolds, R.R., in press. The ecology of dust. *Frontiers in Ecology and Environment*. doi:10.1890/090050.
- Field, J.P., Breshears, D.D., Whicker, J.J., 2009. Toward a more holistic perspective of soil erosion: why aeolian research needs to explicitly consider fluvial processes and interactions. *Aeolian Research* 1 (1–2), 9–17.
- Geist, H.J., Lambin, E.F., 2004. Dynamic causal patterns of desertification. *Bioscience* 54 (9), 817–829.
- Goudie, A., Middleton, N., 2006. *Desert dust in the global system*. Springer, Heidelberg.
- Held, I.M., Delworth, T.M., Lu, J., Findell, K.L., Knutson, T.R., 2005. Simulation of Sahel Drought in the 20th and 21st centuries. *Proceedings of the National Academy of Sciences of the United States of America* 102 (50), 17,891–17,896.
- Hui, J.W.C., Cook, B.I., Ravi, S., D'Odorico, P., Fuentes, J.D., 2008. Dust-rainfall feedbacks in West African Sahel. *Water Resources Research* 44, W05202.
- Huenneke, L.F., Anderson, J.P., Remmenga, M., Schlesinger, W.H., 2002. Desertification alters patterns of aboveground net primary production in Chihuahuan ecosystems. *Global Change Biology* 8 (3), 247–264.
- Hutchinson, C.F., 1996. The Sahelian desertification debate: a view from the American south-west. *Journal of Arid Environments* 33 (4), 519–524.
- Khalaf, F.I., Misak, R., Al-Dousari, A., 1995. Sedimentological and morphological characteristics of some nabkha deposits in the northern coastal plain of Kuwait, Arabia. *Journal of Arid Environments* 29 (3), 267–292. issn:0140–1963.
- Kirkby, M.J., 1978. The stream head as a significant geomorphic threshold. Department of Geography, University of Leeds Working Paper 216.
- Lal, R., 1994. *Soil Erosion Research Methods*. Soil Water Conservation Society, St. Lucie Press, Delray Beach.
- Lal, R., 2001. Soil degradation by erosion. *Land Degradation & Development* 12 (6), 519–539.
- Lal, R., 2003. Soil erosion and the global carbon budget. *Environment International* 29 (4), 437–450.
- Lefever, R., Lejeune, O., 1997. On the origin of tiger bush. *Bulletin Of Mathematical Biology* 59 (2), 263–294.
- Lewis, J.P., Stofella, S.L., Feldman, S.R., 2001. Monk's tonsure-like gaps in the tussock grass *Spartina argentinensis* (Gramineae). *Revista De Biologia Tropical* 49 (1), 313–316.
- Li, J., Okin, G.S., Alvarez, L., Epstein, H., 2008. Effects of wind erosion on the spatial heterogeneity of soil nutrients in two desert grassland communities. *Biogeochemistry* 88 (1), 73–88.
- Ludwig, J.A., Wilcox, B.P., Breshears, D.D., Tongway, D.J., Imeson, A.C., 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology* 86 (2), 288–297.

- Mabbutt, J.A., Wilson, A.E. (Eds.), 1980. *Social and Environmental Aspects of Desertification*. The United Nations University, Tokyo.
- Marshall, J.K., 1973. Drought, land use and soil erosion. In: Lovett, J.V. (Ed.), *The Environmental, Economic and Social Significance of Drought*. Angus and Robertson, Sydney.
- MEA, 2005. *Millennium Ecosystem Assessment: Ecosystems and Human Well-Being: Desertification Synthesis*. World Resource Institute, Washington D.C.
- Middleton, N., Thomas, D., 1997. *World atlas of desertification*. Arnold, London.
- Montgomery, D.R., 2007. *Dirt: the erosion of civilization*. University of California Press, Berkeley.
- Nanson, G.C., Tooth, S., Knighton, A.D., 2002. A global perspective on dryland rivers: perceptions, misconceptions, and distinctions. In: Bull, L.J., Kirkby, M.J. (Eds.), *Dryland Rivers: Hydrology and Geomorphology of Semiarid Channels*. John Wiley and Sons, New York, pp. 17–54.
- Neff, J.C., Reynolds, R.L., Belnap, J., Lamothe, P., 2005. Multi-decadal impacts of grazing on soil physical and biogeochemical properties in southeast Utah. *Ecological Applications* 15 (1), 87–95.
- Nicholson, S.E., Tucker, C.J., Ba, M.B., 1998. Desertification, drought, and surface vegetation: an example from the West African Sahel. *Bulletin of the American Meteorological Society* 79 (5), 815–829.
- Nickling, W.G., Wolfe, S.A., 1994. The morphology and origin of Nabkhas, Region of Mopti, Mali, West Africa. *Journal of Arid Environments* 28, 13–30.
- Okin, G.S., 2008. A new model of wind erosion in the presence of vegetation. *Journal of Geophysical Research* 113, F02S10. doi:10.1029/2007JF000758.
- Okin, G.S., D'Odorico, P., Archer, S., 2009a. Impact of feedbacks on the stability of Chihuahuan desert grasslands. *Journal of Geophysical Research* 114, G01004. doi:10.1029/2008JG000833.
- Okin, G.S., Parsons, A.J., Wainwright, J., Herrick, J.E., Bestelmeyer, B.T., Peters, D.C., Fredrickson, E.L., 2009b. Do changes in connectivity explain desertification? *Bioscience* 59 (3), 237–244.
- Pickup, G., 1998. Desertification and climate change—the Australian perspective. *Climate Research* 11 (1), 51–63.
- Puigdefabregas, J., 2005. The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. *Earth Surface Processes and Landforms* 30 (2), 133–147.
- Ramanathan, V.P., Crutzen, J., Ramanathan, V.P., Crutzen, J., Kiehl, J.T., Rosenfeld, D., 2001. Aerosols, climate, and the hydrological cycle. *Science* 294, 2119–2124.
- Raupach, M.R., 1992. Drag and drag partitioning on rough surfaces. *Boundary-Layer Meteorology* 60 (4), 375–395.
- Ravi, S., D'Odorico, P., Over, T.M., Zobeck, T.M., 2004. On the effect of air humidity on soil susceptibility to wind erosion: the case of air-dry soils. *Geophysical Research Letters* 31, L09501.
- Ravi, S., Zobeck, T.M., Over, T.M., Okin, G.S., D'Odorico, P., 2006. On the effect of wet bonding forces in air-dry soils on threshold friction velocity of wind erosion. *Sedimentology* 53 (3), 597–609.
- Ravi, S., D'Odorico, P., Okin, G.S., 2007. Hydrologic and aeolian controls on vegetation patterns in arid landscapes. *Geophysical Research Letters* 34, L24S23.
- Ravi, S., D'Odorico, P., Wang, L., Collins, S.L., 2008. Form and function of grass ring patterns in arid grasslands: the role of abiotic controls. *Oecologia* 158, 545–555.
- Ravi, S., et al., 2009a. Post-fire resource redistribution in desert grasslands: a possible negative feedback on land degradation. *Ecosystems* 12 (3), 434–444.
- Ravi, S., D'Odorico, P., Zobeck, T.M., Over, T.M., 2009b. The effect of fire-induced soil hydrophobicity on wind erosion in a semiarid grassland: experimental observations and theoretical framework. *Geomorphology* 105, 80–86.
- Ravi, S., D'Odorico, P., Collins, S.L., Huxman, T.E., 2009c. Can biological invasions induce desertification? *New Phytologist* 181 (3), 512–515.
- Redman, C., 1999. *Human impact on ancient environments*. University of Arizona Press, Tucson.
- Reynolds, J.F., et al., 2007. Global desertification: building a science for dryland development. *Science* 316 (5826), 847–851.
- Rietkerk, M., van deKoppel, J., 1997. Alternate stable states and threshold effects in semi-arid grazing systems. *Oikos* 79 (1), 69–76.
- Robertson, J.H., 1939. A quantitative study of true-prairie vegetation after three years of extreme drought. *Ecological Monographs* 9 (4), 431–492.
- Rolett, B., Diamond, J., 2004. Environmental predictors of European deforestation on Pacific Islands. *Nature* 431, 443–446.
- Schlesinger, W.H., Reynolds, J.F., Cunningham, G.L., Huenneke, L.F., Jarrell, W.M., Virginia, R.A., Whitford, W.G., 1990. Biological feedbacks in global desertification. *Science* 247 (4946), 1043–1048.
- Seager, R., Ting, M., Held, M.I., Kushnir, Y., Lu, J., Vecchi, G., Huang, H.-P., Harnik, N., Leetmaa, A., Lau, N.-C., et al., 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316, 1181–1184.
- Singer, M.J., Shainberg, I., 2004. Mineral soil surface crusts and wind and water erosion. *Earth Surface Processes and Landforms* 29, 1065–1075.
- Stout, J.E., Zobeck, T.M., 1997. Intermittent saltation. *Sedimentology* 44, 959–970.
- Swap, R., et al., 1996. The long-range transport of southern African aerosols to the tropical South Atlantic. *Journal of Geophysical Research-Atmospheres* 101 (D19), 23777–23791.
- Taylor, C.M., Lambin, E.F., Stephenne, N., Harding, R.J., Essery, R.L.H., 2002. The influence of land use change on climate in the Sahel. *Journal of Climate* 15 (24), 3615–3629.
- Thomas, D.S.G., 1997. Science and the desertification debate. *Journal of Arid Environments* 37, 599–608.
- Toy, T.J., Foster, G.R., Renard, K.G., 2002. *Soil erosion: processes, prediction, measurement, and control*. John Wiley and Sons, New York.
- Turnbull, L., Wainwright, J., Brazier, R.E., 2008. A conceptual framework for understanding semi-arid land degradation: ecohydrological interactions across multiple-space and time scales. *Ecohydrology* 1 (1), 23–34.
- U.N.C.C.D., 1994. Elaboration of an international convention to combat desertification in countries experiencing serious drought and/or desertification. Particularly in Africa. U.N. Doc.A/AC.241/27, 33 I.L.M., p. 1328.
- Van Auker, O.W., 2000. Shrub invasions of North American semiarid grasslands. *Annual Review of Ecology and Systematics* 31, 197–215.
- van de Koppel, J., Rietkerk, M., van Langevelde, F., Kumar, L., Klausmeier, C.A., Fryxell, J.M., Hearne, J.W., van Andel, J., de Ridder, N., Skidmore, A., Stroosnijder, L., Prins, H.H.T., 2002. Spatial heterogeneity and irreversible vegetation change in semiarid grazing systems. *American Naturalist* 159 (2), 209–218.
- Veron, S.R., Paruelo, J.M., Oesterheld, M., 2006. Assessing desertification. *Journal of Arid Environments* 66 (4), 751–763.
- Visser, S.M., Sterk, G., Ribolzi, O., 2004. Techniques for simultaneous quantification of wind and water erosion in semi-arid regions. *Journal of Arid Environments* 59, 699–717.
- Weaver, J.E., Albertson, F.W., 1936. Effects on the great drought on the prairies of Iowa, Nebraska, and Kansas. *Ecology* 17 (4), 567–639.
- Whicker, J.J., Breshears, D.D., Wasiolek, P.T., Kirchner, T.B., Tavani, R.A., Schoep, D.A., Rodgers, J.C., 2002. Temporal and spatial variation of episodic wind erosion in unburned and burned semiarid shrubland. *Journal of Environmental Quality* 31, 599–612.
- Wolfe, S.A., Nickling, W.G., 1993. The protective role of sparse vegetation in wind erosion. *Progress in Physical Geography* 17, 50–68.
- Worster, D., 1979. *Dust Bowl: the Southern Plains of 1930s*. Oxford University press, New York.
- Xue, Y.K., Shukla, J., 1993. The influence of land-surface properties on Sahel climate. 1. Desertification. *Journal of Climate* 6 (12), 2232–2245.
- Yang, X., Rost, K.T., Lehmkühl, F., Zhenda, Z., Dodson, J., 2004. Evolution of dry lands in northern China and in the Republic of Mongolia since the Last Glacial Maximum. *Quaternary International* 118–119, 69–85.
- Zeng, N., Neelin, J.D., 2000. The role of vegetation–climate interactions and interannual variability in shaping the African savanna. *Journal of Climate* 13, 2665–2670.