

Chapter 2

The Study of Land Degradation in Drylands: State of the Art

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Abstract Land degradation is difficult to define because land can only be considered degraded with respect to some use to which it may be put. However, physical and biological properties of the landscape are typically measured to characterize degradation rather than its inherent or potential utility. One approach to characterizing land degradation is by assessing the provisioning of ecosystem services. Most provisioning ecosystem services depend on water, and water management is crucial to maintaining and increasing ecosystem services in arid lands. In contrast, vegetation change has been most commonly employed as an indicator of land degradation. Nevertheless, the close relationship that exists between vegetation and other biophysical processes of the environment means that any change in vegetation will result in a concomitant change to these other processes also. Of particular importance is a change in vegetation distribution since the spatial distribution of associated biophysical parameters controls landscape fluxes, and

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hence degradation, by controlling landscape connectivity. From a management perspective, an understanding of the degree of connectivity in a landscape can aid in triage of remediation efforts. Areas that are dominated by long connected pathways will not respond to localized, small-scale manipulations because those pathways present inertia that a small-scale manipulation cannot overcome. Two important ecosystem services provided by drylands are grazing land and agricultural land. Both land uses can be drivers of degradation. The role of grazing in land degradation depends on several factors which can be grouped into three categories: number of animals, kind of animal species and grazing system. For agriculture, systematic crop residue removal without fertilisation, poor cultivation practices and extensive soil salinization are examples of mismanagement that may lead to land degradation. Aside from the immediate provisioning of food, drylands provide ecosystem services at a broader scale. Drylands are highly significant to the global carbon cycle. Land degradation in drylands has implications for the effectiveness of carbon sequestration as well as for storage (through soil erosion). Because many dryland soils have been degraded they are currently far from saturated with carbon and as a result their potential to sequester carbon may be highly significant. To understand land degradation better, efforts have been made to develop integrated human-environment research that overcomes the perceived deficiencies of reductionist, discipline-based research. However, much integrated environmental research to-date has resulted in a 'hierarchical relationship' between the human and physical components. Three approaches have been advocated to improve human-environment understanding: (a) systems science that emphasises feedbacks between integrated human and natural systems; (b) computer-simulation modelling that explicitly represents the interaction of individual human decisions

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and physical processes; and (c) participatory research that emphasises engagement with the actors in the region being studied. However, many questions remain open, and advancing beyond narrow scientific disciplinary specialization is vital if the hierarchical relationship in understanding physical and social causes of land degradation is to be broken.

2.1 Introduction: What Is Land Degradation?

Land degradation is difficult to define. At the ESF workshop on which this book is based, the participants were each asked to write down their understanding of the term. The result is presented in Fig. 2.1 as a mind map. What is evident from this mind map is that, in common with previous work (in particular the Millennium Ecosystem Assessment 2005), most associations with land degradation address issues of the use to which land may be put, especially the relationship of land to its utility for humans.

At the heart of the notion of land degradation lays a fundamental contradiction. “Degradation” is a value judgement; land can only be considered degraded with respect to some use to which it may be put, either in the past or potentially in the future (so-called, ecosystem services – see Sect. 2.2). However, in order to characterize how degraded land is, it is the physical and biological properties of the landscape that are typically measured rather than the inherent or potential utility. At best, in the case of current ecosystem services, the link between those physical and biological properties and the chosen land use is not wholly clear. In the case of a potential, but unknown future use there is no apparent link between current biophysical properties and a future assessment of ecosystem services. Inherently, biophysical measures of degradation can only apply to the current value of the land, and is incommensurate with the broader concept of ecosystem services. However, this problem is widely ignored in the discussion about land degradation and ecosystem services (Sect. 2.2).

In this chapter, first we examine the notion of land degradation in relation to ecosystem services. Next, we review indicators that are used to assess land degradation, and consider how these diverse indicators have been integrated through concepts of connectivity, before moving on to investigate, the roles of agriculture and pastoralism in causing land degradation in drylands. Finally, we take a broader perspective. The issues surrounding the links between specific ecosystem services and biophysical measures of land degradation are only part of the story. Land degradation has wider ramifications, and these ramifications justify the focus on describing land degradation in terms of biophysical properties of the land rather than the narrower utilitarian perspective. We consider such wider ramifications through the example of the specific issue of carbon dynamics. The chapter closes with two considerations of wider perspectives on the nature of biophysical and social processes in land degradation.

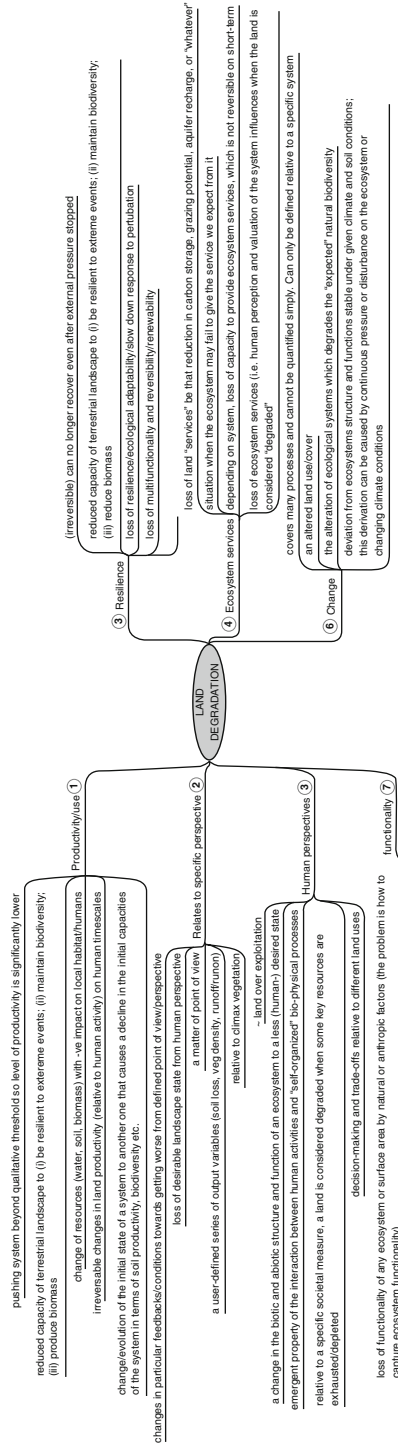


Fig. 2.1 Mind map of the concept of land degradation

2.2 Ecosystem Services and Land Degradation

Ecosystem services are commonly considered to be functions of or processes in ecosystems valued by humans – “things that ecosystems provide that matter to people” (Millennium Ecosystem Assessment 2005, p. 5). In drylands, the main provisioning ecosystem services (i.e. the production of forage, crops, fibre, fuelwood, and biochemicals) are dependent on biological productivity and water availability, which in turn are dependent on supporting services, such as soil development, primary productivity and nutrient cycling (Millennium Ecosystem Assessment 2005, p. 5; Havstad et al. 2007). Ecosystem services are considered to be diminished by land degradation. It is one view that land degradation is synonymous with loss of ecosystem services. The Millennium Ecosystem Assessment (2005, p. 5) defines land degradation as “a persistent reduction of biological and economic productivity” – thus the reduction of ecosystem services. The very reason for calling land ‘degraded’ is that it does not deliver ecosystem services to a level expected by the human beneficiaries of the ecosystem services. Therefore, ‘land degradation’ is influenced by the human expectations regarding ecosystem performance, and perceptions of the state of any ecosystem in terms of degradation can differ between groups with different experience and knowledge of the system (Davis 2005; Roba and Oba 2009). Thus, any discussion on degradation should involve a discussion on desirable ecosystem services and the parts of the ecosystem expected to deliver those them (Riginos and Herrick 2010).

To benefit from ecosystem services in arid lands, humans have to learn to cope with the special dynamics of ecosystems with scarce water resources variability and low predictability (Reynolds et al. 2007a, b). Societies that depend on the use of ecosystem services have usually developed strategies to deal with these characteristics of arid systems that resemble those of desert plants (for a brief review of these comparisons see e.g. Schwinning and Sala 2004), for example, by developing sophisticated storage mechanisms, by spatial and temporal flexibility as to what activities are carried out when and where (often associated with complex mechanisms to build and use expert [elders’] knowledge). One example of such an adaptation is pastoralism, a word that has many meanings, but can be defined as a ‘finely-honed . . . relationship between local ecology, domesticated livestock and people in rangelands, particularly in resource-scarce and ecologically variable regions, often at the threshold of human survival’ (Nori et al. 2005, p. 5). Although there is considerable debate on the sustainability of pastoralism under current environmental and social pressures, there is no doubt that the intricate system of using variable natural resources for livestock production developed by pastoralists could be used as part of a wider strategy to reach development goals in arid lands (Davis 2005; Hagmann and Ifejika Speranza 2010). Other examples are sophisticated irrigation strategies and communal water sharing plans developed by traditional societies, such as the North American Hohokam or the Medieval Muslim society on the Iberian peninsula (San José 2005; Nelson et al. 2010).

Water management is crucial to maintaining and increasing ecosystem services in arid lands. Effective science-based water management implies three aspects: (1) strengthening the knowledge base of when, where, and how much water will be received in the form of precipitation, (2) optimizing the partitioning of this water between infiltration, runoff, evaporation and transpiration, and (3) storing water for dry periods. Considerable spatial and temporal variation in those three aspects exists between landscape units because of their differences in geologic substrate, vegetation, soil, topography and past land use (Bestelmeyer et al. 2009). Suboptimal partitioning of water due to mismanagement can lead to enhanced soil erosion and degradation of provisioning services (Huber-Sannwald et al. 2006). Early recognition of the fundamental change caused by different partitioning of water in different vegetation types led to the proposition to classify arid lands based on how plants partition water (Le Houérou 1984). While a Western scientific approach is not the only approach to optimal management of arid lands (Bryant and Wilson 1998), it is a challenge to scientific research in arid lands to provide populations in these areas with the necessary information to optimize prediction, partitioning and storage of water. For prediction, both climatic forecasts and close observation of the state of vegetation and soil in arid lands is necessary. Land managers themselves should be strongly involved in the assessment and monitoring of their land to enable them to forecast changes in ecosystem (Bestelmeyer et al. 2009). Some indicators have been identified that can at least assist with this task (Roba and Oba 2009; Riginos and Herrick 2010). For partitioning it is important to understand the hydrology of arid lands and all factors influencing it. Great progress on this has been made and is reported in this book. For storage, long-term planning and landscape landscape-level thinking is necessary.

Water management in arid lands is designed to meet different societal needs and often conflicts arise between different social groups regarding their priorities for water use and water-dependent ecosystem services (Warner et al. 2006; Havstad et al. 2007). For example, the use of water and the ploughing of low-lying areas for crops can negatively impact livestock or fish production in a given region (Hagmann and Ifejika Speranza 2010). Not only can conflicts arise between different provisioning services such as crop and livestock production, but also because regulating services such as carbon sequestration, dust control, flood prevention, sanitation, amenity, and climate regulation in urban environments are gaining in importance in arid lands and may compete for water and land resources (Havstad et al. 2007). As a matter of fact, because of the increase in human populations in arid lands, projects to alleviate poverty in these marginal areas often include the development of such alternative ecosystem services for economic gain of the local population. Such projects further enhance the need for optimizing water and land allocation to meet the needs of all groups involved (i.e. the need for better local governance), but they also highlight the need for further scientific and (appropriate) technological advances to enhance ecosystem services (Warner et al. 2006).

While ecosystem management is dependent on societal factors, in order to reach the societally defined best outcome of ecosystem management, all elements of ecosystems have to be considered when optimizing ecosystem services (e.g. infrastructure (houses, roads), livestock and other animals, vegetation) and their interactions can be crucial to explaining outcomes of ecosystem management (Bestelmeyer et al. 2009). Land management influences the provision of ecosystem services through its interaction with biophysical processes (Huber-Sannwald et al. 2006). For example, movement of livestock can significantly contribute to seed dispersal and thus influence vegetation pattern, which in turn can influence water dynamics. Research has demonstrated the link between ecosystem dynamics and provision of ecosystem services and highlighted feedbacks in the system (e.g. Schlesinger et al. 1990; Lawrence et al. 2007). The latter are important because it has been recognized that dryland ecosystems can exhibit threshold dynamics that can severely degrade the ability of the ecosystem to provide expected services (Bestelmeyer et al. 2009). Such threshold dynamics can occur due to positive feedbacks at the patch scale (Schlesinger et al. 1990), but they can also occur due to interactions across scales (Peters et al. 2007). In order to make predictions for ecosystem-service delivery, the scale at which dynamics are investigated should comprise all parts of the ecosystem that are dynamically linked. Because of cross-scale linkages there is a great danger of erroneous conclusions based on investigations of ecosystem dynamics in only a subset of the ecosystem (Peters et al. 2006; Brauman et al. 2007; Bestelmeyer et al. 2009). Indeed, even global-scale processes may sometimes have to be considered to fully understand ecosystem dynamics (Loik et al. 2004, see also Chaps. 3 and 6).

Aggravation of poverty in arid lands is expected if ecosystem services cannot be enhanced or maintained in these areas and their degradation be halted. Ecosystem services in drylands are clearly under major threat from a combined range of pressures including climate change, privatization of natural resources (in particular land and water), and population growth. In addition to these factors, challenges in managing ecosystem services in drylands arise from the great variability of ecosystem dynamics in these areas, with many processes linked to pulse events (Schwinning and Sala 2004; Reynolds et al. 2007a, b). It is important in order to attain sustainability in ecosystem services to prepare for such events and increase resilience of the socio-ecological system (Gunderson 2000). Resilience (see Sect. 3.2) of dryland ecosystems is not only dependent on the biophysical environment, but also on the economic and societal development of the human populations dependent on arid land ecosystem services (Lawrence et al. 2007; Reynolds et al. 2007a, b). It is a challenge for both environmental and sociological research to put in place the necessary support system to build resilience in these systems that are often already in a state of stress and/or degradation (Bestelmeyer et al. 2009). Addressing this challenge requires not only increased understanding of complex socio-ecological systems, but also effective communication between stakeholders (Reynolds et al. 2007a, b; Roba and Oba 2009).

2.3 Indicators of Land Degradation: Physical, Chemical and Biological Parameters

Notwithstanding the crucial social context of land degradation, attempts to quantify it have been made in terms of the physical, chemical and biological parameters of the landscape. Not surprisingly, therefore, in a world of varied social contexts and different requirements for ecosystem services, such indicators of land degradation cannot have universal application. A range of indicators has been used to identify land degradation in drylands, but of these it is probably vegetation that has been most commonly employed. Two reasons may be advanced as to why this is so. First, vegetation change is the most easily observable change in the landscape. Even without quantification, cover and species change can be readily identified. Secondly, the dominant land use of the majority of drylands is grazing for domesticated animals. A reduction in vegetation cover or of palatable species results in a reduction in related ecosystem services. One of the most common vegetation changes in land degradation that has been identified in many areas of the world is a transition from grassland to shrubland, and has been documented in areas such as the Mediterranean (Martinez-Mena et al. 1999; Bochet et al. 1999; Maestre and Cortina 2002), the American southwest (Buffington and Herbel 1965; Gibbens et al. 2005) and in many parts in Africa (Kraaij and Milton 2006; Dean et al. 1995; Hoffman et al. 1999). However, even in terms of grazing, it cannot always be assumed that shrublands represent poorer ecosystem services than grasslands. In the Upper Karoo of South Africa, for example, there are instances of unpalatable grasses replacing palatable shrubs.

Because of the close relationship that exists between vegetation and other biophysical processes of the environment, any change in vegetation will result in a concomitant change to these processes also. However, a simple cause-effect relationship whereby vegetation change causes changes to other biophysical processes is to be eschewed (Thornes 1985; Wainwright and Parsons 2010). Although climate change or overgrazing are often regarded as drivers of vegetation change, in reality there are feedbacks between vegetation and other environmental processes. These feedbacks are often positive, so that the triggering of land degradation may be as much to do with autogenic drivers as with allogenic ones. Land degradation is, therefore, characterized by many changes other than easily recognised vegetation change. One way or another, such changes are manifest through changes to spatial distributions.

One of the key concepts in characterising the change in spatial distributions of biophysical parameters accompanying vegetation change is that of Islands of Fertility (Charley and West 1975; Goldberg and Turner 1986; Schlesinger and Pilmanis 1998). The significance of the development of such Islands of Fertility for runoff and erosion is depicted in Fig. 2.2. This figure illustrates two of the significant changes that accompany vegetation change: increases in both runoff and erosion. In a series of experiments on large plots, Parsons et al. (1996a) demonstrated a

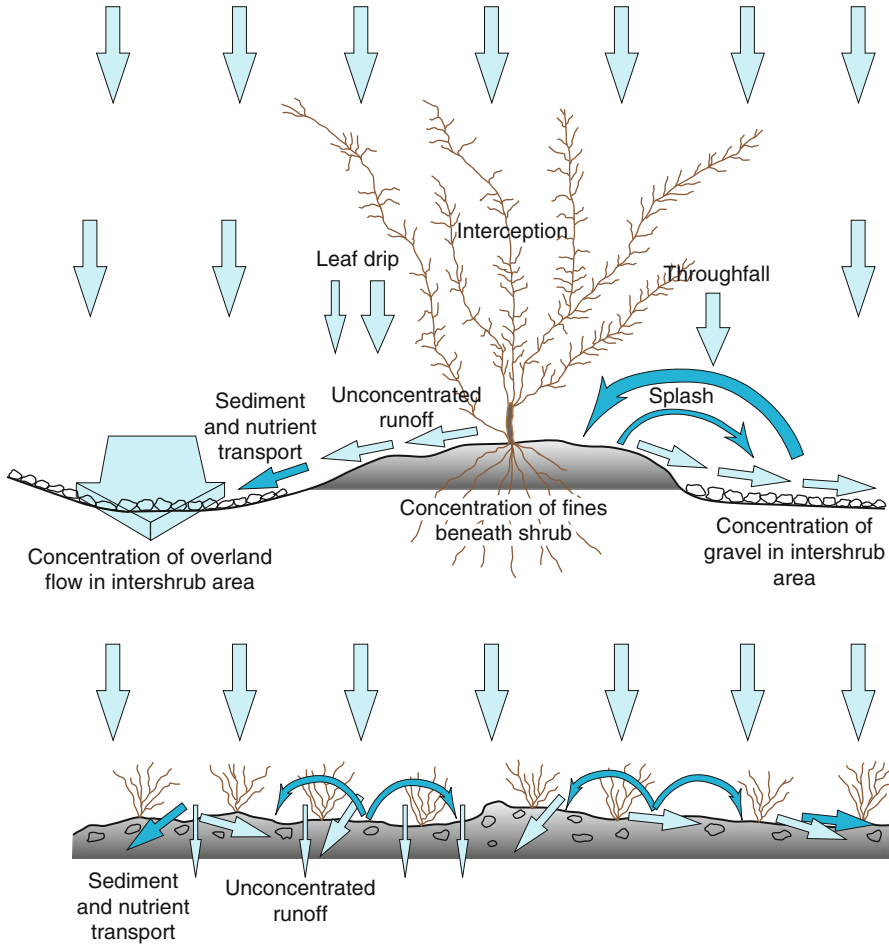


Fig. 2.2 Interactions between Islands of Fertility and water movements and erosion. Upper diagram shows runoff pattern in shrubland; lower diagram shows runoff pattern in grassland. (After Wainwright et al. 2000)

significantly larger runoff coefficient and approximately twice as much erosion on shrubland as on grassland (even though the grassland they considered was on a steeper slope than the shrubland). They attributed these differences to differences to the spatial distribution of runoff under the two plant communities that resulted from differences in the microtopography. Whereas the shrubland is characterized by a distinct across-slope topography in which the shrubs sit atop small mounds, the grassland exhibits no significant across-slope microtopography. In contrast, the grassland exhibits a distinct, stepped downslope microtopography, whereas the shrubland exhibits none. The consequence is a marked difference in runoff hydraulics, and consequently erosion (Parsons et al. 1996b). Turnbull et al. (2010a)

also found significantly higher erosion on shrub- compared to grass-dominated surfaces and suggested that these changes may also not proceed in a linear fashion as the shrubs invade.

Not only may be a change from grassland to shrubland promote an increased loss of soil, but it may also promote a change in nutrient loss, though such changes may also not be straightforward. Using rainfall simulation, Schlesinger et al. (1999) recorded less than half the loss of dissolved nitrogen in runoff from shrubland plots compared to grassland plots. However, from plots that recorded runoff from natural rainfall events approximately twice as much dissolved nitrogen was lost from shrubland plots (Schlesinger et al. 2000). Although the plots used for studying the runoff from natural events were four times the size of those used in the rainfall-simulation studies, it is doubtful that the explanation for the difference lies in scale differences. More likely, it is due to differences in rainfall intensity. Nevertheless, scale differences do affect measurements of nutrient loss, as shown by Brazier et al. (2007). Turnbull et al. (2010b) also studied nutrient (nitrogen and phosphorus) losses from plots under natural rainfall, and found that losses from shrubland were higher than those from grassland. Importantly, this study compared nutrient content of rainfall to that contained in the water and sediment leaving the plot, and showed that, whereas only 16 % of annual input was lost from grassland, 90 % was lost from shrubland, indicating that vegetation change can cause significant changes to nutrient cycling.

The previous examples use indices of land degradation that measure changes to fluxes of resource (be it water, soil or nutrients), but degradation may also be manifest through changes to *in situ* characteristics. In part, of course, such changes to *in situ* characteristics may be the straightforward result of these flux changes. Thus, Dickie and Parsons (2012) identified lower organic matter and plant-available phosphorus on shrublands compared to grasslands in the semi-arid Karoo of South Africa, and it is most probable that these differences result from greater fluxes of organic matter and phosphorus from the shrublands (cf Turnbull et al. 2010b, above). However, these authors also showed higher soil bulk density and lower soil shear strength on shrublands compared to grasslands. These differences may be interpreted both as secondary effects of the enhanced fluxes, but also as part of the positive feedback mechanisms. Both lower soil shear strength and lower organic matter content will favour increased erosion – especially rill erosion – and, hence, further land degradation (Parsons and Wainwright 2006). Dickie (2006) presented a conceptual model (Fig. 2.3) linking many of the physical, chemical and biological changes that may be associated with land degradation. At the heart of this model is the link between vegetation and the spatial distribution of nutrients. In essence, what this model indicates is that, though land degradation may be assessed through measuring a variety of parameters, it will not be understood without an understanding of the spatial distribution of these parameters and the drivers of changes to these spatial distributions. Mueller et al. (2007) argued that differences in the spatial distribution of biophysical parameters controls landscape fluxes, and hence degradation, by controlling landscape connectivity.

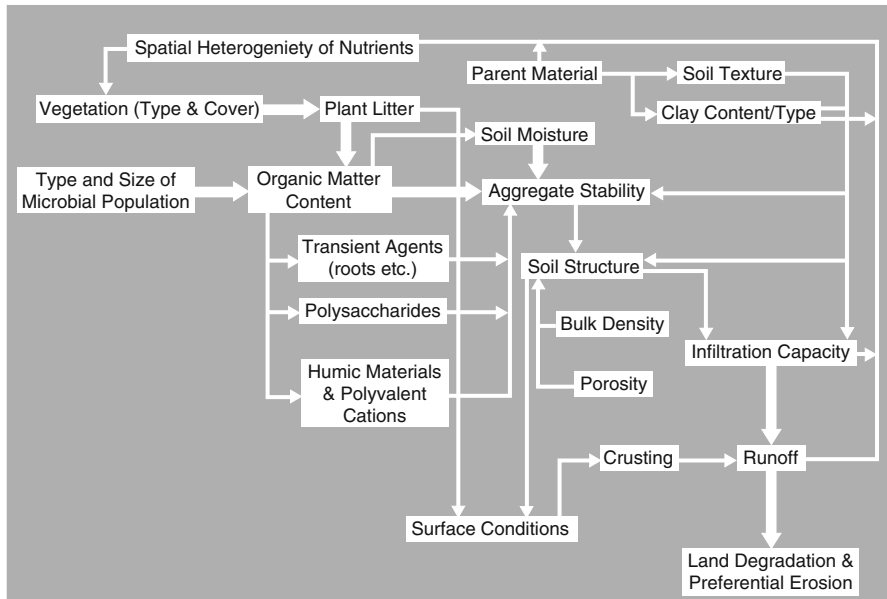


Fig. 2.3 A conceptual model depicting the linkages between soil physical and chemical indicators of land degradation (After Dickie 2006, reproduced by permission of the author)

2.4 Connectivity as an Integrative Measure of Land Degradation

Understanding connectivity between landscape compartments is pivotal in explaining spatial relationships, the behaviour of biophysical fluxes and associated trajectories of adjustment. These insights must be framed to appraise the sensitivity of differing parts of landscape to disturbance, any limiting factors or pressures that occur and the likely nature of cumulative off-site responses (Brierley et al. 2006). Hence, specific insights are required to predict likely landscape futures, recognizing differing forms and scales of (dis)connectivity. Analysis of the character and behaviour of landscape compartments, how they fit together (their assemblage and pattern) and the connectivity between them, provides a platform to interpret the operation of ecogeomorphic processes in any given system (e.g. Brunsdon and Thornes 1979; Caine and Swanson 1989; Lane and Richards 1997; Michaelides and Wainwright 2002; Bracken and Croke 2007).

Within an ecological context, landscape connectivity refers to the degree to which the landscape facilitates or impedes movement among resource patches. Landscape connectivity may be described in terms of structural connectivity, which is the extent to which landscape elements are contiguous or physically linked to one another, and functional connectivity, the linkage of habitat site by a process (Turnbull et al. 2008). Within a hydrological context, connectivity may be static and dynamic

(Bracken and Croke 2007). These authors' conceptualization of static hydrological connectivity refers to spatial patterns such as hydrological response units, while their dynamic representation refers to longer term landscape development and short-term variations in antecedent conditions and rainfall inputs to the system that result in nonlinearities in the hillslope and catchment response to rainfall. In terms of its hydrological connectivity, the overall hydrological behaviour of a system (used in this sense to describe how well runoff-producing areas interconnect to yield continuous flows, and thus cause erosion and redistribute sediment and nutrients) can also be considered in terms of functional connectivity. As with the ecological interpretation of structural and functional connectivity, in hydrology, it is the connectivity of structural attributes such as soil moisture that affects the functional connectivity of the landscape in terms of its ability to yield continuous flows (Mueller et al. 2007).

Ecosystem structure, and thus biotic and abiotic connectivity, evolves through time and space, determined by processes operating over a continuum of timescales. By drawing upon the notion of connectivity to transcend spatial and temporal scales, thereby avoiding transposition of scale errors, there is a recognition that structure and function at one scale is influenced by structure and function at other scales; thus a mechanistic interpretation of the behaviour of a system can only be derived by assessment of the extent to which ecosystem structure and function are connected through time and space. In this way, the structural connectivity of the landscape determines the propensity of the landscape to possess biotic and abiotic functional connectivity (but the degree of functional connectivity that arises from structural connectivity will be species/vector specific), which in turn modifies biotic and abiotic structural connectivity.

Dryland areas function as tightly coupled ecogeomorphological systems with strong feedbacks and interactions occurring across fine to coarse scales. In these areas, the interaction and feedbacks between climate, soils, vegetation and topography give rise to the emergence of distinct patterns of surface-water re-distribution and vegetation. The amount of water and sediments retained by the landscape is related to the surface connectivity between the upstream and downstream areas which depends on the (evolving) vegetation cover (Saco and Willgoose 2008). From the ecogeomorphological framework of land degradation, it is hypothesized that dynamics of land degradation are conceptualized by a cusp-catastrophe model (Turnbull et al. 2008) in which the two controlling variables are abiotic structural connectivity and abiotic functional connectivity, which implicitly account for ecosystem resilience, and biotic structural and function connectivity.

From a management perspective, an understanding of the degree of connectivity in a landscape can aid in triage of remediation efforts. Areas that are dominated by long connected pathways will not respond to localized, small-scale manipulations because those pathways present inertia that a small-scale manipulation cannot overcome. As a consequence, assessment and monitoring protocols must be sensitive to changes in connectivity at the management-unit scale to provide the information required to rapidly adapt management (Okin et al. 2009).

2.5 The Role of Grazing in Causing Land Degradation in Drylands

2.5.1 Grazing Process

Grazing is widespread in drylands, and has been identified as one of the main drivers of dryland degradation. It is a process by which energy and nutrients are transferred from the producers (plants) to the first level consumers (herbivores) (Fig. 2.4). It is a complex activity that involves several aspects. As animals graze they defoliate the plants thus affecting their growth and reproduction. At the same time, they trample and compact the soil, remove its plant cover and expose it to erosion, make trails, break up soil aggregates and cover the plant seeds that fall on the ground. In addition, as they move around searching for feed, they take nutrients from one place and deposit them as manure in another or redistribute propagules and other animals, mainly invertebrates, thus altering plant and animal diversity. All these aspects are closely related with animal behaviour since each grazing species has its different habits (Heady and Child 1994; Vavra and Ganskopp 1998).

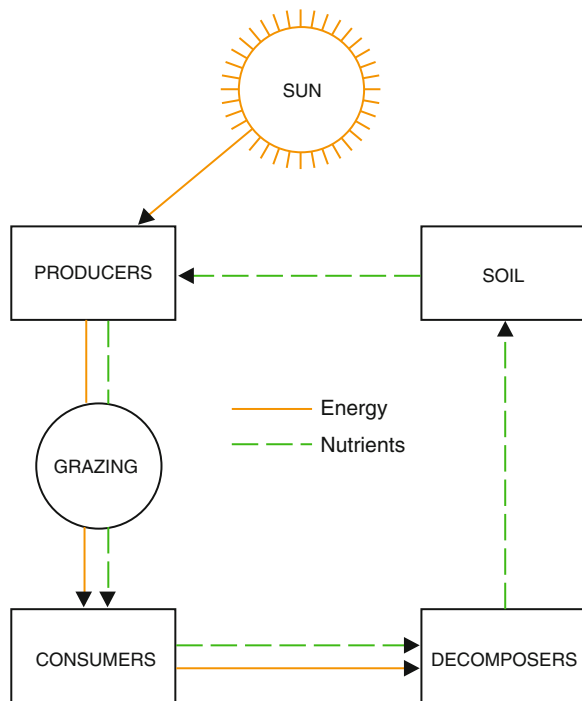


Fig. 2.4 The rangeland ecosystem

2.5.2 *Factors Affecting Grazing Pressures*

As a result of its complexity, the role of grazing on in land degradation depends on several factors which can be grouped into three categories: number of animals, kind of animal species and grazing system.

2.5.2.1 **Number of Animals**

The number of grazing animals and its relation with land degradation has been a highly debated issue in ecological literature. Two theories have been proposed. According to the equilibrium theory, vegetation changes are gradual because they are affected by more or less predictable patterns of rainfall (Scoones 1994) which suggests a tight coupling of plant and animals in the sense that animal numbers are regulated by the available forage, especially in key resource areas (Illius and O'Connor 1999). This theory implies that degradation is caused by livestock overgrazing, and therefore, is human-induced. The non-equilibrium theory, on the contrary, postulates that drylands are susceptible to high levels of climate variability (Ellis and Swift 1988). For this reason, plant, production is largely determined by rainfall and less so by animal numbers since die-offs during droughts keep animals below a theoretical maximum. Consequently, grazing does not cause irreversible changes through over-use of vegetation (Behnke and Scoones 1993; Scoones 1994). However, Sullivan and Rohde (2002) suggest not polarizing the terms “equilibrium” and “non-equilibrium” because they are conceptual ideals rather than identifiable states in nature. They argue that degradation related to overgrazing may exist but it should be attributed to political factors that intervene in grazing management, particularly in communally used rangelands. Also, Derry and Boone (2010) argue that this debate is founded on the semantics of terminology rather than on published evidence and suggest that finding mechanisms to cope with climatic variability is the real problem in drylands.

Overgrazing occurs when too many animals are trying to be fed on a limited supply of forage. It is considered as one of the most important factors causing degradation in drylands (Geist and Lambin 2004; Dregne 2002; Msangi 2007), mainly because overgrazing results in reduction of plant cover and, consequently, in exposing soil to erosion by wind or water. According to Thornes (2007), erosion increases as vegetation canopy cover falls and it becomes dramatic below 30 % cover (Elwell and Stocking 1976; Francis and Thornes 1990). However, the interaction of grazing, vegetation and erosion is more complex and can result in variable outcomes in space and time (Thornes 2005). The problem of erosion is enhanced by the fact that the soil surfaces and internal profiles are materially transformed by the compaction caused by an increased number of animals. In Mediterranean Europe in particular, soil erosion and degradation is further aggravated if overgrazing is combined with pastoral wildfires aiming at controlling the unpalatable to livestock plant species (Papanastasis 2004a). Nevertheless, overgrazing is often thought to be

over-emphasized as a cause of land degradation. It is evident around permanent settlements and watering points, where livestock concentrate, but much less in open areas where animal mobility is unrestricted. It has been argued that dryland ecosystems are ecologically resilient as long as some degree of livestock mobility is maintained (IUCN 2010). Also, there is not enough evidence to blame overgrazing for the widespread erosion in drylands in many parts of the world. According to Rowntree et al. (2004), the overgrazing hypothesis must be treated with caution. After reviewing two case studies from communal rangelands in South Africa, these authors came to the conclusion that stocking rates above notional carrying capacity (hence 'overgrazed') did not necessarily cause land degradation.

Undergrazing, i.e. grazing by fewer animals than the available forage in a rangeland, is also a potential cause of land degradation although it has not been much studied. This degradation occurs because the plant material not removed by animals becomes a very flammable fuel to be burned by wildfires, which in turn often leads to soil erosion (Papanastasis 2004b). If not burned, grasslands may be invaded by woody species thus resulting in loss of biodiversity (Papadimitriou et al. 2004). In the semi-arid savannah ecosystems, in particular, undergrazing results in an increase of the grass layer that promotes wildfires which damage the tree layer (van Langevelde et al. 2003). In the same ecosystems, Beerling and Osborne (2006) suggest that, apart from the forest loss, fires also accelerate C₄ grassland expansion and that each of them promotes drought and more fires.

2.5.2.2 Kind of Animal Species

The kind of animal species is another important factor affecting degradation in drylands. Various kinds of livestock have different preferences for forage species or groups of species. Cattle are considered as grazers in the sense that they prefer herbaceous to woody species. Sheep are intermediate feeders because they eat both herbaceous and woody species, although they greatly prefer the former than the latter. Goats are considered as browsers because they typically prefer and eat woody rather than herbaceous species (Holechek et al. 2004). The same is true for the wild animals, too. Giraffes, for example, are obligate browsers while elephants are both browsers and grazers (Beerling and Osborne 2006). As a result, their impact on grazing lands is different.

Overgrazing by sheep can result in completely denuded grasslands (Fig. 2.5). Sheep can move over steeper land and for longer distances than cattle, resulting in the creation of a dense network of trails in sloping grasslands. Goats have an affinity for the leaves and twigs of shrubs and trees and they are often blamed for the destruction of Mediterranean forests. In discussing the factors underlying deforestation in the Mediterranean countries, Thirwood (1981) considers goats as the main catastrophic agents for their predilection of woody forage. The arguments against these animals were so strong in the middle of the past century that several countries had to take decisive measures to reduce their numbers or even eliminate them completely by subsidizing their slaughter (FAO 1964). However, putting



Fig. 2.5 Semi-arid grassland heavily grazed by sheep in south-east Spain

the blame on goats for destroying the environment is confounded with mismanagement for which the sole responsibility is with people and not with animals (Papanastasis 1986). Nowadays, however, goats are considered important animals not only for utilizing poor quality vegetation but also for controlling the woody understorey of Mediterranean forests and reducing the fire hazard (Tsiouvaras 2000; Jauregui et al. 2009).

2.5.2.3 Grazing System

Grazing too many animals of the wrong kind in a particular rangeland oversimplifies the case. The effects of livestock grazing also depend very much on the type of grazing system employed. Grazing systems may be continuous or rotational. Continuous systems involve grazing of livestock throughout the rangeland for the whole growing period or the year. The system is usually applied in communal grazing lands where more farmers have the right to graze their animals in the same area. By contrast, rotational grazing systems involve the division of the rangeland into paddocks and their sequential use by animals over the growing season. Such a system is more common in privately owned pastures, where fences can be constructed. Both systems have advantages and disadvantages (Holechek et al. 2004). For example, continuous grazing is beneficial to the animals because they have the freedom to select the best plants or plant parts available in the pasture. For the rangeland itself, however, this system is detrimental because plants



Fig. 2.6 Watering point by overgrazed cattle in north-west South Africa

are not given a chance to recover. On the contrary, deferring grazing for some time during the growing period, grazed plants has the chance to recover thus avoiding vegetation degradation (Savory and Butterfield 1999). Nevertheless, no clear ecological advantages of the rotational over the rotational grazing were found by Briske et al. (2008) after reviewing the relevant literature.

On the other hand, the distribution of available infrastructure is very decisive on whether a rangeland will be degraded or not. Usually, animals tend to graze more intensively near the watering points (Fig. 2.6) and animal sheds than away from them resulting in degraded piospheres (Derry and Dougill 2008; Klintenberg and Verlinden 2008).

2.5.3 Grazing Policies

Degradation of drylands is often caused by implementation of wrong policies rather than by improper grazing. In several developing countries, policies have been implemented to convert the traditional mobile livestock production systems, which were communally managed, into semi-sedentary and privately managed ones including the building of enclosures. These policies have resulted in the

degradation of large areas in drylands due to the reduced mobility (Schwartz 2005). Also, small but productive buffer zones used by pastoralists were converted into irrigated lands or fenced off for wildlife and forest reserves, further aggravating rangeland degradation (IUCN 2010). European Union policies which paid subsidies per animal capita to farmers in the 1980s and early 1990s had a direct impact on livestock husbandry and grazing lands, too. In the Mediterranean countries, these policies resulted in further increasing livestock numbers, which in turn resulted in overgrazing and desertification of several dry and semi-dry areas (Lorent et al. 2008). In addition, subsidies for livestock production pushed farmers to replace many local animal breeds, which had been capable of using natural vegetation, with breeds that were more productive but less efficient in utilizing grazing lands thus resulting in their deterioration due to undergrazing. Most of these policies are usually implemented without the participation of the farmers. Taylor (2006) suggests that community-based grazing practices should be included in the formulation of grassland management policy. Without community participation it is unlikely that any technical intervention to halt degradation due to livestock grazing in drylands will succeed.

2.6 The Role of Agriculture in Causing Land Degradation in Drylands

At the beginning of the twenty-first century, rain-fed croplands overall were roughly 74 % (i.e. 1.1×10^9 ha) of the world cropland areas (1.5×10^9 ha), supplying 60–65 % of the world's food (Biradar et al. 2009). Croplands occupied 25 % of drylands (World Bank 2010), mostly in the dry-subhumid and semi-arid drylands and represented 44 % of all cultivated land (Safrield and Adeel 2005). An analysis of 23-years (1981–2003) of GIMMS NDVI data revealed a declining trend of land quality across some 24 % of the global land area, one fifth of which was cropland, which represent more than 20 % of all cultivated areas (Bai et al. 2008). However, the largest fraction (78 %) of the land area affected by soil degradation was located in humid regions of the globe; a much smaller fraction of global degradation occurred in dryland regions (Bai et al. 2008). The complex nature of the global food security system emerged dramatically during the financial crisis of 2006–2008, that revealed its fragility (McMichael and Schneider 2010) because of the interdependencies between the agricultural commodities production system (e.g. soybean, maize, wheat), concentrated in few geographical areas and often causing land degradation, the increasing frequency of climatic anomalies in these areas and the vulnerability to speculations of a globalized deregulated market (Lambin and Meifroidt 2011).

Degraded drylands are often located where overpopulation pressures, leading to unsustainable farming practices on land, can trigger the “classic collapse” in poor countries associated with the underdevelopment vicious circle. For instance,

systematic crop residue removal without fertilisation, which is frequent in poor countries, can cause negative soil nutrient balances and may trigger feedback loops leading to degradation of soil structure, accelerated erosion and a persistent decline reduction in soil fertility, which is the pre-condition for poverty in rural communities (Lal 2009). However, there are examples of extensive land degradation of dry croplands in developed countries that reveal that neither investment nor technical capacities of developed countries are sufficient to prevent or restore irreversible transitions. An example is the extensive soil salinization that occurred in the last decade on millions of hectares of arable drylands of southern Australia. Here, desertification dramatically broke out following a very slow but progressive saline watertable rise caused by the changes of the hydrologic balance generated by almost one century of cultivation of annual water-saving crops and pastures that replaced the native Eucalyptus forests upstream of huge catchments (Lambers 2003). This unexpected collapse came out in the affected regions a few decades ago and it can be interpreted as the unintended consequence of the collective narrow perspective of considering dry croplands as just a component of the commodity industry, regardless in particular of the long-term off-site environmental consequences, which are more difficult to reveal than on-site impacts and hence can lead to endless controversies on the adoption of mitigation measures. A decade of public and private investments and interventions in the saline areas proved to be ineffective and inefficient in mitigating the disasters (Pannell and Roberts 2010). This example reveals that the cost of intervention to reclaim the degraded soils to desirable conditions is amplified by the sudden changes or nonlinearities associated with thresholds that, if crossed, can cause the irreversible undesirable transition (Reynolds et al. 2007a, b). Such costs are not sustainable even for countries having high investment capacities.

Dryland agriculture plays a fundamental role in producing high quality grains such as wheat (Saint Pierre et al. 2008; Passioura and Angus 2009) and is characterized by intrinsic vulnerability to climatic change and land degradation, particularly in tropical areas (Kurukulasuriya et al. 2006). Water availability, soil fertility and agronomic management are key components of the complexity of dryland agricultural systems. From an agronomic perspective, drought is considered the main constraint to potential crop yield at a time scale of weeks to months, during the growing season (Passioura 2002, 2007). However, crop yields in drylands are also limited by other factors, with actual yields being often far below expectations if rainfall water was indeed the main limitation. While most irrigated world croplands are not far from 80 % of the theoretical yield potential, there are much higher margins for increasing actual crop productivity in drylands by reducing the yield gap between actual and potential water-limited yield, rather than seeking the improvement of the yield potential. This yield gap is caused by many factors such as poor rain distribution in space and time, untimeliness of operations (especially late sowing), weeds, poor establishment of seedlings, pest and diseases, inadequate nutrition, frost, heat, excessively acid topsoil and inhospitable subsoil for roots (Passioura and Angus 2009). It is possible to bridge this yield gap by replacing agronomic inputs with knowledge, such as with timely applications of fertilizers.

For example, the model-based benchmarking of the water-limited yield potential in Australian dryland agriculture helped farmers to monitor and manage crops better in a variable climate (Passioura and Angus 2010). Achieving the water-limited yield potential in drylands is not incompatible with improving soil fertility and carbon sequestration and can contribute to poverty alleviation, but there is no simple technical fix (Bekunda et al. 2010; Bossio et al. 2010).

Soil erosion by wind, water and tillage are major drivers of dryland degradation caused by agriculture in both rich and poor countries. Tillage erosion refers to the gradual soil translocation downhill caused by repeated intensive ploughing with mechanical or animal power without the action of water or wind and may significantly contribute to yearly erosion in hilly drylands (Nyssen et al. 2000, Blanco and Lal 2010). Effective modelling approaches have been developed to support minimization of tillage erosion (De Alba et al. 2006). However, the shift from mouldboard ploughing to conservative no-till or chiselling practices, requires start-up investments on equipment and increased use of chemical weed control, which are not accessible to smallholder farmers, particularly in developing countries. No-till practices is not without its problems, such as soil compaction, which can be best offset by periodical chiselling (Melero et al. 2011), increased use of chemical herbicides and other limitations for rainfed crops under specific soil conditions (De Sanctis et al. 2012).

The uncertainties about land degradation in drylands related to the impact of agricultural intensification or abandonment are outcomes of the interdependencies between agricultural and other socio-economic dynamics. In the past, conversion of forests to pastures or croplands was one of the main causes of high rates of water and wind erosion in many traditional agrarian systems, including southern Europe. In these areas, the abandonment of croplands in most marginal lands and the implementation of forest plantation programmes resulted in increased woody vegetation cover in recent decades (see Chap. 14). All these dynamics were always driven by socio-economic factors in marginal lands, such as population pressure and local or global dynamics of the industrial sector (García Ruiz 2010).

Agricultural intensification may lead to degradation or improvement of soil fertility in drylands. In dry croplands, soil erosion by water is often the consequence of a few intense rainfall events occurring on bare soil, e.g. at the end of the dry season or even in summer (De Santisteban et al. 2006). In many cases, both wind and water soil loss can be lowered to less than 10 % of that from a reference bare soil, with a soil cover >40 %. Crop-residue management, either by stubble mulch tillage or grazing management, is recognised to be a key practice for water- and wind-erosion control. However, crop yield in drylands is often insufficient to ensure effective standing residue to guarantee soil cover, either because of drought limitations, limited inputs, lack of adapted varieties to drought, climatic limitations that restrict protective cropping strategies or overexploitation of crop production because of insufficient production of fuel and food or forage (Peterson et al. 2006). Sometimes, even full crop cover may not be sufficient to prevent dramatic erosion

rates on slopes under heavy summer thunderstorms, particularly when crop canopy structure is not effective (e.g. sunflower, vineyards) in preventing raindrop splash and surface runoff (Roggero and Toderi 2002; García Ruiz 2010).

The effectiveness of trees in controlling soil erosion is well known, and in spite of their transpiration, trees in drylands need not necessarily reduce soil water storage more than the herbaceous vegetation. The practice of growing trees and crops in association in agroforestry systems is recognized worldwide as an integrated approach to sustainable land use (Nair et al. 2010). However, the implementation of large-scale adoption of agroforestry conservation practices in drylands is not always straightforward in areas with poor infrastructure and socio-economic development, because of the long-term nature of the projects and the difficulty of producing immediate benefits (Kelso and Jacobson 2011). Furthermore, when water is the limiting factor, the association or intercropping of tree and field crops may exacerbate the interspecific competition that could be mitigated with appropriate tree crop management (Bayala et al. 2012).

Cropland abandonment may also result in increased soil degradation. In Spanish drylands, Sauer and Ries (2008) found that an increase of vegetation cover in abandoned croplands does not necessarily lead to a decrease in soil erosion, unless cover increased over 60 %. A well designed surface drainage system, complemented with reduce tillage practices proved to be very important conservation measures both in croplands and abandoned marginal lands (De Santisteban et al. 2006). The abandonment of large-scale grazing systems is also creating new environmental issues in many southern European contexts, with loss of grassland biodiversity and increased hazards of natural or anthropogenic disasters such as fire (Caballero et al. 2009). The strategies to achieve the conservation objectives prescribed by the EU Habitats Directive to protect target agricultural habitats are still controversial, as little is known about what is really needed to maintain in an effective and efficient way unprofitable agricultural activities in the area being abandoned (see Chap. 14 for further discussion).

Further uncertainty about agricultural dryland degradation is associated with climate change and changes in climate variability, both of which will affect soil-erosion rates, probably adversely, even if socio-economic drivers may have greater impacts than any likely climatic change in most situations (Slaymaker 2001; Wainwright and Thornes 2004; O'Neal et al. 2005; Wilkinson and McElroy 2007; Nielsen and Reenberg 2010). Site-specific erosion vulnerability and risks related to land use or climatic changes are often assessed through mathematical models (Li et al. 2011) to support the understanding of the driving processes, the quantification of on-site and off-site impacts, the identification of options for mitigation and the assessment of the performance of soil conservation practices (Blanco-Canqui and Lal 2008). Such an approach is supporting the agri-environmental schemes in the EU (Montanarella 2007), where unrestrained production, vulnerable to land degradation, seems a thing of the past, and also because of the abandonment of marginal areas.

2.7 Land Degradation in Drylands and the Carbon Cycle

Although plant biomass per unit area is typically lower in drylands than many other terrestrial ecosystems (about $6 \text{ kg}\cdot\text{m}^{-2}$ as opposed to a typical level of $10\text{--}18 \text{ kg}\cdot\text{m}^{-2}$) their large spatial extent makes drylands highly significant to the global carbon cycle (Trumper et al. 2008). In particular, total dryland soil organic carbon reserves comprise 27 % of the global soil organic carbon reserves (Zafar et al. 2005). In addition, because many dryland soils have been degraded they are currently far from saturated with carbon and as a result their potential to sequester carbon may be highly significant (Farage et al. 2003).

2.7.1 Biotic Controls on the Terrestrial Carbon Pool in Drylands

The inputs, outputs and storage of carbon (Fig. 2.7) within the terrestrial pool determine whether drylands ecosystems act as carbon sources or sinks. Carbon enters the terrestrial ecosystem through a single process, photosynthesis (Trumbore 2006), can be stored in above/below ground plant biomass and as both organic and inorganic forms in soil and can leave the terrestrial ecosystem via a number of

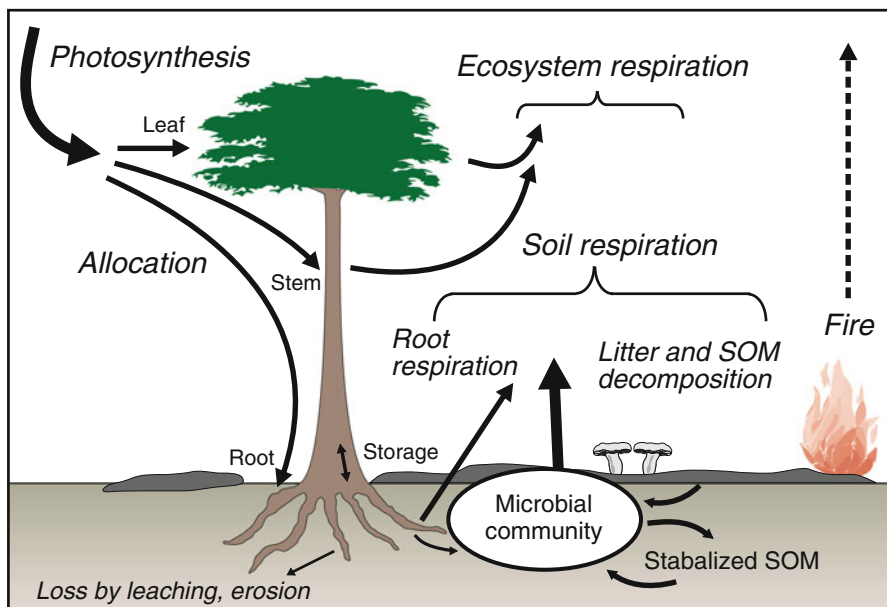


Fig. 2.7 Pathways of carbon flow through ecosystems (Reproduced from Trumbore (2006) with permission of John Wiley & Sons)

processes including fire, soil erosion, leaching and respiration. The loss of carbon from the ecosystem via respiration in itself covers a variety of processes including respiration from living plant, leaves, stems and stalks (autotrophic respiration) and carbon released during the decomposition of organic matter as a result of microbial activity (heterotrophic respiration; Trumbore 2006). The balance between inputs, storage and outputs, along with the processes and pathways involved is in turn controlled and influenced by a large number of interacting physical and biological factors. The assimilation of carbon into the environment as a result of photosynthesis is limited by environmental stresses such that the maximum rate of photosynthesis is rarely if ever achieved (Schulze 2005). If light is not limiting, as is often the case in drylands, then dry air or dry soil are limiting. If soils are wet, light is most likely limiting due to cloud cover. As a result of these limitations, many enzyme systems operate at about 50 % of their maximum performance in order to allow for a maximum range of regulation under variable conditions (Schulze 2005).

Photosynthesis rates in dryland environments are often limited by water availability, as is illustrated by a study of 19 desert shrubs (Comstock et al. 1988) in which results showed none of the species studied were capable of high rates of photosynthesis during the low water potentials characteristic of the dry season. In another study, Haase et al. (2000) found the rates of photosynthesis of *Anthyllis cytisoides* shrubs in semi-arid areas of south-eastern Spain to be limited by low plant-water status for most of the observation period. In contrast to shrubs subject to natural dry season conditions, which showed a drop in photosynthesis rates to nearly zero, artificially watered plants were able to maintain the maximum rate of photosynthesis (Haase et al. 2000). The resultant drop in carbon assimilation because of lower rates of photosynthesis during the dry periods can result in significant and highly variable inter-annual fluxes in carbon.

Vegetation in semi-arid environments often exhibits a number of adaptations to environmental conditions, which, in turn, result in changes to carbon pools. These adaptations include drought-deciduous plants such as the *Anthyllis cytisoides* *L* studied by Haase et al. (2000), which shed their leaves during dry periods, resulting in reduced water loss, but also reduced carbon assimilation. However, the growth of new leaves is a water-, energy- and time-consuming process, putting drought-deciduous plants at a disadvantage during the period immediately following rainfall when resources are most abundant (Whitford 2002). As a result of this disadvantage, most desert plants are not drought deciduous and alternative adaptations such as leaf morphology are more common (Whitford 2002).

Dryland plant species also exhibit three photosynthetic pathways (Fig. 2.8) which it has been argued is an adaptation to climatic variables (Ehleringer and Monson 1993). The most common and basic is C_3 photosynthesis (Fig. 2.8a), which follows the Calvin-Benson Cycle involves the addition of CO_2 to ribulose biphosphate, to form two molecules of a three carbon compound, phosphoglyceraldehyde (Whitford 2002). C_4 photosynthesis (Fig. 2.8b), hypothesised to be an evolutionary adaptation to low atmospheric CO_2 levels (Sage 2004; Osborne and Beerling 2006) involves the combining of CO_2 with phosphoenolpyruvate, to form four carbon molecules,

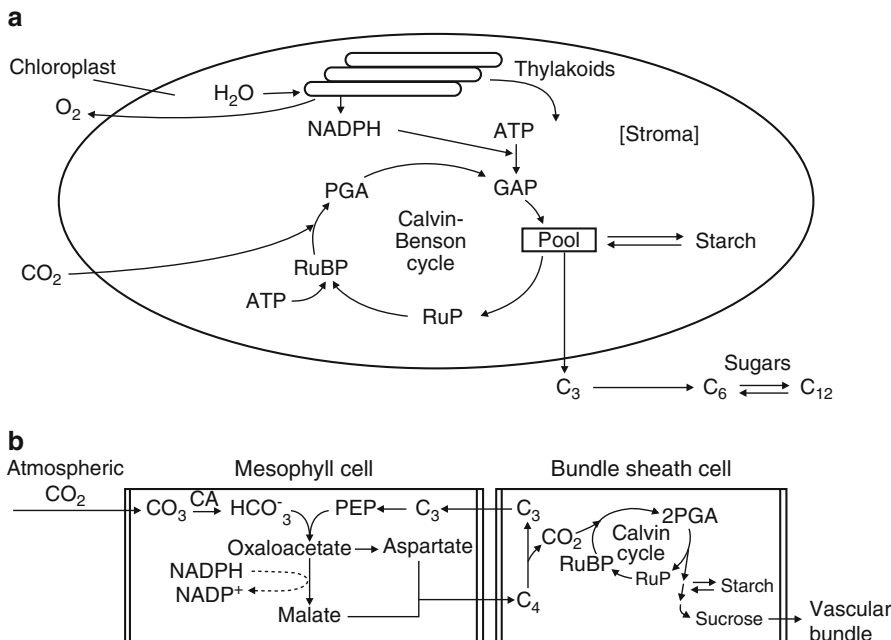


Fig. 2.8 Different photosynthesis pathways in **(a)** C_3 plants and **(b)** C_4 plants (Reproduced from Larcher 1995, with kind permission of Springer Science+Business Media)

Table 2.1 Comparisons of different photosynthetic pathways (Reproduced from Whitford 2002, with permission of Elsevier)

	C_3	C_4	CAM
Light saturation	$\frac{1}{4}$ full sunlight	Higher than full sunlight	Fixes C at night
Optimum temperature	Around 25 °C	Around 45 °C	Around 35 °C
Water loss rate (g H_2O g^{-1} fixed)	Around 600	Around 250	Around 50
Photosynthetic rate (mg CO_2 dm^{-2} leaf $^{-1}$ h $^{-1}$)	25	60	3

oxaloacetic acid or malate. Significantly for drylands, C_4 photosynthesis collects CO_2 in bundle sheath cells, allowing the plant to bind CO_2 at lower concentrations, reducing water loss in hot environments (Whitford 2002).

Finally the CAM (crassulacean acid metabolism) pathway, which is a modification of the C_4 pathway involving the temporal segregation of CO_2 assimilation and the Calvin Cycle only allows plants to open their stomata at night, reducing water loss per unit of carbon gained (Whitford 2002). Each photosynthetic pathway results in different environmental preferences and restrictions and changing rates of carbon uptake. A comparison of the general properties of C_3 , C_4 and CAM plants is illustrated in Table 2.1.

Land degradation in drylands can result in a shift in the balance of plants exhibiting these photosynthetic pathways. For example, the change from grassland to shrubland in the US Southwest is also a change from C_3 grassland to C_4 shrubland. Consequently, land degradation in drylands has implications for the effectiveness of carbon sequestration as well as for storage (through soil erosion).

2.7.2 *Abiotic Controls on the Terrestrial Carbon Pool in Drylands*

In addition to carbon fluxes as a result of the biological processes of photosynthesis and respiration, rapid loss of carbon from the terrestrial ecosystem can occur as a result of fire (Breshears et al. 2001; Tilman et al. 2000). The occurrence and frequency of wildfires is linked to climate and interacts with other factors such as topography, soil, herbivores, and amount of herbaceous fuel to determine the nature, density, and location of woody plants in a landscape (Van Auken 2000). However, many fires are caused by human activity, so these controls on fire frequency are only partial. Whilst above-ground woody biomass can represent a significant *sink* for atmospheric carbon (Houghton et al. 1999) it is also one that is extremely vulnerable to fire and is then a *source* of atmospheric carbon. Fire suppression in areas of the United States, for example, has been used to increase the storage of carbon in woody biomass. However, while this results in an increase in carbon storage, the increased forest density also dramatically increases the risk of catastrophic forest fire and the return of large quantities of carbon to the atmosphere (Breshears et al. 2001; Harden et al. 2000). Fire is especially significant as a source of carbon in water-limited drylands, where decomposition can be limited by drought (Trumbore 2006).

Carbon can also be either lost or redistributed as a result of leaching (Keller and Goldstein 1998) and erosion (Lal 2003), the latter being particularly prominent in degrading ecosystems. Leaching and erosion of carbon within a soil profile and across a landscape is influenced by many factors such as vegetation or stone pavement cover, soil erodibility, topography and soil infiltration rates (Lowery and Arriaga 2004). Being concentrated in the surface soil and of low density, soil organic matter, including soil carbon is preferentially removed by surface runoff and blowing wind (Lal 2005). The key stages of soil erosion along with potential losses of carbon at each stage are illustrated in Fig. 2.9.

Whilst acknowledging losses of terrestrial carbon as important on a century or millennial timescales, Trumbore (2006) dismisses losses of terrestrial carbon either through leaching of dissolved organic or inorganic carbon, or by erosion, as too small to be major contributors to inter-annual carbon balance. However, when drylands begin to degrade, rapid vegetation transitions occur, leading to a significant change in vegetation cover and increasing erosion rates (Turnbull et al. 2010a). Degrading drylands therefore export significant quantities of soil organic carbon which has significant implications for local and regional carbon pools in drylands (Brazier et al., in press).

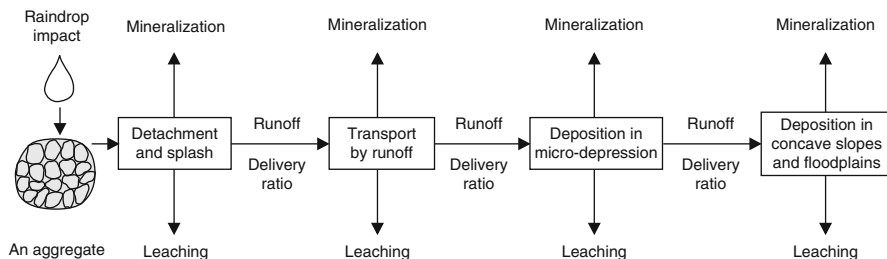


Fig. 2.9 Fate of eroded soil organic carbon (Reproduced from Lal 2005, with permission of Elsevier)

2.8 Carbon Dynamics in Dryland Agricultural Systems

Agriculture is considered one of the possible sources or sinks of atmospheric CO_2 . Soils globally represent some 30 % of the sink of the total atmospheric CO_2 increase, after the atmospheric and ocean sinks. Land-use changes are estimated to contribute about 20 % of global greenhouse gas (GHG) emissions at a rate of $1.5 \pm 0.5 \text{ Pg C year}^{-1}$, now almost exclusively from tropical Asia (where it is increasing) and American (where it is decreasing) regions (Canadell et al. 2007).

Dryland agriculture can be a soil carbon source or sink. The soil carbon sequestration in soils of arid climates occurs through both organic (SOC) and inorganic (SIC) compounds (calic horizon). The rate of SOC vs SIC sequestration is generally greater by one or two orders of magnitude (Lal 2009) but SIC has much longer residence time than SOC and net sinks are only for noncarbonate terrains (Feng et al. 2002). Wang et al. (2010) observed that the SIC sink significantly increased with increasing mean annual temperature and decreasing annual rainfall. Massive use of acidic N and P fertilisers and the use of lime or gypsum to buffer soil pH may also contribute to CO_2 emissions from carbonates (e.g. West and McBride 2005).

The SOC stock change is recognised to be a strong integrating factor between UNFCCC, UNCCD and CBD (Cowie et al. 2011). The SOC stock of drylands tend to be around half that of soils in moist environments in the same temperature regime (IPCC 2006 in Cowie et al. 2011). Because of land degradation, dry croplands are often carbon depleted, hence can be a potential sink for atmospheric CO_2 , but SOC sequestration rates are constrained by the water-limited inputs of organic carbon from crop residuals, the cost of nutrient balance for storing carbon (Lal 2004) and the faster SOC turnover under warm climate (Kimetu et al. 2009). In fact, it is uncommon to find successful and low cost experiences of long-term rehabilitation of semi-arid croplands (e.g. Hamza and Anderson 2010). The carbon-sink potential through land management is estimated at $0.3\text{--}0.5 \text{ Pg C year}^{-1}$ for desertification control (Lal et al. 1999) and $0.4\text{--}1.0 \text{ Pg C year}^{-1}$ for reclaiming salt affected soils (Lal 2010). The water limitation of dry croplands constrains SOC sequestration to a range of $0\text{--}200 \text{ kg ha}^{-1} \text{ year}^{-1}$, i.e. between 2/5 and 1/5 of the non-limited water croplands (Lal 2004).

The implementation of adaptation and mitigation strategies is particularly urgent in the Sahel region, where financial constraints are considered the trigger for a vicious circle of underdevelopment leading to environmental damages and human live losses (Sissoko et al. 2010). Here smallholder farmers are at the centre of the soil fertility restoration processes but they systematically fail to intensify agricultural production in a sustainable way, hence crop productivity is continuously declining. Nutrient depletion rates depend on site-specific conditions including environmental, institutional, social and economic constraints, like farm size, education, farming experience, land tenure, distance to markets, off-farm income, and access to credit and to technical knowledge (Bekunda et al. 2010).

From an agronomic perspective, improving SOC in dry croplands implies addressing water-limited crop yield potential. Because of the relatively low income per unit area of dryland cropping systems, agronomic options for sustainable dryland cropping systems must be low cost. For instance, timeliness of nitrogen fertilization to fit crop requirements in relation to phenology can contribute to maximize nitrogen-use efficiency with no or low additional costs (Passioura and Angus 2009). Cropping sequence, fallow and crop residue management, tillage methods and nutrient management are among the main drivers of the carbon balance in dryland agroecosystems (e.g. Suddick et al. 2010). However, some of the SOC sink benefits associated with these practices may be offset by the associated direct or indirect GHG emissions, as it may happen with fossil fuel consumption for fertilizer production and transport and with the increased N_2O emissions due to N fertilisation (IPCC 2007). For instance, the systematic use of organic amendments can slowly but significantly contribute, with long-lasting effects, to increase SOC sink in drylands and to trigger positive feedbacks on overall soil fertility and crop productivity (Diacono and Montemurro 2010). However, organic amendments are not free from N_2O emissions (e.g. Zaman et al. 2004; Alluvione et al. 2010), and further long-term field experiments are needed to quantify the energetic and emission efficiency of organic amendments *vs.* mineral fertilizers considering their life-cycle in dry croplands.

There are still uncertainties on the impact of global temperature and CO_2 rise on SOC dynamics under dryland farming (e.g. Lal 2001; Smith et al. 2008; Johnston et al. 2009). Recent findings on SOC dynamics have opened new spaces for both field and lab research on this topic. For instance, Kleber et al. (2011) explored the controversy about the temperature sensitivity of SOC decomposition in relation to SOC lability (Davidson and Janssens 2006; Fang et al. 2006; Conant et al. 2008) through the application of NEXAFS (Near edge X-ray Absorption Fine structure Spectroscopy) in combination with differential scanning calorimetry and alkaline cupric oxide oxidation. This new technique allows examination of SOC molecular characteristics without the analytical constraints that sometimes complicate classical methods of organic matter analyses. The same authors showed that the so-called SOC “passive” (i.e. stable) pools are not necessarily related to specific molecular structures or thermodynamic stability and that assuming that old carbon is composed

of complex recalcitrant compounds will erroneously attribute a greater temperature sensitivity to those materials. Sorptive protection, aggregation, occlusion and facultative non utilization seem to be the processes allowing labile seemingly materials to persist, rendering classical explanations like “humification pathways” obsolete (Kleber and Johnson 2010). The persistence of sunk SOC is therefore to be related to the interaction between substrates, microbial actors and abiotic driving factors, in which microbial ecology and the resource availability may play a key role in a given physical environment (Kleber et al. 2011). The implications of these new findings for dryland agriculture are still open to further research, from molecular to ecosystem scales (King 2011) to identify strategies to develop sustainable agronomic practices able to achieve the water-limited SOC potential for dry croplands (e.g. Suddick et al. 2010).

The expected impact of climate change and CO₂ rise on N₂ fixation, denitrification and leaching are also worth to be investigated further, to quantify the potential positive feedbacks between CO₂ rise and SOC sequestration (Esser et al. 2011). Implications about this topic are worth to be explored especially for rainfed cropping systems e.g. about the impact of different tillage methods (e.g. Baker et al. 2007; Angers and Eriksen-Hamel 2008), their interactions with expected climate change projections (e.g. Farina et al. 2011) or their interactions with biotic factors, such as weed cover during intercropping under no-till (De Sanctis et al. 2012) and the processes controlling microbial functions and soil heterotrophic respiration dynamics in drylands (e.g. Lagomarsino et al. 2011). Also agroforestry systems can provide an option for enhancing carbon sequestration, although their potential in drylands is substantially lower than in humid areas because of the exacerbated interspecific competition for water (Nair et al. 2009). However, there are a number of options that are worth to be explored further, in particular those based on the intercropping of food crops with multipurpose drought resistant shrubs such as *Jatropha curcas*, which is being increasingly used as a renewable energy source alternative to firewood in tropical drylands (Contran et al. 2013). Further promising opportunities for SOC sinks are also offered by new research findings and applications on biochar (Sohi et al. 2010; Baronti et al. 2010) and the role of both natural or manufactured nanoparticles in the control of the soil processes (Qafoku 2010).

The objective of covering the yield gap and mitigating land degradation in dryland agriculture is a complex challenge in which the improvement of soil fertility and carbon sequestration are often coupled with poverty alleviation and rural development (Bekunda et al. 2010; Thomas et al. 2012). Although numerous soil conservation practices in dryland agriculture are available, soil erosion remains a serious problem in many cases (Unger et al. 2006). Despite the continuous progress in studying land degradation, a robust and operative assessment tool is not yet accepted by the scientific community and this constrains political and social areas of intervention (Veròn et al. 2006).

2.9 Breaking the ‘Hierarchical Relationship’ in Understanding Physical and Social Causes of Land Degradation

Many environmental systems are now dominated by human activity and it is generally accepted that land degradation is the result of both physical and human factors (e.g. Millennium Ecosystem Assessment 2005; Muchena et al. 2005; Baartman et al. 2007; Reynolds et al. 2007a, b). To understand land degradation better, efforts have been made to develop integrated human-environment research that overcomes the perceived deficiencies of reductionist, discipline-based research (Newell et al. 2005). Despite its best intentions however, much integrated environmental research to-date has resulted in a ‘hierarchical relationship’ between the human and physical components, with one providing the initial and boundary conditions for the other to do its work (Demeritt 2009). In this all-too-frequent hierarchical mode of investigation, identification and examination of reciprocal feedbacks between physical processes and human activities are limited or absent. For example, scenarios of future human activity are often used as an input to physical models to explore alternative potential system states and futures (e.g. Bellot et al. 2001; Märker et al. 2008). Alternatively, knowledge of physical processes or forecasts of the future state of physical systems may be used as inputs to studies that examine potential impacts of physical change on policy choices or economic valuation of ecosystem services (e.g. Cohen et al. 2006; Lestrelin 2010).

In the remainder of this chapter we discuss three approaches that have been advocated to improve human-environment understanding. These approaches are: (a) systems science that emphasizes feedbacks between integrated human and natural systems (e.g. Liu et al. 2007b), (b) computer-simulation modelling that explicitly represents the interaction of individual human decisions and physical processes (e.g. Wainwright and Millington 2010), and (c) participatory research that emphasizes engagement with the actors in the region being studied (e.g. Demeritt 2005). Each has its own advantages to help break the ‘hierarchical relationship’ in understanding physical and social causes of land degradation.

With the implicit aim of overcoming the deficiencies of the hierarchical relationship, systems science approaches are being increasingly applied to examine Coupled Human And Natural Systems (CHANS, e.g. Liu et al. 2007b). The CHANS approach, for example, sets out to produce integrative frameworks for understanding human-nature interactions, with an emphasis on reciprocal interactions and feedbacks, and a focus on patterns and processes that link human and natural systems (Liu et al. 2007a). Some have questioned claims that systems science can produce reciprocally integrated research where other approaches have failed (e.g. Richards and Clifford 2008; Wainwright 2009), and it would be both naïve and arrogant to assume that the inherently quantitative approach of systems science can apply to all aspects of human-environment interactions. For example, Turner and Robbins

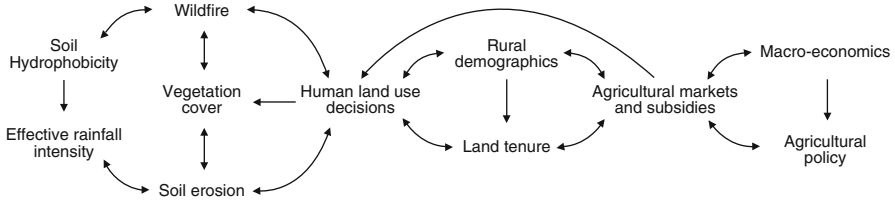


Fig. 2.10 Multiple process feedbacks exist between ecological, hydro-geomorphological and social processes. *Arrows* represent influences. Although effects of individual interactions may be well understood, reciprocal feedbacks – including human activity – are less well understood

(2008) highlight the commonalities and differences between the problem framings and explanatory perspectives of land-change science and political ecology for understanding human-environment dynamics. However, an emphasis on feedbacks between human activity and physical processes will be important for producing reciprocally integrated understanding about the physical and social causes of land degradation.

For example, in dryland areas future land-use patterns will likely vary depending on the strength of feedbacks between human activity and erosion/sedimentation processes (Verburg 2006). Furthermore, feedbacks between ecological, hydro-geomorphological and social processes can produce a complex web of interactions (e.g. Fig. 2.10). Important considerations for untangling this complexity are identifying whether feedbacks are positive, negative or produce qualitative change, how feedbacks vary in space and time, if thresholds and non-linearities exist, and the importance of initial conditions on feedbacks. Although individual effects may be well understood, how these effects cascade through other processes to create a reciprocal effect is more difficult to assess. For example, the effects of wildfire on soil hydrophobicity are well understood (Letey 2001; Shakesby and Doerr 2006), but how these effects feedback via soil erosion, vegetation change and human land-use decisions to influence the wildfire régime (Fig. 2.10) are not clear. Reciprocal feedbacks with human activity add a layer of complexity on top of ecogeomorphic feedbacks, yet they must be identified and investigated if the relative importance of the physical and social causes of land degradation is to be adequately understood.

One approach to deal with the added layer of complexity introduced by considering human activity, whilst also transcending narrowly empirical approaches to examining human–environment interactions, is the use of Agent-Based Models (ABMs). Agent-based modelling provides a means to investigate complex feedbacks between individuals' context-dependent and heterogeneous decision-making and eco-geomorphic processes (Wainwright and Millington 2010). ABMs represent individuals or groups of actors in a virtual environment as agents linked by relationships which allow them to perform operations to perceive, produce, transform and manipulate objects according to rules (Ferber 1999). To produce integrated human-environment ABMs, interdependencies and feedbacks

are specified between agents and their environment, which is often represented by a cellular grid (Parker et al. 2003; Matthews et al. 2007). For example, the CYBEROSION model (Wainwright 2008) represents pre-historic cattle, pig, sheep, goat, deer and human actors as agents in a grid-based cellular environment which has characteristics including vegetation type and biomass, and soil thickness, texture, and moisture. Agents modify vegetation through consumption (for food), and vegetation cover in turn influences local rates of diffuse (splash, creep) erosion and physical weathering. Changes in land-surface (soil) conditions reciprocally affect agent behaviour via influences on vegetation, and the model has been shown to demonstrate emergent behaviour in spatial and temporal patterns of erosion due to these interactions (Wainwright 2008). More contemporary examples are the use of the SPASIM model to investigate feedbacks similar to those in Fig. 2.10 (Millington et al. 2008; Wainwright and Millington 2010) and the *CauxOpération* model to explore future potential erosive runoff in an agricultural landscape with multiple stakeholders (Souchère et al. 2010). Results from these studies indicate that ABMs can act as useful tools for examining the effects of land tenure and land-zoning policy on land degradation (via land-use decision-making, e.g. Muchena et al. 2005). Current drawbacks to the implementation of ABMs for breaking the hierarchical relationship are that these model types are computationally intensive and if not appropriately developed may approach levels of complexity similar to the real-world system they are representing (O'Sullivan 2004). Furthermore, although empirical methods for building agent-based models are being established (Robinson et al. 2007) and protocols for describing (Grimm et al. 2010), replicating (Wilensky and Rand 2007) and comparing (Polhill et al. 2008) models are becoming available, difficulties in validating, evaluating and interpreting ABMs remain (Windrum et al. 2007; Millington et al. 2011).

Participatory research approaches, which seek to draw on the knowledge and expertise of citizens and stakeholders not usually consulted by the traditional scientific method, have been suggested as one means to overcome these difficulties in ABMs specifically (e.g. Millington et al. 2011), but also in environmental research more broadly (Chilvers 2009). Motivations for pursuing participatory research may be normative, instrumental or substantive (Stirling 2006) and often arise from the recognition that stakeholders (local residents, farmers, land-use planners, etc.) have a form of non-certified, experience-based expertise about their environment that allows them to contribute to the knowledge-making activities of certified scientists (e.g. Collins and Evans 2002). One participatory approach that incorporates simulation modelling with role-playing games is 'Companion Modelling' (Barreteau 2003; Castella et al. 2005). Along with their *CauxOpération* model, Souchère et al. (2010) used this approach to improve understanding and facilitate negotiations on the future management of erosive runoff, via agricultural practices. In a similar vein, Hessel et al. (2009) used maps and GIS in participatory land-use use-planning workshops to investigate land-use problems and conflicts, their causes, effects and possible solutions. Hessel et al. showed that locals are knowledgeable about the driving forces behind land degradation and have ideas

about alternative land-use options to combat effects of degradation. Participatory research is a process of continuous learning that aims to produce knowledge and understanding that enables people to identify actions to change their lives and environment according to their own ideas (Demeritt 2005). If successful in these aims, participatory approaches, potentially incorporating modelling and simulation in the ‘companion’ approach, may be the ultimate means by which to shatter any hierarchical relationship between understanding physical and human causes of land degradation.

By using techniques and perspectives from systems science, agent-based modelling and participatory research, understanding is improving about how human and physical factors cause land degradation. However, many questions remain open:

- How do we investigate human-environment interactions across spatial and temporal scales?
- How do we validate, evaluate and test human-environment models?
- How do we account for initial system conditions, contingency and history when using models?
- How do we effectively implement participatory research approaches to minimize social, economic, political and cultural biases?

Furthermore, these approaches are inherently interdisciplinary and, as such, projects that intend to use them open themselves up to uncertainties possibly not present in reductionist, discipline-based research;

Engaging with other perspectives and approaches is not just demanding, but also risky too. . . . Progress in science has always come precisely from exposing ourselves to the possibility of getting it wrong or that things might not work out quite as planned. (Demeritt 2009, p. 9)

Thinking carefully about the specific needs for engagement with other disciplines, and about which of the various available approaches are most appropriate, will help to limit these risks. Confronting these risks and advancing beyond narrow scientific disciplinary specialization is vital if the hierarchical relationship in understanding physical and social causes of land degradation is to be broken.

2.10 Conclusion

This chapter has focused on both what we understand by the expression ‘land degradation’, with particular reference to drylands, how we currently assess it, some of the key land uses in drylands and their roles in land degradation, and the importance of land degradation for one of the global roles of drylands. The emphasis has been on how we conceptualize the process, what we know of it and why it is important. Subsequent chapters in this book will address in far greater detail what we know of the process, and consider different approaches and problems associated with understanding land degradation in drylands.

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