Chapter 11
Vegetation Change in the Southwestern USA: Patterns and Processes

Laura Turnbull, John Wainwright, and Sujith Ravi

Abstract The southwestern USA has experienced multiple drivers of land degradation, combined with the interplay of both aeolian and hydrological processes. The southwestern USA has been one of the most intensively studied dryland systems in the world, and thus this region presents a great opportunity to explore ecogeomorphic linkages between drivers and disturbances with patterns and processes. In this case study we investigate the causes and consequences of land degradation in the southwestern USA, and explore how ecogeomorphic patterns and processes are changed as a result. A new exploratory modelling approach is then presented, which has been made possible because of the rich empirical data sets available from this region that enable parameterizing and testing of model outputs.

11.1 Introduction

One of the main characteristics of land degradation in drylands is an increase in erosion. Water erosion is significant in about 42 % of degrading drylands and wind erosion is significant in another 42 % of drylands (UNEP 1992). Wind dominates erosions in the dryland regions in Africa, runoff dominates erosion in the dryland regions of Australia (UNEP 1992), while in the dryland regions of

---

L. Turnbull (✉)
Institute of Hazards, Risk and Resilience, Department of Geography, Durham University,
Science Laboratories, South Road, Durham DH1 3LE, UK
e-mail: laura.turnbull@durham.ac.uk

J. Wainwright
Department of Geography, University of Durham, Durham DH1 3LE, UK
e-mail: john.wainwright@durham.ac.uk

S. Ravi
Department of Environmental Earth System Science, Stanford University, Stanford, CA, USA
e-mail: sujith@stanford.edu

E.N. Mueller et al. (eds.), *Patterns of Land Degradation in Drylands: Understanding Self-Organised Ecogeomorphic Systems*, DOI 10.1007/978-94-007-5727-1_11,

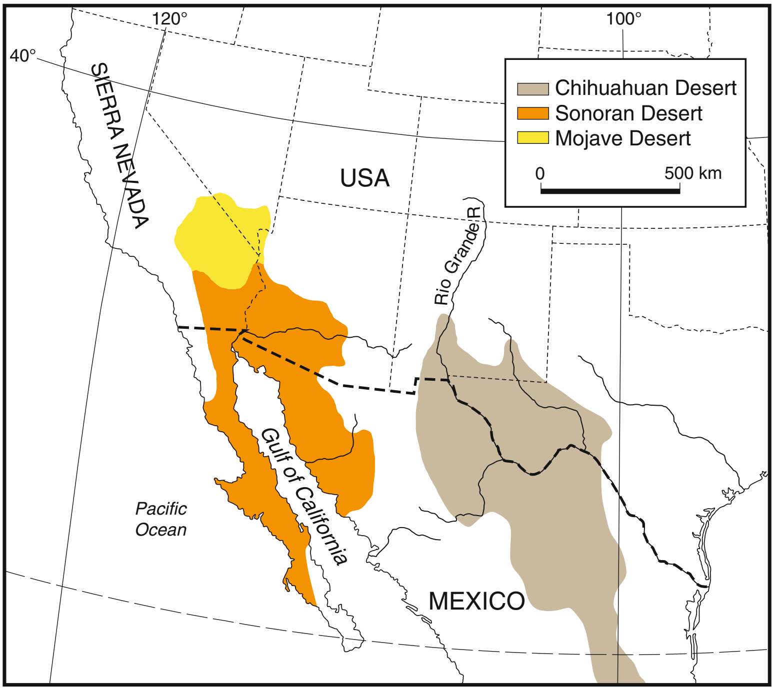


Fig. 11.1 Map showing the extent of the Sonoran, Mojave and Chihuahuan Deserts

southwestern USA, erosion by wind and water both contribute to land degradation. Over the last century this region has experienced a major increase in population, which has coincided with widespread vegetation transitions and land degradation across its major three desert regions; the Sonoran desert, the Mojave desert and the Chihuahuan desert (Fig. 11.1). The southwestern USA has experienced multiple drivers of degradation over the last century, including both changes in environmental drivers and human-induced disturbances. In contrast with previous episodes of vegetation transitions in this region, the most recent vegetation transition shows no signs of being readily reversible. The magnitude of changes in environmental drivers and human-induced disturbances is projected to increase in the future across this region. In this chapter, we explore how changes in environmental drivers and disturbances have altered patterns, processes and ecogeomorphic feedbacks that have led to observed changes in ecosystem state and land degradation across this region. The effects of multiple drivers and disturbances combined with the interplay of aeolian and hydrological processes make the southwestern USA an intriguing case study for the exploration of ecogeomorphic linkages between drivers and disturbances with patterns and processes.

The southwestern USA is one of the most intensively studied dryland regions of the world, and thus, using rich empirical data sets we explore changes in pattern-process relationships for the major types of vegetation change that have occurred across this region: the invasion of desert grasses by creosotebush and mesquite, the spread of piñon-juniper woodland and tree die off, and the invasion of desert shrublands by invasive grasses. Then, to develop further the advances in process understanding that have been made through field experimentation in these systems of the southwestern USA, we describe a modelling study which investigates the emergence of vegetation patterns following changes in drivers and disturbances. Notably, the southwestern USA is rare in terms of the richness of available empirical datasets, which are required to make this type of modelling approach possible.

11.2 Causes and Consequences of Land Degradation

Over the last century, the most widespread form of land degradation in the southwestern USA has resulted from relatively rapid and widespread invasion of native grasslands by shrubs (Brown et al. 1997; Van Auken 2000; Fig. 11.2), for

Enchanted Mesa, New Mexico

1899 1977

Walnut Gulch, southern Arizona

1883 1960

Fig. 11.2 Photographs of vegetation change in the southwestern USA. Juniper encroachment into rangeland at Enchanted Mesa, New Mexico between 1899 and 1977 (*upper*) and creosotebush encroachment into grassland and Walnut Gulch, southern Arizona between 1883 and 1960 (*lower*) (Sources: Allen et al. (1998) and Hastings and Turner (1965)) example in the lower-lying regions of the northern Chihuahuan Desert. The spread of piñon-juniper woodlands which are more common in higher elevation regions has also occurred, although chronic droughts have also caused widespread piñon (*Pinus edulis*) mortality. In New Mexico, the extent of grassland has reduced, primarily due to the encroachment of shrubs including creosotebush (*Larrea tridentata*) and mesquite (*Prosopis glandulosa*) into grasslands from the south, and juniper encroachment into grasslands from the north (Grover and Musick 1990). For example, at the Jornada Experimental Range in southern New Mexico, in 1916 mesquite was the primary dominant on 26 % of the area, but by 1998 it was the primary dominant on 59 % of the area (Gibbens et al. 2005). More recently, the invasion of native desert shrublands by exotic grasses has become a prolific problem, for example across regions of southern Arizona. These vegetation changes are widely perceived to have negative consequences for provision of essential ecosystem services; for example, the spread of invasive grasses such as Lehmann lovegrass (*Eragrostis lehmanniana*) reduces plant and animal diversity (Jones and Bock 2005; McClaran and Anable 1992), and the spread of piñon-juniper woodlands is perceived by some to cause increased erosion and net ecosystem degradation (Belsky 1996), although this will depend on site-specific characteristics of areas they are spreading into. Changes in vegetation in the southwestern USA continue to occur, and these changes and are anticipated to continue in the future (Archer and Predick 2008).

Disentangling the causes of these observed changes in vegetation across the southwestern USA is challenging because overgrazing, warming and fire suppression have all occurred simultaneously. Overgrazing is widely considered to be the primary cause of shrub encroachment into native grassland (Buffington and Herbel 1965) which is one of the most widespread changes in vegetation across the southwestern USA. Intensive cattle grazing in the southwestern USA started in the latter part of the 1800s, as coinciding with European settlement. In New Mexico, between 1870 and 1890 the number of livestock in New Mexico increased from 300,000 to 2,300,000 (Peterson 1950). In Arizona, the onset of intensive cattle grazing coincided with the arrival of the Southern Pacific railroad which enabled ranchers to transport their cattle to market (Bahre 1991; Sayre 1999). Within a few years of the onset of intensive grazing, Arizona's grasslands were already suffering from overgrazing with a major loss of grass cover (Noonan 2011). The legacy of overgrazing across the southwest USA persists today across many regions. Land degradation associated with shrub encroachment into native grasslands has also been attributed to climate warming. In the deserts of the southwestern USA the northern boundary of shrubs such as creosotebush and mesquite is limited by temperature, because freezing-induced cavitation (usually occurring a within a few degrees below 0 ºC) can cause stress and mortality (Felker et al. 1982; Pockman and Sperry 2000; Martinez-Vilalta and Pockman 2002). Therefore, with climate warming, shrubs are able to expand their spatial range into more northern latitudes. There have also been periods of severe drought, especially during the 1930s and 1950s. The USA has a history of fire suppression, which was carried out because of the fear of incontrollable and destructive spread of wildfire. However, the suppression of fires in the southwestern USA is thought to have exacerbated shrub encroachment, because fire kills woody vegetation, thus minimizing its spread. In combination with policy-driven fire suppression, overgrazing naturally serves to reduce the frequency of fires because a reduction in fuel load and thus a reduction in the capacity of grazed grasslands to carry fire. Conversely, when exotic grasses invade native shrublands, vegetation cover becomes more continuous and fuel loads increase, facilitation intense fires. The southwestern USA is also experiencing increasing rates of atmospheric N deposition (Baez et al. 2007; Fenn et al. 2003) which may affect species diversity. These different types of vegetation change in the southwestern USA are concurrent with increases in runoff and soil erosion (Abrahams et al. 1995; Wainwright et al. 2000; Wainwright et al. 2002), by both wind and water. These increases in runoff and erosion are widespread land-degradation problems because of their contributions to water quality problems and soil-fertility losses (Lado and Ben-Hur 2004; Martinez-Mena et al. 2001).

Vegetation change in the southwestern USA is not a new phenomenon. Vegetation proxy data from the northern Chihuahuan Desert indicate that during the Holocene there were episodes of grassland to shrubland transitions that were followed by a reversion to grassland (see review in Wainwright 2005). Pollen data indicate that during the full glacial conditions of the late Wisconsin period, mild winters and relatively cool summers persisted with a decline in summer rainfall and an increase in winter rainfall (Spaulding 1983). Other proxy data show recurrent droughts with a 100–130 year periodicity (Clark et al. 2002). The widespread changes in vegetation that have occurred over the southwestern USA over the last 150 years that are unprecedented in comparison with historic vegetation change in this region, in terms of both the rate at which these changes have occurred and their aerial extent. It is roughly estimated that mesquite is now present on more than 38 million hectares of former semi-arid grassland in the southwestern USA, while creosotebush is now the dominant shrub on more than 19 million hectares of former semi-arid grassland (Van Auken 2000).

Although vegetation change in the southwestern USA is not a new phenomenon, indicated by historic records, efforts to reverse vegetation change have proved largely unsuccessful. The key challenge is understanding vegetation change and associated land degradation is therefore to determine what makes this recent bout of vegetation change different from the previously reversible cycles of vegetation change. It is well established that while historic vegetation change was driven principally by climatic fluctuation, this recent bout of vegetation change has coincided with the anthropogenic drivers outlined previously. One plausible explanation is that the recent onset of anthropogenic drivers, in addition to environmental drivers, alters ecogeomorphic processes in such a way so as to cause hysteresis, rendering transitions largely irreversible.

Vegetation change in the southwestern USA is of great consequence to people, because of potential reductions in herbaceous productivity and therefore food production in the case of shrub invasion, and an increase the occurrence of widespread and intense fires in the case of exotic grass invasion. Shrub encroachment into grassland is associated with change in surface processes, notably increased runoff and erosion (Abrahams et al. 1995; Parsons et al. 1996a; Wainwright et al. 2000) and a change in the spatial distribution of soil properties that affect ecological and hydrological processes (Müller et al. 2008; Schlesinger et al. 1990, 1996; Turnbull et al. 2011). Biophysical and biogeochemical changes that occur during the invasion of grasslands by shrubs may affect land surface-atmospheric interactions, causing widespread biogeochemical feedbacks (Peterjohn and Schlesinger 1990; Schlesinger et al. 1990). Understanding how vegetation change alters the eco-geomorphic structure and function of drylands is necessary in order to develop management strategies for any given policy concerning vegetation change and land degradation, and to be able to provide policy makers and land managers with relevant information about the ecogeomorphic implications of land-management decisions that may accelerate vegetation change (Wilcox and Thurow 2006). To develop such management strategies to mitigate and reverse transitions in the southwestern USA, an in depth ecogeomorphic understanding is necessary, in which the in interactions between patterns and processes and biotic and abiotic components of the system are known.

11.3 Vegetation and Soil Responses to Climate Change and Disturbances

Drivers of vegetation change, such as climate and human-induced disturbances, directly affect different structural and functional components of the ecosystem and occur over a range of spatial and temporal scales (Turnbull et al. 2012). In the past, the southwestern USA has experienced prolonged changes in climate – for example, changes in annual precipitation and changes in the frequency and magnitude of rainfall events, and temperature change (Brown and Archer 1999; Gao and Reynolds 2003; Leopold 1951; Neilson 1986). Environmental drivers such as changes in the frequency and magnitude of precipitation have been shown to alter critical ecosystem processes including photosynthesis rates, soil respiration and net primary production. For example, in grasslands in central New Mexico, Annual Net Primary Productivity (ANPP) is highly variable – varying between 17 and 180 g m^{-2} a^{-1} over a 9-year period (Muldavin et al. 2008). This variability in ANPP arises due to inter-annual variability in precipitation. Rainfall manipulation experiments have shown that the size of the rainfall pulse has a great effect on the magnitude of response in ANPP (Thomey et al. 2011). At the same grassland site in central New Mexico, Thomey et al. (2011) found that while all functional types responded to increased precipitation pulses, the dominant C_4 grass species, black grama (*Bouteloua eriopoda*), responded most significantly. More abrupt changes and disturbances are also common in this region, such as sudden changes in land use, fire and drought (Scheffer et al. 2001). These short-duration disturbances often affect structural components of the system directly, rather than altering processes which in turn cause a change in the structural components of the system. For example, trampling by grazing animals or outdoor recreational vehicle may cause soil compression (Belnap and Eldridge 2003). Fire on the other hand can reduce or remove vegetation cover within a very short period of time.

Due to vegetation-soil feedbacks, changes in vegetation type and distribution can also affect soil properties. Soil is a critical resource since it is the medium through which plant-essential resources are made available. Soil is a critical resource in drylands in particular, because rates of soil production are low. It has been widely demonstrated that soil-structural and biological properties are heterogeneous, resulting from plant-scale feedbacks, which improve soil structure and nutrient content in the area underneath vegetation forming “islands of fertility”. This phenomenon has been well documented in the grasslands and shrublands of the southwestern USA (Müller et al. 2008; Schlesinger et al. 1996; Titus et al. 2002; Turnbull et al. 2010a). The distribution of soil microorganisms, which play a central role in the retention and release of nutrients (Gallardo and Schlesinger 1995), is also associated with these islands of fertility, since the distribution of soil microbial biomass depends on the ratio of carbon to extractable N (Gallardo and Schlesinger 1992) which is therefore also heterogeneous. Spatial variations in soil resources in piñon-juniper woodland, resulting from the extraction of nutrients from interspace soils by tree roots, and the concentration of these nutrients in the soil underneath canopies, through the deposition of plant litter (Covington and deBano 1990). Similarly Reiley et al. (2010) observed variations in soil carbon between canopy and intercanopy patches in piñon–juniper woodland, with higher concentrations under vegetation. Fewer studies have been carried out on the effects of exotic grass invasion on soil resources. The growth of invasive grasses may temporarily stabilize surface soils, as less bare ground is exposed to wind and water erosion. However, an increase in vegetation cover that occurs during exotic grass invasion increases the likelihood of fires occurring, following which bare surface soils are more susceptible to runoff and erosion, by both wind and water (Wilcox et al. 2012) that will contribute to progressive soil degradation. These changes in vegetation and soil structure have considerable implications for water-related processes, wind erosion and fire dynamics, which are discussed in more detail in the following sections.

11.4 The Roles of Water, Wind and Fire

In the deserts of the southwestern USA, soil-moisture dynamics have been demonstrated to exert a strong control over exchanges of water and carbon between the land surface and the atmosphere (Kurc and Small 2007; Muldavin et al. 2008; Barron-Gafford et al. 2011; Thomey et al. 2011) and the generation of runoff, and connectivity of runoff during rainfall events (Wainwright et al. 2000; Parsons et al. 1996a; Turnbull et al. 2010b). The partitioning of rainfall between runoff, infiltration and evapotranspiration is largely dependent upon the size of the rainfall event. The current precipitation regime across the southwestern USA tends to be characterized by many small storms and fewer large storms (Cavanaugh et al. 2011; Turnbull et al., in review). Ultimately, soil-moisture dynamics tend to be controlled by the partitioning of rainfall between infiltration and runoff, the depth of infiltration and evapotranspiration. Runoff coefficients have been shown to vary significantly with total event rainfall, with a greater proportion of rainfall being lost to runoff during particularly large and intense rainfall events (Turnbull et al. 2010b). For grassland and shrubland in the northern Chihuahuan desert, runoff coefficients can reach as high as 0.47 over grassland and 0.57 over shrubland at a spatial scale of 300 m^2 (Turnbull et al. 2010b). The portion of rainfall that infiltrates often only wets the top few tens of millimetres of the soil, with this surface soil layer forming the main water-storage reservoir in both grassland and shrubland soils (Kurc and Small 2004), except for larger rainfall events when water may infiltrate deeper. Native grass species such as black grama have a finely divided, well developed root system, mainly located in the uppermost 0.25 m of the soil (Campbell and Bomberger 1934). Thus, black grama tends to be very responsive to summer moisture and can greatly increase its cover in enhanced moisture conditions (Gosz and Gosz 1996; Noy-Meir 1973). Shrubs such as creosotebush and mesquite have tap roots that access deep soil-moisture reserves, extending to depths greater than 5 m (Gile et al. 1998; Martinez-Meza and Whitford 1996; Whitford et al. 1997). Therefore, productivity of shrubs such as creosotebush tends to be less sensitive to the timing of precipitation events that moisten the surface soil horizons (Cunningham et al. 1979). Sharifi et al. (1988) found that during successive wet periods, nitrogen becomes limiting, resulting in 50 % reduced vegetative production, thus decoupling growth from soil-moisture availability.

In the Chihuahuan and Sonoran Deserts which receive summer rainfall, drying out of the surface soil to the pre-rainfall soil-moisture content following a rainfall event typically only takes a few days (Kurc and Small 2004; Turnbull et al. 2010b). Most rainfall inputs are lost as evaporation, although the partitioning of evapotranspiration losses between evaporation and transpiration depends on the timing of precipitation with plant phenology (Cavanaugh et al. 2011). Evaporation dominates evapotranspiration until transpiration rates increase as plants start to respond to elevated soil moisture. However, a lag time of 3 weeks has been observed from the onset of summer rainfall to increases in transpiration rates (Cavanaugh et al. 2011). Overall, in creosotebush shrubland of southern Arizona, transpiration accounts for between 42 and 47 % of total evapotranspiration. These increases in transpiration rates causes a slight, but significant reduction in soil-moisture content in soil under vegetation compared with bare-surface soil in both black grama grassland and creosotebush shrubland (Turnbull et al. 2010b). Vegetation change also significantly alters runoff responses in the southwestern USA, for example, Turnbull et al. (2010b) found that peak discharges reached 479 $l\ min^{-1}$ over a shrub-dominated plot compared with only 289 $l\ min^{-1}$ over a grass-dominated plot. This difference in peak discharges arises because of the amount of runoff generated and because of the development of concentrated flow paths or rills (Abrahams et al. 1994; Lister et al. 2009; Schlesinger et al. 1999; Turnbull et al. 2010b; Wainwright et al. 2000). The high connectivity of areas of reduced infiltration in intershrub areas promotes enhanced runoff generation and flow connectivity as less runon infiltration occurs. It has been argued that these altered runoff dynamics make less water available for groundwater recharge (Hibbert 1983). Erosion on semi-arid hillslopes is controlled by interactions between raindrop-erosion processes and surface-flow processes (see Chap. 5). Rainsplash-erosion rates vary considerably over grassland and shrubland, with rates of 0.342 g m^{-2} min^{-1} over shrubland (Parsons et al. 1991) and 0.012 g m^{-2} min^{-1} (increasing to 0.054 g m^{-2} min^{-1}) over grassland (Parsons et al. 1994). The degree of flow concentration ultimately controls the proportion of this splash-detached sediment that is transported downslope by interill and rill flow (Abrahams et al. 1991; Luk et al. 1993). In an experiment comparing sediment loads over grassland and shrubland at a transport length of 21 m, loads of between 11.4 and 31.93 01 g m^{-1} min^{-1}, and 0.42 and 14.01 g m^{-1} min^{-1} were measured over shrubland and grassland respectively (Abrahams et al. 1991, 1995; Parsons et al. 1996a, b; Wainwright et al. 2000). Thus, where there is greater vegetation cover, there is an increase in the interception of raindrops which reduces their kinetic energy, and an increase in the hydraulic roughness of the surface due to plant stems and an increase in plant roots which bind the soil reducing its erodibility (Wainwright et al. 2000).

These runoff and erosion dynamics have great implications for lateral nutrient transfers. Schlesinger et al. (1999) investigated the transport of dissolved nutrients by conducting rainfall simulation experiments on grassland, shrubland and intershrub plots. Results of their study showed that grasslands produced greater nitrogen (N – the second-most limiting resource in these systems) losses in runoff with a weighted average concentration of 1.72 mg l^{-1} despite producing smaller quantities of runoff, while shrublands produce lower N losses, with a weighted average concentration of 0.55 mg l^{-1}. These differences in N losses in runoff can be explained by the spatial variation in soil-N content. For example, at the Sevilleta Long Term Ecological Research site in the northern Chihuahuan desert, the soil-N content is significantly higher under vegetation in shrubland. Here, the ratio of the mean concentration of available N measured under vegetation to that measured between vegetation is 2.93 for shrubland, while it is only 1.17 for grassland (Schlesinger et al. 1996). Thus, despite greater runoff generation in the soils between shrubs, the soil-N content in this zone is typically lower, reducing the amount of nitrogen available for dissolved transport in runoff (Turnbull et al. 2010a). The frequency and size (used here to refer to the length of the connected flow path) of runoff events has also been shown to affect catchment-scale nutrient export, since the length of time since the previous storm determines the amount of nutrient build up from atmospheric deposition, while the length of the flow path determines the length over which nitrification occurs which increases NO_3 concentrations (Welter et al. 2005).

Previously, it was thought that runoff events in these dryland systems do not cause a net loss of N (e.g. Schlesinger et al. 1999). However, more recently, it has been realized that there is a strong coupling between erosion and nutrient losses, with significant amounts of N being lost in particulate forms (Turnbull et al. 2011). Taking particle-bound nutrient losses into consideration, Turnbull et al. (2011) found that losses of nutrients from the system during rainfall-runoff events can

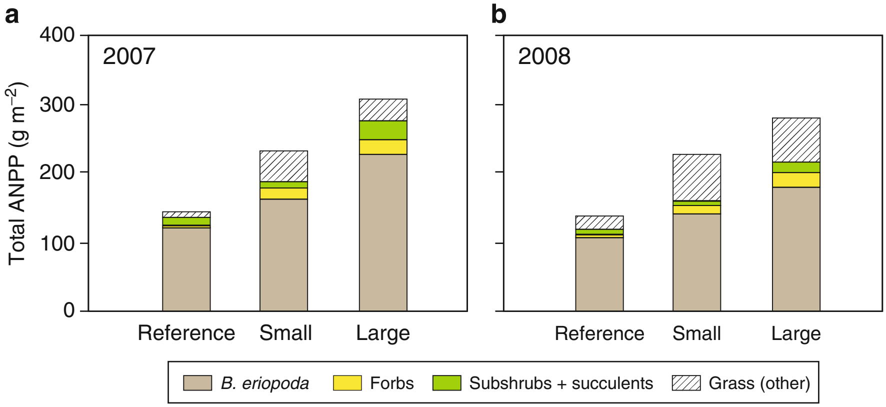


Fig. 11.3 Results of rainfall manipulation experiments carried out in black grama grassland at the Sevilleta National Wildlife Refuge, New Mexico, showing aboveground net primary productivity (ANPP) for each plant functional type in (**a**) 2007 (92.2 mm ambient rainfall) and (**b**) 2008 (125.8 mm ambient rainfall). *Bars* represent mean ANPP for reference (ambient rainfall conditions; $n = 3$), small rainfall (5 mm per week; $n = 5$), and large (20 mm per month; $n = 5$) treatment plots (Source: Thomey et al. 2011. Reprinted with permission from John Wiley and Sons, 2012)

greatly exceed inputs of nutrients to the system with up to ~2 kg N ha^{-1} a^{-1} being lost from shrublands in runoff – with the majority of this N as particle-bound N. Taking particle-bound N into consideration, total N losses from shrubland were over five times greater than total N losses from grassland. Therefore, in large events, these elevated N losses can lead to a net loss of N which is likely to contribute further to the degradation process.

At the Jornada Experimental Range in southern New Mexico, there has been a significant increase in precipitation over the last decade, but this increase in rainfall is made up as smaller and more frequent rainfall events, which do not generate as much runoff (Turnbull et al., in review). Therefore, on an annual basis, more water is retained within the ecosystem, for either uptake by plants or groundwater recharge. It is also probable that this reduction in runoff will cause a reduction in erosion, although this has not yet been tested. To investigate further the effects of rainfall variability on ecosystem processes, Thomey et al. (2011) carried out experimental rainfall manipulations to determine the effect of rainfall variability on ecosystem processes in grassland, while keeping the total annual amount of rainfall constant. Results of their study show that over a 2-year period, aboveground net primary productivity (ANPP) is consistently higher for rainfall regimes characterized by large rainfall events (20 mm received once per month) than for rainfall regimes characterized by small rainfall events (5 mm received once per week) (Fig. 11.3), because larger rainfall pulses increase soil-moisture content for a more prolonged period of time (Thomey et al. 2011). In another set of experiments, Reynolds et al. (1999) excluded rainfall to simulate the effects of seasonal drought on creosotebush growth dynamics. They found that creosotebush has a great capacity to shift its activity patterns to alternate periods to take advantage of resource availability. Shrubs also tend to be less sensitive to precipitation variability than grasses because they can access deeper down soil moisture reserves (Martinez-Meza and Whitford 1996; Whitford et al. 1997).

In piñon-juniper woodland it has been shown that there are large differences in runoff and erosion between different patch types in piñon-juniper woodland, with highest amounts of runoff and erosion generated in intercanopy patches compared with canopy areas (Reid et al. 1999). These patch/interpatch differences are comparable with dynamics observed in shrub/intershrub patches in the southwestern USA (Wainwright et al. 2000). In piñon-juniper woodland, lateral subsurface flow has also been found to be significant. Wilcox et al. (1997) found that during one winter, lateral subsurface flow was equivalent to 20 % of the snowpack. This contribution of subsurface flow to hydrological dynamics has implications for biogeochemical processing within these systems, since it has been observed that concentrations of chloride and dissolved organic carbon increase in lateral subsurface flow (Newman et al. 1998). The density of piñon-juniper stands has been shown to affect soil-moisture dynamics. Low-density stands tend to have higher soil-moisture content than high density stands, which is likely to be because of much greater transpiration occurring in the high density stands (Zhou et al. 2008).

There is little empirical data showing the effects of exotic grass invasion into native shrublands on runoff and erosion dynamics. In a recent study by Wilcox et al. (2012) demonstrated the potential for an initial reduction in runoff and water-driven erosion when exotic grasses invade native shrublands, due to an increase in vegetation cover. However, this increase in vegetation cover also has the effect of increasing fuel load, thereby increasing the potential for widespread and intense fires to occur. Model results suggest that large increases in runoff and erosion may occur following the occurrence of fire in exotic grasslands (Wilcox et al. 2012). Experiments have shown that following a precipitation pulse, evapotranspiration is highest from invasive grassland, but this increase is thought to be due to increases in soil evaporation (Huxman et al. 2004). Invasive grasses were not shown to increase carbon accumulation compared with native species (Huxman et al. 2004).

In the southwestern USA where water is the most limiting resource, the duration of dry periods differentially affects shrubs and grasses. For native grasses, the potential for growth in the summer is affected by the length of the spring drought, because the death of the root and shoot tissue reduces the number of growing points capable of utilizing the summer rainfall (Gao and Reynolds 2003). In contrast, because shrubs have deeper tap roots, they are more drought resistant and are therefore able to survive drought (Herbel and Gibbens 1996), even over multiple years (Reynolds et al. 1999). Drought has greatly affected piñon-juniper stands across the southwestern USA since the late 1990s. Piñon pine stands have experienced 40–95 % mortality over a million hectares, while co-occurring juniper (*Juniperus monosperma*) experienced lower (2–25 %) mortality (Breshears et al. 2005). However, the effects of drought-induced piñon mortality on hydrological processes and other key ecosystem functions such as carbon uptake and storage are not yet understood fully. The timing of drought is critical. For example, in areas where buffelgrass (*Cenchrus ciliaris*) has invaded in southern Arizona, if a wet period, which increases biomass and thus fuel load is followed by a dry period, the moisture content of the vegetation is low which increases is susceptibility to burn, potentially leading to raging fires (McDonald and McPherson 2011), such as those experienced in regions of southern Arizona in 2011.

Aeolian processes – the entrainment, transport and deposition of sediments by wind – are recognized as major abiotic drivers in dryland ecosystems. In the case of the southwestern USA, observed reduction in vegetation cover due recurrent droughts, vegetation transformations such as shrub encroachment and tree die off, and disturbances (natural and anthropogenic) have exponentially increased aeolian erosion and subsequent dust emissions (Munsen et al. 2011). Aeolian processes affect dryland ecosystems at multiple spatial scales. For example, impacts of accelerated aeolian erosion include loss and redistribution of soil resources and mechanical injury to growing plants, while the resulting dust emissions impact precipitation processes by providing cloud concentration nuclei, human health, climate and desertification (Field et al. 2010; Ravi et al. 2011). It is important to emphasize that in the southwestern USA where both aeolian and hydrological processes predominate, aeolian and hydrological processes are intimately coupled.

If soil is directly sheltered by vegetation, it is not easily eroded by wind because vegetation shelters and stabilizes the soil surface, in addition to extracting momentum from the wind (Okin et al. 2006). As a result, undisturbed grasslands tend not to lose much sediment due to wind erosion. However, when vegetation cover is reduced or removed, the erodibility of the surface increases (Okin et al. 2006). As with hydrological processes, the erosion and redistribution of fine sediments by aeolian activity can result in considerable changes in the soil properties, creating a heterogeneous landscape with a mosaic of sources and sinks with bare soil interspaces acting as sources and vegetated patches as sinks of nutrients and sediments (Okin et al. 2006). The deposition of fine sediments by wind onto vegetation patches result in considerable changes in the soil texture and consequent changes in soil hydrological properties like soil moisture, infiltration and runoff. Thus, the differential rates of soil deposition and removal by wind result in differential rates of hydrological processes, thereby affecting the formation and expansion of vegetation patterns as observed in the case of mesquite nebkhat and blue-grama grass rings in the Chihuahuan Desert (Ravi et al. 2007a, b, 2008). Several studies have highlighted the prominent role played by aeolian erosion in grasslands undergoing encroachment by shrubs (Okin and Gillette 2001), and grasslands that have experienced extensive grazing (Belnap et al. 2009). Sediment removal from undisturbed grasslands in the southwestern USA ranges between 1.5 and 23 g m^{-2} day^{-1} (Belnap et al. 2009; Breshears et al. 2003; Offer and Goossnes 2004; Visser et al. 2004; Whicker et al. 2008). Conversely, sediment removal from a grassland site disturbed by current grazing generates 41 times more wind-eroded sediment (Belnap et al. 2009).

The removal and redistribution of sediment through wind erosion has important implications for the biogeochemistry of these systems. The removal of nutrient-rich soil from intercanopy areas and subsequent redistribution onto the vegetated shrub patches by aeolian processes may contribute to the formation of “islands of fertility” (Schlesinger et al. 1990; Okin and Gillette 2001; Li et al. 2008, and discussion in Chap. 3). For example, using a multi-year grass removal experiment in the Chihuahuan desert grassland, Li et al. (2007) showed that accelerated wind erosion resulting from grass removal removed up to 25% of total organic carbon and total nitrogen from the top 50 mm of soil over the course of three wind seasons. The amount of grass cover removal was critical; with grass cover reduction of up to 25–50 %, the balance between the net loss of nutrients and biotic was positive and thus nutrients were still accumulating, but with further grass cover reduction, the balance switched to negative indicating a net loss of nutrients (Li et al. 2007). As grass cover declines as shrubs invade, the dominance of aeolian transport processes increases (Breshears et al. 2003), largely because of two characteristics that enhance wind erosion: wake interference flow and bare intercanopy areas (Breshears et al. 2003). Thus, in systems undergoing a transition from shrubland to grassland, the major ecosystem functions can be explained and predicted in terms of spatial and temporal distribution of soil resources, which in turn is controlled by sediment transport processes, and wind erosion accounts for significant amounts of soil and nutrient loss and redistribution from shrub-grass transition systems.

Fire is a major factor controlling the ecosystem structure and function in desert grasslands of the American South West. Fires play a critical role in determining the dominance or co-dominance of grasses and woody plants in arid ecosystems. On the other hand, vegetation affects the fire regime as both the intensity and frequency of fires depend on the relative abundance of trees and grasses (Van Wilgen et al. 2003). A fire return frequency of 10 years is usually sufficient to suppress woody vegetation (McPherson 1995); however, a reduced fire frequency due to fire suppression may be insufficient to curtail shrub growth. Vegetation cover affects soil erosion both directly, by limiting the exposure of the soil surface to the erosive action of wind and water. Fire, in turn, affects erosion processes by vegetation removal and altering soil properties. Fires induced changes in soil properties – such as soil hydrophobicity – are known to affect infiltration, runoff and water erosion (see Chap. 5). The effect of fire on soil properties depends on fire regime (e.g. fire temperature), soil properties and vegetation type. More recently it has been showed that fire-induced water repellency enhances soil erosion and subsequent dust emissions by weakening the strength of interparticle forces between soil grains (Ravi et al. 2006).

Grasslands and shrublands in the southwestern USA that are currently not significant dust sources may turn into major dust sources following disturbances such as fires. In the case of grasslands, periodic fires are known to increase the availability of some nutrients and prescribed fires are often used as a management tool to control shrubs and to enhance grass productivity. However, fire occurrences followed by dry and windy conditions may result in substantial losses of soil resources, potentially causing a reduction in grass productivity (Whicker et al.

2002; Ravi et al. 2007a, b; Sankey et al. 2009). Enhanced, post-fire soil erosion is considered to be a significant but largely underestimated mechanism for the removal of soil and nutrient-rich burnt material immediately following wild fires. Frequent fires are thought to extend the “window of disturbance” in which soil erosion is intensified (i.e. the post-fire period of enhanced sediment yields). Human activities have profoundly impacted the fire frequency in desert grasslands of the Southwest. The introduction of cattle after European settlement led to an enhancement of mesquite seed dispersal, the degradation of the grass layer, and a reduction in fire frequency and intensity (Archer 1989; van Auken 2000). These changes resulted in a self-sustained cycle of soil erosion, depletion of soil resources, and vegetation loss, which may have, contributed to the irreversible denudation of grass-dominated areas (Archer et al. 1995; Schlesinger et al. 1990) and facilitated the further encroachment of shrubs. Recently, it has been shown that in the early stages of shrub encroachment, reintroducing fires may enhance grass cover, by redistributing resources from burned shrub islands to the interspaces leading to the homogenization of the landscape (Ravi et al. 2009a, b). Thus fire-erosion feedbacks during this phase can favor the redistribution of soil resources from the resource islands beneath the shrub canopies to the nutrient-depleted interspaces, thereby promoting the reconversion of the landscape into a state with more uniform distribution of grass and depleted soil properties (Ravi et al. 2009a, b; Ravi and D’Odorico 2009).

In many arid shrubland systems of the southwestern USA invasion by exotic annual grasses have increased the frequency and intensity of fires. The invasive grasses such as buffelgrass, Lehmann love grass (*Eragrostis lehmanniana*) and red brome (*Bromus rubens*) all provide persistent fine fuels and are fire adapted, while the native vegetation in the Sonoran and Mojave are not adapted to fire. Thus, the fire cycle created by the invasive grasses threatens both native annual and perennial species in these ecosystems. Further, less diverse ecosystems are less resilient and hence are more susceptible to irreversible changes under the influence of external drivers (Chapin et al. 1997). More recently it has been proposed that interactions between annual grass invasion and inter-annual precipitation variability may drive stable desert shrublands to a desertified state. The invasion by exotic annual grasses can destroy the heterogeneity of soil resources typical of desert shrublands (fire-induced shrub mortality and resource redistribution) and favour the conversion into exotic grasslands. However, recurrent droughts, which are predicted to be more frequent in the North American deserts (Seager et al. 2007), may displace these invasive grass cover followed by even higher erosion rates and irreversible losses of soil resources (Brooks and Pyke 2001; Ravi et al. 2009a, b). Thus, climate change and exotic grasses invasions, the two major drivers of global environmental change, may act in concert to amplify each other’s effect on land cover and soil resources (Ravi et al. 2009a, b).

11.5 Modelling Interactions Between Ecology, Hydrology and Geomorphology and the Emergence of Patterns

The previous sections have explored some of the processes that have received lots of attention in trying to understand land degradation processes in the southwestern USA. In an attempt to understand the combined effects of different types of drivers on patterns and processes, Stewart et al. (in press) have carried out integrated modelling work on the effects of wind, water and animal redistribution of resources on vegetation and soil patterns in the southwestern USA. Their model links drivers and responses (Fig. 11.4), to enable processes to be considered across landscape scales, integrating understanding about patch-scale dynamics with process representation at larger spatial scales. They use a connectivity framework (see Chaps. 5 and 7) to control how vectors of movement are modified in the presence of vegetation (Fig. 11.5). An advantage of this form of modelling of the development of pattern is that it is explicitly process based, and therefore can be interpreted in terms of observable parameters from the field.

The model was parameterized using field and published data from the Jornada LTER site. The site has the advantage of providing a wealth of information from both the LTER project and the USDA-ARS site established since 1914 (Havstad

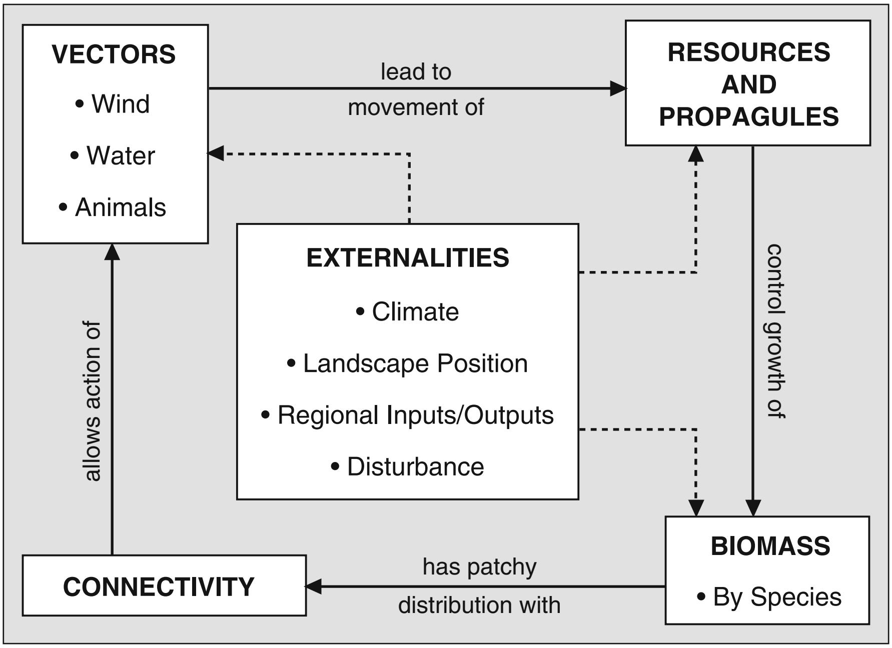


Fig. 11.4 Conceptual integrated model of Stewart et al. (perpetually in press) showing how different vectors of movement may redistribute resources and propagules in ways that may lead to pattern formation, as driven by external drivers and local feedbacks

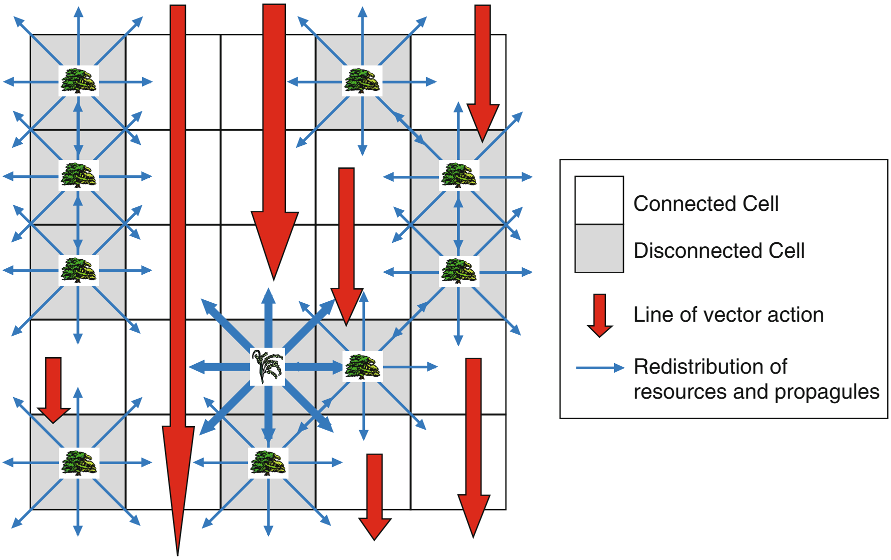


Fig. 11.5 Illustration of how different vegetation types affect the redistribution of resources and propagules as distributed by the different vectors (see Fig. 11.5) in the model of Stewart et al. (in press)

et al. 2006; Jornada LTER 2012). Direct climate records go back to 1914, and Wainwright (2005) extended these back to 1659 using tree-ring proxies. Historical grazing levels are provided by Havstad et al. (2006).

Initial results demonstrated that the formation of patterns in the model was directly controlled by the interaction of advective (movement by vectors) and diffusive (redistribution around plants) processes, which is compatible with other interpretations of pattern-forming mechanisms (see discussion in Chaps. 3 and 8). However, they also demonstrated that although vegetation patterns showed a significant amount of equifinality when driven by different surface processes, there were notable differences in subsurface resource characteristics with these different drivers. Thus, future research needs to emphasize the coupling of surface and subsurface conditions.

A second set of model experiments evaluated the extent to which persistent droughts affect the formation of pattern, using the conditions of the major droughts in the 1930s and 1950s as analogues. Stewart et al. (in press) demonstrated that pattern formation was strongest in the period immediately following prolonged drought (Fig. 11.6), and that single drought episodes were unlikely to lead to the invasion of shrubland and the development of patchy landscapes, unless there was significant grazing pressure. These results showed that it is possible to represent the whole range of observed behaviour of vegetation change at Jornada over the twentieth century, in terms of the timing and pattern of grass loss, when compared to the quadrat measurements of Yao et al. (2006) (Fig. 11.7). Thus, future

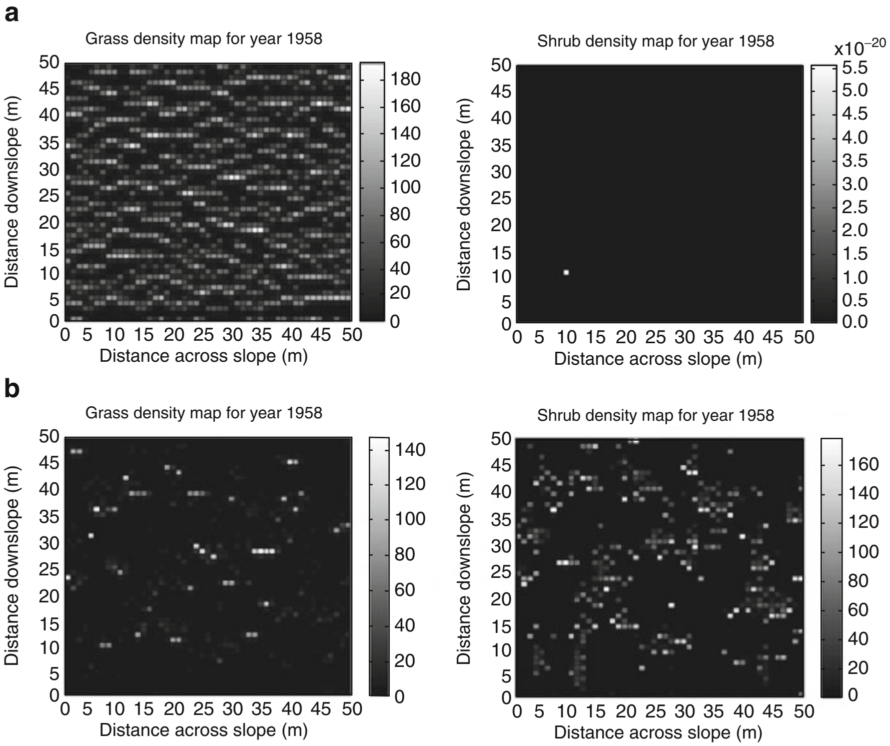


Fig. 11.6 Grass and shrub densities for example simulations using the model of Stewart et al. (in press). (**a**) Baseline simulation showing the dominant of grass patches at a variety of scales; (**b**) the combined effect of drought and high grazing pressure producing a landscape dominated by shrub patches

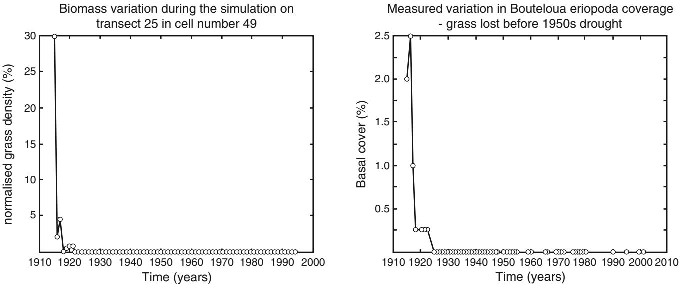
developments of the model show great promise in both the interpretation of dryland ecogeomorphic dynamics and the derivation of appropriate management strategies for mitigating past and future land degradation.

11.6 Conclusion

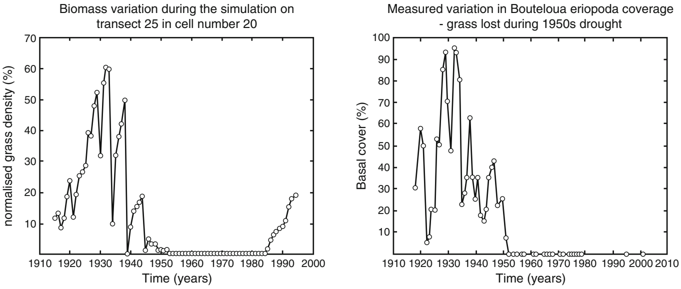
Land degradation in the US Southwest is a complex, evidenced by intimate coupling between hydrological and aeolian processes and anthropogenic influences – fire and grazing. It has been known for decades that interpreting the mechanisms of change in the drylands of the southwest USA is difficult, not least because of issues of equifinality (Cooke and Reeves 1976). In the southwestern USA, detailed ecological, hydrological and geomorphic data have been collected over multiple decades, which notably is considerably longer than the length of data

a Data illustrating grass biomass lost prior to the 1950s drought

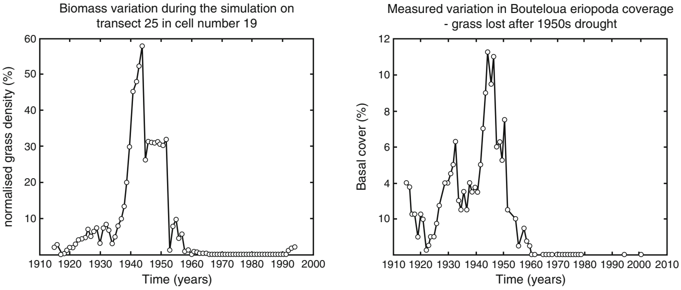
Modelled grass response **Measured grass response**



b Data illustrating grass biomass lost during the 1950s drought



c Data illustrating grass biomass lost after the 1950s drought



d Data illustrating grass biomass that persists after the 1950s drought

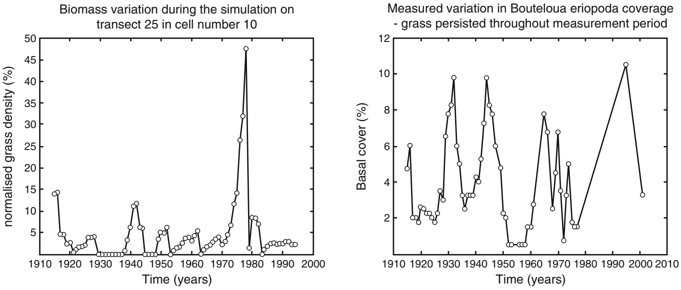


Fig. 11.7 Comparison of data simulated by Stewart et al. (in press) from the *centre line* transect of the grid, as compared with the field observations of Yao et al. (2006)

collection across most other drylands regions, and there is information on the extent and intensity of grazing intensities over the last, yet it remains difficult to unpick the multiple lines of causality of land degradation. Furthermore, the all too common fragmentation of investigation into different disciplinary areas has meant that scientific advances have been slow and disconnected. Thus, it is unsurprising that we struggle to overcome the problem of equifinality in regions where such detailed information is not available. However, ongoing research in this region is becoming increasingly coupled and holistic, in the sense that it is increasingly being recognized that the coupling between hydrological and aeolian processes is critical in understanding system dynamics at a range of scales, from the patch scale up to the landscape scale (Belnap et al. 2011; Field et al. 2009; Ravi et al. 2010).

The spatial and temporal extent of field studies relating to land degradation in the southwestern USA, have provided a wealth of data that have enabled the development of a range of models that have been extensively parameterised and tested for these systems, including a wind erosion model, vegetation dynamics models and runoff and erosion models (see Chap. 7 for more detail). Furthermore, this wealth of available data has enabled the development of a new modelling approach (Stewart et al., in press), that deals with the combined effects of resource and propagule redistribution by ecogeomorphic vectors and resulting feedbacks between different components of the system. The heuristic approach to modelling pattern emergence – i.e. using the model to decide what needs to be observed in a landscape, and what might be missing from our current empirical understanding of the ecogeomorphic system – is a new approach to investigate land degradation in the southwestern USA. Using their model to simulate vegetation patterns typical in the southwestern USA in response to different types of vectors and diffusive processes, they were not able to overcome the problem of equifinality. They were however able to identify that some processes affect patterns of resource distributions more than others, which are likely to have significant implications in terms of species interactions and species abundance, and thus, the long term composition of vegetation communities.

It is this combination of detailed field-based studies and model development that has been widely undertaken drylands of the southwestern USA that has enabled incremental improvements of our understanding of patterns, processes and their interactions. However, in order to be able to predict with greater certainty how these systems will respond to future changes in climate and changes in land use/management the problem of equifinality needs to be resolved more fully by continued investigation of effects of system derives on patterns, processes and feedbacks.

Acknowledgments This chapter is a contribution to the book *Patterns of Land Degradation in Drylands: Understanding Self-Organised Ecogeomorphic Systems*, which is the outcome of an ESF-funded Exploratory Workshop – "Self-organized ecogeomorphic systems: confronting models with data for land degradation in drylands" – which was held in Potsdam, Germany, 7–10 June 2010. The work of LT and JW in the US Southwest has been supported by NSF award DEB-0217774 to the University of New Mexico for Long-Term Ecological Research at the Sevilleta National Wildlife Refuge, NSF award DEB-0618210 for Long-Term Ecological Research at the

Jornada Experimental Range, The Worshipful Company of Farmers, and the Royal Society Dudley Stamp Memorial Fund ; SR has been supported by NSF award DEB-0217774 to the University of New Mexico for Long-Term Ecological Research at the Sevilleta National Wildlife Refuge.

References

Abrahams AD, Parsons AJ, Luk SH (1991) The effect of spatial variability in overland flow on the downslope pattern of soil loss on a semiarid hillslope, southern Arizona. Catena 18:255–270

Abrahams AD, Parsons AJ, Wainwright J (1994) Resistance to overland-flow on semiarid grassland and shrubland hillslopes, Walnut Gulch, Southern Arizona. J Hydrol 156:431–446

Abrahams AD, Parsons AJ, Wainwright J (1995) Effects of vegetation change on interrill runoff and erosion, Walnut Gulch, southern Arizona. Geomorphology 13:37–48

Allen CD, Betancourt JL, Swetnam TW (1998) Landscape changes in the southwestern United States: techniques, long-term data sets, and trends. In: Sisk TD (ed) Perspectives on the land-use history of North America: a context for understanding our changing environment. US Geological Survey, Biological Resources Division, Biological Science Report USGS/BRD/BSR 1998-0003 (Revised September 1999), 104 pp

Archer S (1989) Have Southern Texas savannas been converted to woodlands in recent history? Am Nat 134:545–561

Archer S, Predick KI (2008) Climate change and ecosystems of the southwestern United States. Soc Range Manage 30:23–28

Archer S, Schimel DS, Holland EA (1995) Mechanisms of shrubland expansion: land use, climate or CO^2? Clim Chang 29:91–99

Baez S, Fargione J, Moore DI, Collins SL, Gosz JR (2007) Atmospheric nitrogen deposition in the northern Chihuahuan desert: temporal trends and potential consequences. J Arid Environ 68:640–651

Bahre CJ (1991) A legacy of change: historic human impact on vegetation of the Arizona Borderlands. University of Arizona Press, Tucson

Barron-Gafford GA, Scott RL, Jenerette DG, Huxman TE (2011) The relative controls of temperature, soil moisture, and plant functional group on soil CO_2 efflux at diel, seasonal and annual scales. J Geophys Res 116, G01023

Belnap J, Eldridge D (2003) Disturbance and recovery of biological soil crusts. In: Belnap J, Lange OL (eds) Biological soil crusts: structure, function, and management, vol 150, Ecological studies series. Springer, Berlin, pp 363–383

Belnap J, Reynolds R, Reheis MC, Phillips SL, Urban FE, Goldstein HL (2009) Sediment losses and gains across a gradient of livestock grazing and plant invasion in a cool, semi-arid grassland, Colorado Plateau, USA. Aeolian Res 1:27–43

Belnap J, Munson SM, Field JP (2011) Aeolian and fluvial processes in dryland regions: the need for integrated studies. Ecohydrology 4:615–622. doi:10.1002/eco.258

Belsky AJ (1996) Viewpoint: Western juniper expansion: is it a threat to arid northwestern ecosystems? J Range Manage 49:53–59

Breshears DD, Whicker JJ, Johanse P, Pinder JE (2003) Wind and water erosion and transport in semi-arid shrubland, grassland and forest ecosystems: quantifying dominance of horizontal wind-driven transport. Earth Surf Process Landf 28:1189–1209

Breshears DD, Cobb NS, Rich PM, Price KP, Allen CD, Balice RG, Romme WH, Kastens JH, Floyd ML, Belnap J, Anderson JJ, Myers OB, Meyer CW (2005) Regional vegetation die-off in response to global-change-type drought. Proc Natl Acad Sci 102:15144–15148

Brown JR, Archer S (1999) Shrub invasion of grassland: recruitment in continuous and not regulated by herbaceous biomass or density. Ecology 80:2385–2396

Brown JR, Valone TJ, Curtin CG (1997) Reorganization of an arid ecosystem in response to recent climate change. Proc Natl Acad Sci 94:9729–9733

Brooks ML, Pyke DA (2001) Invasive plants and fire in the deserts of North America. In Galley KEM, Wilson TP (eds) Proceedings of the invasive species workshop: the role of fire in the spread and control of invasive species. Fire conference 2000: the first national congress on fire ecology, prevention, and management. Miscellaneous Publication No. 11. Tall Timbers Research Station, Tallahassee, FL, pp 1–14

Buffington LC, Herbel CH (1965) Vegetational changes on a semidesert grassland range from 1858 to 1963. Ecol Monogr 35:139–164

Campbell RS, Bomberger EH (1934) The occurrence of *Gutierrezia sarothrae* on *Bouteloua eriopoda* ranges in southern New Mexico. Ecology 15:49–61

Cavanaugh ML, Kurc SA, Scott RL (2011) Evapotranspiration partitioning in semiarid shrub-land systems: a two-site evaluation of soil moisture control on transpiration. Ecohydrology 4:671–681

Chapin FS, Walker BH, Hobbs RJ, Hooper DU, Lawton JH, Sala OE, Tilman D (1997) Biotic control over the functioning of ecosystems. Science 277:500–504

Clark JS, Grimm EC, Donovan JJ, Fritz SC, Engstrom DR, Almendinger JE (2002) Drought cycles and landscape responses to past aridity on prairies of the northern great plains, USA. Ecology 83:595–601

Cooke RU, Reeves RW (1976) Arroyos and environmental change in the American Southwest. Clarendon, Oxford

Covington WW, DeBano LF (1990) Effects of fire on pinyon-juniper soils. In: Krammes JS (ed) Effects of fire management of southwestern natural resources. Workshop proceedings, USDA Forest Service, General Technical Report RM-191, Fort Collins, pp. 78–86

Cunningham GL, Syvertsen JP, Reynolds JF, Willson JM (1979) Some effects of soil-moisture availability on above-ground production and re-production allocation in Larrea tridentata (DC) cov. Oecologia 40:113–123

Felker P, Clark PR, Nash P, Osborn JF, Cannell GH (1982) Screening Prosopis (mesquite) for cold tolerance. For Sci 28:556–562

Fenn M, Haeuber R, Tonnesen GS, Baron JS, Grossman-Clarke S, Hope D, Jaffe DA, Copeland S, Geiser L, Rueth HM, Sichman JO (2003) Nitrogen emissions, deposition and monitoring in the western United States. Bioscience 53:391–403

Field JP, Breshears DD, Whicker JJ (2009) Toward a more holistic perspective of soil erosion: why aeolian research needs to explicitly consider fluvial processes and interactions. Aeolian Res 1:9–17

Field JP, Belnap J, Breshears DD, Neff JC, Okin GS, Whicker JJ, Painter TH, Ravi S, Reheis MC, Reynolds RR (2010) The ecology of dust. Front Ecol Environ 8:423–430

Gallardo A, Schlesinger WH (1992) Carbon and nitrogen limitation of soil microbial biomass in desert ecosystems. Biogeochemistry 18:1–17

Gallardo A, Schlesinger WH (1995) Factors determining soil microbial biomass and nutrient immobilization in desert soils. Biogeochemistry 28:55–68

Gao Q, Reynolds JF (2003) Historical shrub–grass transitions in the northern Chihuahuan Desert: modeling the effects of shifting rainfall seasonality and event size over a landscape. Glob Chang Biol 9:1–19

Gibbens RP, McNeely RP, Havstad KM, Beck RF, Nolen B (2005) Vegetation changes in the Jornada Basin from 1858 to 1998. J Arid Environ 61:651–668

Gile LH, Gibbens RP, Lenz JM (1998) Soil induced variability in root systems of creosotebush (Larrea tridentate) and tarbush (Flourencia cerna). J Arid Environ 39:57–78

Gosz RJ, Gosz JR (1996) Species interactions on the biome transition zone in New Mexico: response of blue grama (*Bouteloua gracilis*) and black grama (*Bouteloua eriopoda*) to fire and herbivory. J Arid Environ 34:101–114

Grover HD, Musick HB (1990) Shrubland encroachment in southern New Mexico, U.S.A.: an analysis of desertification processes in the American Southwest. Clim Change 17:305–330

Hastings JR, Turner RM (1965) The changing mile: an ecological study of vegetation change with time in the lower mile of an arid and semi-arid region. University of Arizona Press, Tucson, 317 pp

Havstad KM, Huenneke LF, Schlesinger WH (eds) (2006) Structure and function of a Chihuahuan Desert ecosystem: the Jornada Basin Long-Term Ecological Research site. Oxford University Press, New York

Herbel CH, Gibbens RP (1996) Post-drought vegetation dynamics on arid rangelands in southern New Mexico, Agricultural Experiment Station Bulletin 776. New Mexico State University, Las Cruces

Hibbert AR (1983) Water yield improvement potential by vegetation management on western rangelands. Water Resour Bull 19:375–381

Huxman TE, Cable JM, Ignace DD, Eilts JA, English NB, Weltzin J, Williams DG (2004) Response of net ecosystem gas exchange to a simulated precipitation pulse in a semi-arid grassland: the role of native versus nonnative grasses and soil texture. Oecologia 141:295–305

Jones ZF, Bock CE (2005) The Botteri's Sparrow and exotic Arizona grasslands: an ecological trap or habitat regained? Condor 107:731–741

Jornada LTER (2012) Jornada Basin long term ecological research. http://jornada-www.nmsu.edu/. Last accessed 28 Mar 2012

Kurc SA, Small EE (2004) Dynamics of evapotranspiration in semiarid grassland and shrubland ecosystems during the summer monsoon season, central New Mexico. Water Resour Res 40, W09305

Kurc SA, Small EE (2007) Soil moisture variations and ecosystem-scale fluxes of water and carbon in semiarid grassland and shrubland. Water Resour Res 43, W06416

Lado A, Ben-Hur A (2004) Soil mineralogy effects on seal formation, runoff and soil loss. Appl Clay Sci 24:209–224

Leopold LB (1951) Rainfall frequency: an aspect of climatic variation. Trans Am Geophys Union 32:347–357

Li J, Okin GS, Alvarez L, Epstein H (2007) Quantitative effects of vegetation cover on wind erosion and soil nutrient loss in a desert grassland of southern New Mexico, USA. Biogeochemistry 85:317–332

Li J, Okin GS, Alvarez L, Epstein H (2008) Effects of wind erosion on the spatial heterogeneity of soil nutrients in two desert grassland communities. Biogeochemistry 88:73–88

Lister D, Michaelides K, Wainwright J, Parsons AJ (2009) Controls on small-scale runoff and erosion dynamics in different vegetation communities in a dryland environment. Hydrol Process 23:1617–1630

Luk S, Abrahams AD, Parsons AJ (1993) Sediment sources and sediment transport by rill flow and interill flow on a semi-arid piedmont slope, southern Arizona. Catena 20:93–111

Martinez-Meza E, Whitford WG (1996) Stemflow, throughfall and channelization of stemflow by roots in three Chihuahuan desert shrubs. J Arid Environ 32:271–287

Martinez-Mena M, Castillo V, Albaladejo J (2001) Hydrological and erosional response to natural rainfall in a semi-arid area of south-east Spain. Hydrol Proc 15:557–571

Martinez-Vilalta J, Pockman WT (2002) The vulnerability to freezing induced xylem cavitation of Larrea tridentata (Zygophyllaceae) in the Chihuahuan desert. Am J Bot 89:1916–1924

McClaran MP, Anable ME (1992) Spread of introduced Lehmann lovegrass along a grazing intensity gradient. J Appl Ecol 29:92–98

McDonald CJ, McPherson GR (2011) Fire behaviour characteristics of buffelgrass-fueled fires and native plant community composition in invaded patches. J Arid Environ 75:1147–1154

McPherson GR (1995) The role of fire in the desert grassland. In: McClaran MP, Van Devender TR (eds) The desert grassland. University of Arizona Press, Tucson

Muldavin EH, Moore DI, Collins SL, Wetherill KR, Lightfoot DC (2008) Aboveground net primary production dynamics in a northern Chihuahuan Desert ecosystem. Oecologia 155:123–132

Munsen SM, Belnap J, Okin GS (2011) Responses of wind erosion to climate-induced vegetation changes on the Colorado Plateau. Proc Natl Acad Sci 108:3854–3859

Müller EN, Wainwright J, Parsons AJ (2008) Spatial variability of soil and nutrient parameters within grasslands and shrublands of a semi-arid environment, Jornada Basin, New Mexico. Ecohydrology 1:3–12

Neilson RP (1986) High-resolution climatic analysis and southwest biogeography. Science 232:27–34

Newman BD, Campbell AR, Wilcox BP (1998) Lateral subsurface flow pathways in a semiarid ponderosa pine hillslope. Water Resour Res 34:3485–3496

Noonan GR (2011) The overgrazing of Arizona rangelands. You could actually throw a rock from one carcass to another. Sci Quest Tech Paper 2

Noy-Meir I (1973) Desert ecosystems: environment and producers. Ann Rev Ecol Syst 4:25–41

Offer ZY, Goossens D (2004) Thirteen years of aeolian dust dynamics in a desert region (Negev desert, Israel): analysis of horizontal and vertical dust flux, vertical dust distribution and dust grain size. J Arid Environ 57:117–140

Okin GS, Gillette DA (2001) Distribution of vegetation in wind-dominated landscapes: implications for wind erosion modeling and landscape processes. J Geophys Res 106:9673–9683

Okin GS, Gillette DA, Herrick JE (2006) Multi-scale controls on and consequences of Aeolian processes in landscape change in arid and semi-arid environments. J Arid Environ 65:253–275

Parsons AJ, Abrahams AD, Luk S (1991) Size characteristics of sediment in interill overland flow on a semi-arid hillslope, Southern Arizona. Earth Surf Proc Landf 16:143–152

Parsons AJ, Abrahams AD, Wainwright J (1994) Rainsplash and erosion rates in an interill area on semi-arid grassland, southern Arizona. Catena 22:215–226

Parsons AJ, Abrahams AD, Wainwright J (1996a) Responses of interrill runoff and erosion rates to vegetation change in southern Arizona. Geomorphology 14:311–317

Parsons AJ, Wainwright J, Abrahams AD (1996b) Runoff and erosion on semi-arid hillslopes. In: Anderson MG, Brooks SM (eds) Advances in hillslope processes. Wiley, Chichester, pp 1061–1078

Peterjohn WT, Schlesinger WH (1990) Nitrogen loss from deserts in the southwestern United States. Biogeochemistry 10:67–79

Peterson HV (1950) The problem of gullying in western valleys. In: Trask PD (ed) Applied sedimentation. Wiley, New York, pp 407–434

Pockman WT, Sperry JS (2000) Vulnerability to xylem cavitation and the distribution of Sonoran desert vegetation. Am J Bot 87:1287–1299

Ravi S, D'Odorico P (2009) Post-fire resource redistribution and fertility island dynamics: a modeling approach. Landsc Ecol 24:325–335. doi:10.1007/s10980-008-9307-7

Ravi S, Breshears DD, Huxman TE, D'Odorico P (2010) Interactions between geomorphic processes and vegetation at desert margins: implications for desertification. Geomorphology 116:236–245

Ravi S, D'Odorico P, Herbert B, Zobeck TM, Over TM (2006) Enhancement of wind erosion by fire-induced water repellency. Water Resour Res 42, W11422

Ravi S, D'Odorico P, Okin GS (2007a) Hydrologic and aeolian controls on vegetation patterns in arid landscapes. Geophys Res Lett 34, L24S23

Ravi S, D'Odorico P, Zobeck TM, Over TM, Collins SL (2007b) Feedbacks between fires and wind erosion in heterogeneous arid lands. J Geophys Res 112, G04007

Ravi S, D'Odorico P, Wang L, Collins SL (2008) Form and function of grass ring patterns in arid grasslands: the role of abiotic controls. Oecologia 158:545–555

Ravi S, D'Odorico P, Collins SL, Huxman TE (2009a) Can biological invasions induce desertification? New Phytol 181:512–515

Ravi S, D'Odorico P, Wang L, White CS, Okin GS, Macko SA, Collins SL (2009b) Post-fire resource redistribution in desert grasslands: a possible negative feedback on land degradation. Ecosystems 12:434–444

Ravi S, D'Odorico P, Breshears DD, Field JP, Goudie AS, Huxman TE, Li J, Okin GS, Swap RJ, Thomas AD, Van Pelt S, Whicker JJ, Zobeck TM (2011) Aeolian processes and the biosphere. Rev Geophys 49:RG3001

Reid KD, Wilcox BP, Breshears DD, MacDonald L (1999) Runoff and erosion in a piñon-juniper woodland: influence of vegetation patches. Soil Sci Soc Am J 63:1968–1979

Reiley DK, Breshears DD, Zedler PH, Ebinger MH, Meyer CW (2010) Soil carbon heterogeneity in pinon-juniper woodland patches: effect of woody plant variation on neighboring inter-canopies is not detectable. J Arid Environ 74:239–246

Reynolds JF, Virginia RA, Kemp PR, de Soyza AG, Tremmel DC (1999) Impact of drought on desert shrubs: effects of seasonality and degree of resource island development. Ecol Monogr 69:69–106

Sankey JB, Germino MJ, Glenn NF (2009) Aeolian sediment transport following wildfire in sagebrush steppe. J Arid Environ 73:912–919

Sayre N (1999) The cattle boom in southern Arizona: towards a critical political ecology. J Southwest 41:239–271

Scheffer M, Carpenter S, Foley JA, Folke C, Walker B (2001) Catastrophic shifts in ecosystems. Nature 413:591–596

Schlesinger WH, Reynolds JF, Cunningham GL, Huenneke LF, Jarrell WM, Virginia RA, Whitford WG (1990) Biological feedbacks in global desertification. Science 147:1043–1048

Schlesinger WH, Raikes JA, Hartley AE, Cross AF (1996) On the spatial pattern of soil nutrients in desert ecosystems. Ecology 77:364–374

Schlesinger WH, Abrahams AD, Parsons AJ, Wainwright J (1999) Nutrient losses in runoff from grassland and shrubland habitats in southern New Mexico: I. Rainfall simulation experiments. Biogeochemistry 45:21–34

Seager R, Ting M, Held I, Kushnir Y, Lu J, Vecchi G, Huang H, Harnik N, Leetmaa A, Lau N, Li C, Velex J, Nail N (2007) Model projections of an imminent transition to a more arid climate in southwest Northern America. Science 316:1181–1184

Sharifi MR, Meizner CF, Nilsen ET, Rundel PW, Virginia RA, Jarrell WM, Herman DJ, Clark PC (1988) Effect of manipulation of water and nitrogen supplies on the quantitative phenology of Larrea tridentata (creosote bush) in the Sonoran Desert of California. Am J Bot 75:1163–1174

Spaulding WG (1983) Late Wisconsin macrofossil records of desert vegetation in the American Southwest. Quat Res 19:256–264

Stewart J, Parsons AJ, Wainwright J, Okin GS, Bestelmeyer BT, Fredrickson EL, Schlesinger WH (in press) Modelling emergent patterns of dynamic desert ecosystems. Ecol Monogr

Thomey ML, Collins SL, Vargas R, Johnson JE, Brown RF, Natvig DO, Friggens MT (2011) Effect of precipitation variability on net primary production and soil respiration in a Chihuahuan Desert grassland. Glob Change Biol 17:1505–1515

Titus JH, Nowak RS, Smith SD (2002) Soil resource heterogeneity in the Mojave Desert. J Arid Environ 52:269–292

Turnbull L, Parsons AJ, Wainwright J (in review) Rainfall variability and runoff responses in creosotebush shrubland. J Arid Environ

Turnbull L, Wainwright J, Brazier RE (2010a) Changes in biotic and abiotic components of ecosystem structure over a transition from semi-arid grassland to shrubland in South-Western USA. Ecosystems 13:1239–1255

Turnbull L, Wainwright J, Brazier RE (2010b) Changes in hydrology and erosion over a transition from grassland to shrubland. Hydrol Process 24:393–414

Turnbull L, Wainwright J, Brazier RE (2011) Nutrient dynamics during runoff events over a transition from grassland to shrubland in south-western USA. Hydrol Process 25:1–17

Turnbull L, Wilcox BP, Belnap J, Ravi S, D'Odorico P, Childers DL, Gwenzi W, Okin GS, Wainwright J, Caylor KK, Sankey T (2012) Understanding the role of ecohydrological feedbacks in ecosystem-state change in drylands. Ecohydrology. doi:10.1002/eco.265

UNEP (1992) World atlas of desertification. UNEP/Edward Arnold, Nairobi/London, 69 Plates

Van Auken OW (2000) Shrub invasions of North American grasslands. Ann Rev Ecol Syst 31:197–215

Van Wilgen BW, Trollope WSW, Biggs HC, Potgieter ALF, Brockett BH (2003) Fire as a driver of ecosystem variability. In: Du Toit JT, Rogers KH, Biggs HC (eds) The Kruger experience: ecology and management of savanna heterogeneity. Island Press, Washington, DC, pp 149–170

Visser SM, Sterk G, Ribolzi O (2004) Techniques for simultaneous quantification of wind and water erosion in semi-arid regions. J Arid Environ 59:699–717

Wainwright J (2005) Climate and climatological variations in the Jornada Experimental Range and neighbouring areas of the US Southwest. Adv Environ Monit Model 2:39–110

Wainwright J, Parsons AJ (2002) The effect of temporal variations in rainfall on scale dependency in runoff coefficients. Water Resour Res 38:1271. doi:10.1029/2000WR000188

Wainwright J, Parsons AJ, Abrahams AD (2000) Plot-scale studies of vegetation, overland flow and erosion interactions: case studies from Arizona and New Mexico. Hydrol Process 14:2921–2943

Welter JR, Fisher SG, Grimm NB (2005) Nitrogen transport and retention in an aridland watershed: influence of storm characteristics on terrestrial-aquatic linkages. Biogeochemistry 76:421–440

Whicker JJ, Breshears DD, Wasiolek PT, Kirchner KB, Tavani RA, Schoep DA, Rodgers JC (2002) Temporal and spatial variation of episodic wind erosion in unburned and burned semiarid shrubland. J Environ Qual 31:599–612

Whicker JJ, Pinder JE III, Breshears DD (2008) Thinning semiarid forests amplifies wind erosion comparably to wildfire: implications for restoration and soil stability. J Arid Environ 72:494–508

Whitford WG, Anderson J, Rice PM (1997) Stemflow contribution to the ‘fertile island’ effect in creosotebush, *Larrea tridentata*. J Arid Environ 35:451–457

Wilcox BP, Thurow TL (2006) Emerging issues in rangeland ecohydrology: vegetation change and the water cycle. Rangel Ecol Manage 59:220–224

Wilcox BP, Newman BD, Brandes D, Davenport DW, Reid K (1997) Runoff from a semiarid ponderosa pine hillslope in New Mexico. Water Resour Res 33:2301–2314

Wilcox BP, Turnbull L, Young MH, Williams J, Ravi S, Seyfried MS, Bowling DR, Scott RS, Germino M, Caldwell T, Wainwright J (2012) Ecohydrological consequences of grasses invading shrublands: comparing cold and warm deserts. Ecohydrology. doi:10.1002/eco.247

Yao J, Peters DPC, Havstad KM, Gibbens RP, Herrick JE (2006) Multi-scale factors and long-term responses of Chihuahuan Desert grasses to drought. Landsc Ecol 21:1217–1231

Zhou CB, Breshears DD, Newman BD, Wilcox BP, Gard MO, Rich PM (2008) Soil water dynamics under low-versus high-ponderosa pine tree density: ecohydrological functioning and restoration implications. Ecohydrology 1:309–315